

Chapter 2: Climate, Disturbance, and Vulnerability to Vegetation Change in the Northwest Forest Plan Area

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Introduction

Climate change is expected to alter the composition, structure, and function of forested ecosystems in the United States (Vose et al. 2012). Increases in atmospheric concentrations of greenhouse gases (e.g., carbon dioxide [CO₂]) and temperature, as well as altered precipitation and disturbance regimes (e.g., fire, insects, pathogens, and windstorms), are expected to have profound effects on biodiversity, socioeconomics, and the delivery of ecosystem services within the Northwest Forest Plan (NWFP, or Plan) area over the next century (Dale et al. 2001, Franklin et al. 1991). The ecological interactions and diversity of biophysical settings in the region are complex. The effects of climate change on ecological processes will occur through a variety of mechanisms at a range of spatial scales and levels of biological organization, ranging from the physiological responses of individual plants to the composition and structure of stands and landscapes (Peterson et al. 2014a). Understanding and incorporating how climate change projections and the potential ecological effects and uncertainties differ within the region (e.g., Deser et al. 2012) is essential for developing adaptation and mitigation strategies.

Climate change has the potential to affect all ecological and socioeconomic components of the NWFP, as well as other objectives for federal forest managers in this region. However, climate change is only one factor that managers must consider when addressing conservation and other goals for the NWFP region. The overarching goal

of this chapter is to lay a general foundation of current knowledge and understanding of climate change for the subsequent chapters in this synthesis report, and not to analyze and report the projected effects of climate change on all the different components of the Plan in detail. The chapters that follow address the role of climate change in the context of their particular topics (e.g., northern spotted owls, aquatic ecosystems). This chapter focuses on the following topics:

- Regional climate setting, including an introduction to the major vegetation zones and disturbance regimes of the region (see chapter 3 for a more detailed discussion of disturbance regimes)
- Climate history of the region from the Holocene through the 20th century
- Overview of climate modeling approaches and limitations
- Projected changes in climate and how these vary across the region
- Mechanisms of vegetation change and potential climate change vulnerabilities
- Projected effects on vegetation at regional scales
- Uncertainties associated with models and knowledge of climate change effects
- Management considerations and strategies for adaptation and climate change mitigation goals. (See chapters 3 and 12 for a more complete discussion of management options)

This chapter does not address broader issues of NWFP ecological and socioeconomic goals in the context of climate change. These topics are covered in chapter 12, in which climate change is considered along with other factors (e.g., nonnative species, ecosystem vs. species approaches to conservation, and tradeoffs) in a discussion of the science underlying the goals of the NWFP and the 2012 planning rule. This chapter is also guided by questions from managers, as follows:

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Guiding Questions

This chapter addresses the following:

1. How did climate and vegetation change from the early Holocene to the late 20th century, and how did these changes vary across the NWFP area?
2. What are recent trends in climate change and how do they vary geographically across the NWFP area?
3. What are the major tools for projecting climate change and what are the associated uncertainties and limitations?
4. What changes in climate are projected for the NWFP area and how do these projections differ across the region?
5. What are the implications of recent and projected climate trends for vegetation change?
6. What are the mechanisms of vegetation change associated with climate change?
7. Which ecosystems and species are most vulnerable to climate change?
8. What are the key adaptation strategies that could reduce vulnerability to climate change?

Background and Setting

The NWFP area covers approximately 24.4 million ac (9.9 million ha) and includes multiple physiographic provinces across Washington, Oregon, and northern California (fig. 2-1). These physiographic provinces encompass a variety of disturbance regimes (see chapter 3 for more discussion and information) as well as a broad range of environmental and climatic gradients (fig. 2-2). Climate is cooler and wetter toward the north in the coastal and inland mountains, but transitions to a more Mediterranean climate with warmer, drier summers and greater interannual variability to the south (fig. 2-3). Most precipitation in the region falls during the winter months, often as snow at higher elevations. The Olympic Peninsula, Western Lowlands, and Coast Range are located in the western portion of the region. These receive the greatest annual precipitation and often experience a summer fog layer along the coast that can partially moderate

summer moisture stress. The crest of the Cascade Range extends from northern Washington to northern California, bisecting much of the region and creating a steep gradient in precipitation from west to east. The western Cascades encompass a wide range of elevations, temperatures, and precipitation, which generally decreases toward the south. The eastern Cascades extend in a narrow band from Washington to the California border and are generally much drier than the western Cascades and most of the NWFP area. The Klamath Mountains, in southwest Oregon and northwest California, represent the most climatically and geologically diverse province in the area, with a strong west-to-east gradient in precipitation and summer moisture stress. The Willamette Valley makes up a relatively small portion of the NWFP area and is predominantly nonforested.

The broad range of environmental and climatic gradients is reflected in the distribution of several potential vegetation zones across the region (figs. 2-1, 2-2, and 2-3) (Simpson 2013) (<https://www.ecoshare.info/category/gis-data-vegzones>). Potential vegetation zones represent climax vegetation types that would eventually develop in the absence of disturbance; therefore, existing or current vegetation varies often within zones depending on seral stage (i.e., successional stage or stage of structural development) and time since disturbance. For example, the most abundant vegetation zone in the NWFP area, western hemlock (*Tsuga heterophylla*), is currently dominated by Douglas-fir (*Pseudotsuga menziesii*). Vegetation zones provide an ecological framework for discussing climate and vegetation change across broad geographic extents (chapter 3). Vegetation zones have overlapping species pools but consist of unique plant community assemblages, as well as similar but internally variable biophysical conditions and historical disturbance regimes that differ geographically (Winthers et al. 2005; chapter 3). Vegetation zones have characteristic pathways of structural development that differ in complexity and reflect regional gradients in productivity as well as historical and contemporary disturbance regimes (Reilly and Spies 2015).

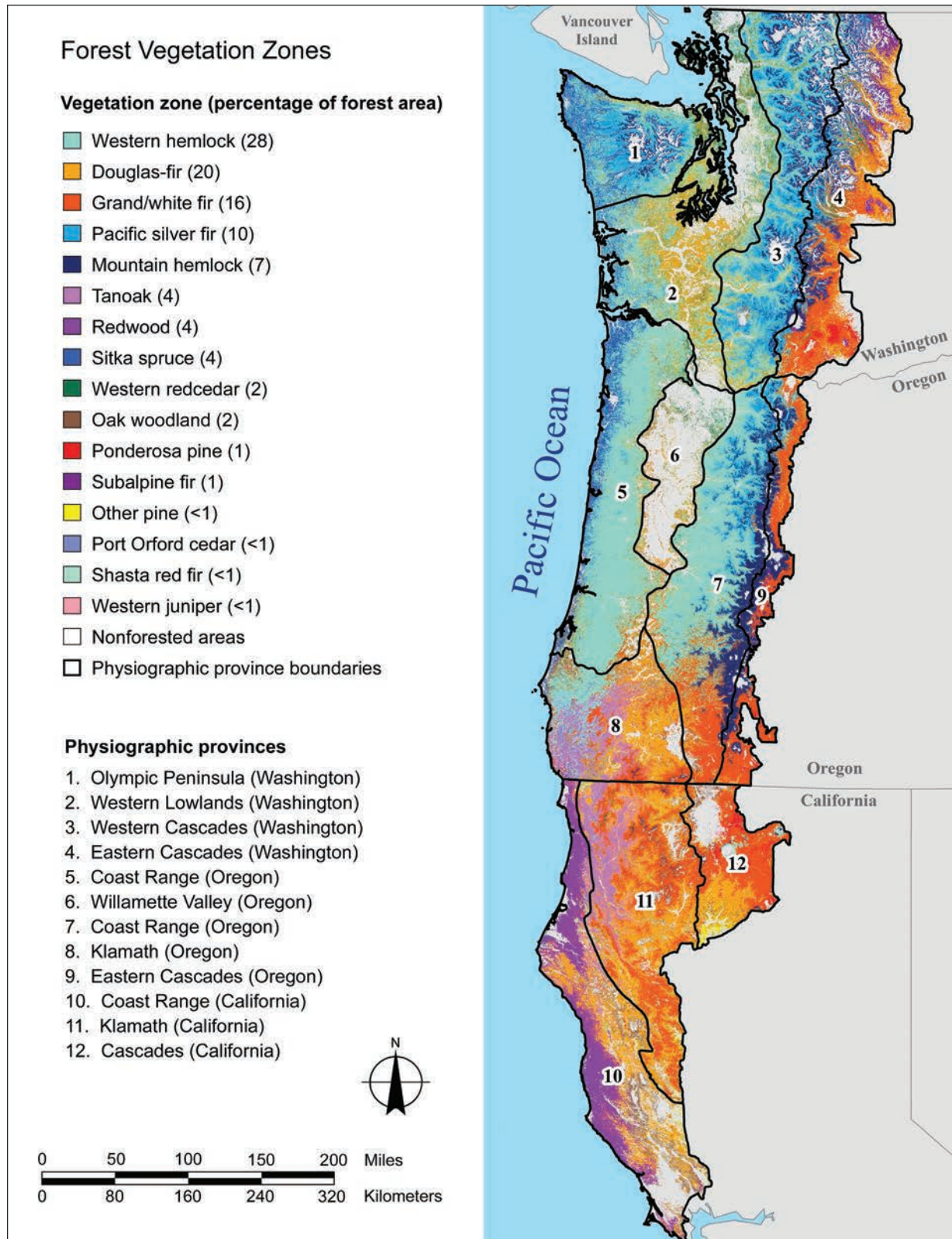


Figure 2-1—Geographic distribution of potential vegetation zones (Simpson 2013) and physiographic provinces within the Northwest Forest Plan area.

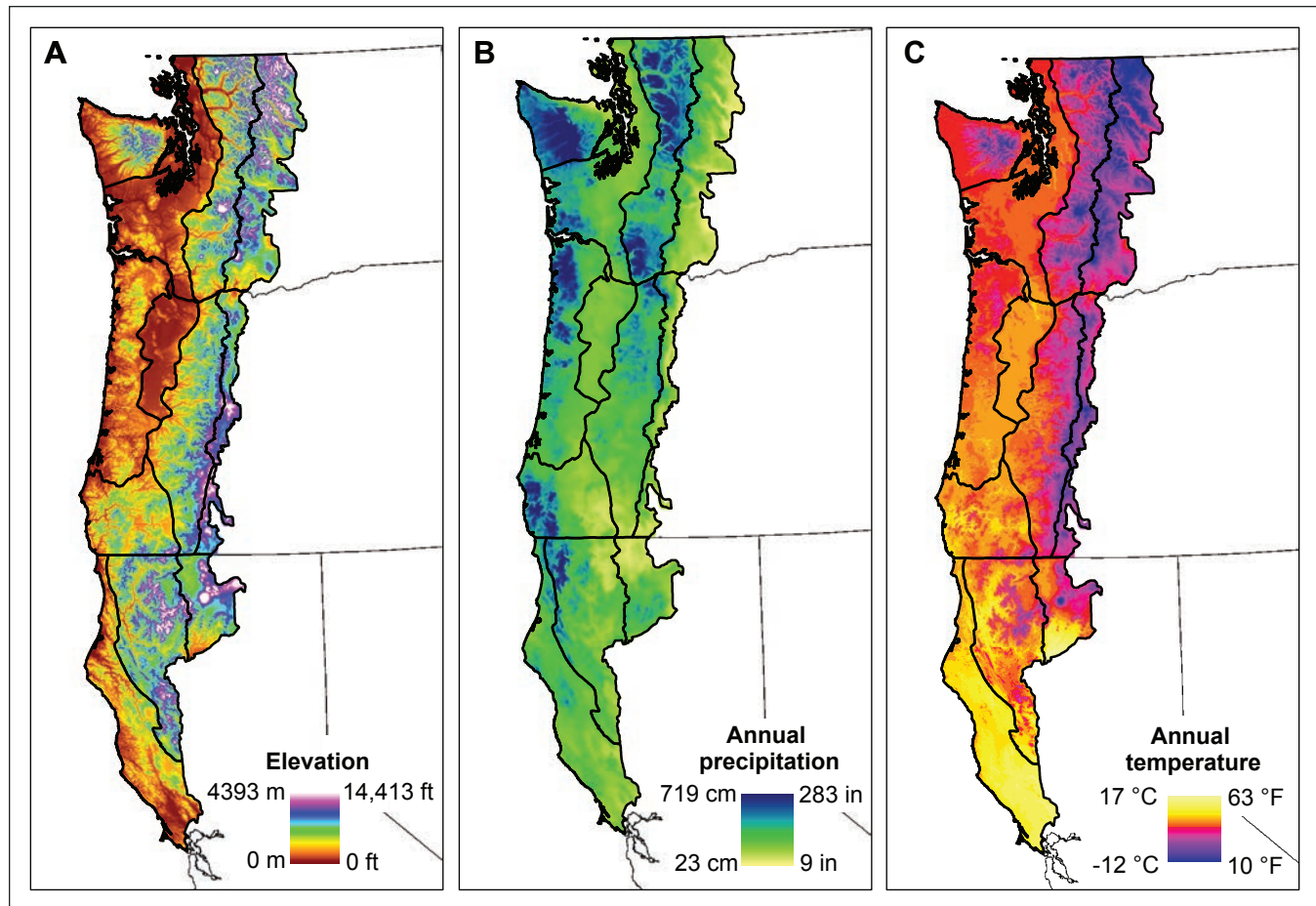


Figure 2-2—Maps of (A) elevation, (B) annual precipitation, and (C) annual temperature in the Northwest Forest Plan area. Temperature and precipitation are derived from 30 arc-second (~800 m) PRISM (parameter-elevation regressions on independent slopes model) (Daly et al. 2008) grids averaged from 1971 to 2000, and were obtained from the Landscape Ecology, Modeling, Mapping and Analysis group at Oregon State University. Darker lines outline physiographic provinces shown in figure 2-1; lighter black lines show state boundaries.

The major vegetation zones (figs. 2-1 and 2-4) of the region generally correspond to those presented by Franklin and Dyrness (1973) and were broken into moist and dry forests in the NWFP (chapter 3). This characterization is overly simplistic, as annual precipitation in any given zone varies geographically. Moist vegetation zones make up about 60 percent of the region, and are primarily located in coastal areas and west of the Cascade crest. These include Sitka spruce (*Picea sitchensis*), redwood (*Sequoia sempervirens*), tanoak (*Lithocarpus densiflorus*), western hemlock, western redcedar (*Thuja plicata*), Pacific silver fir (*Abies amabilis*),

and mountain hemlock (*Tsuga mertensiana*). Dry forest vegetation zones are located east of the Cascade crest, and also comprise a large portion of inland areas in southwest Oregon and northwest California. They include western juniper (*Juniperus occidentalis*), ponderosa pine (*Pinus ponderosa*), Douglas-fir, grand fir (*Abies grandis*) and white fir (*Abies concolor*), and subalpine forests dominated by subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmannii*), and whitebark pine (*Pinus albicaulis*). A more detailed and comprehensive characterization of plant communities in individual vegetation zones can be found in Franklin and Dyrness (1973).

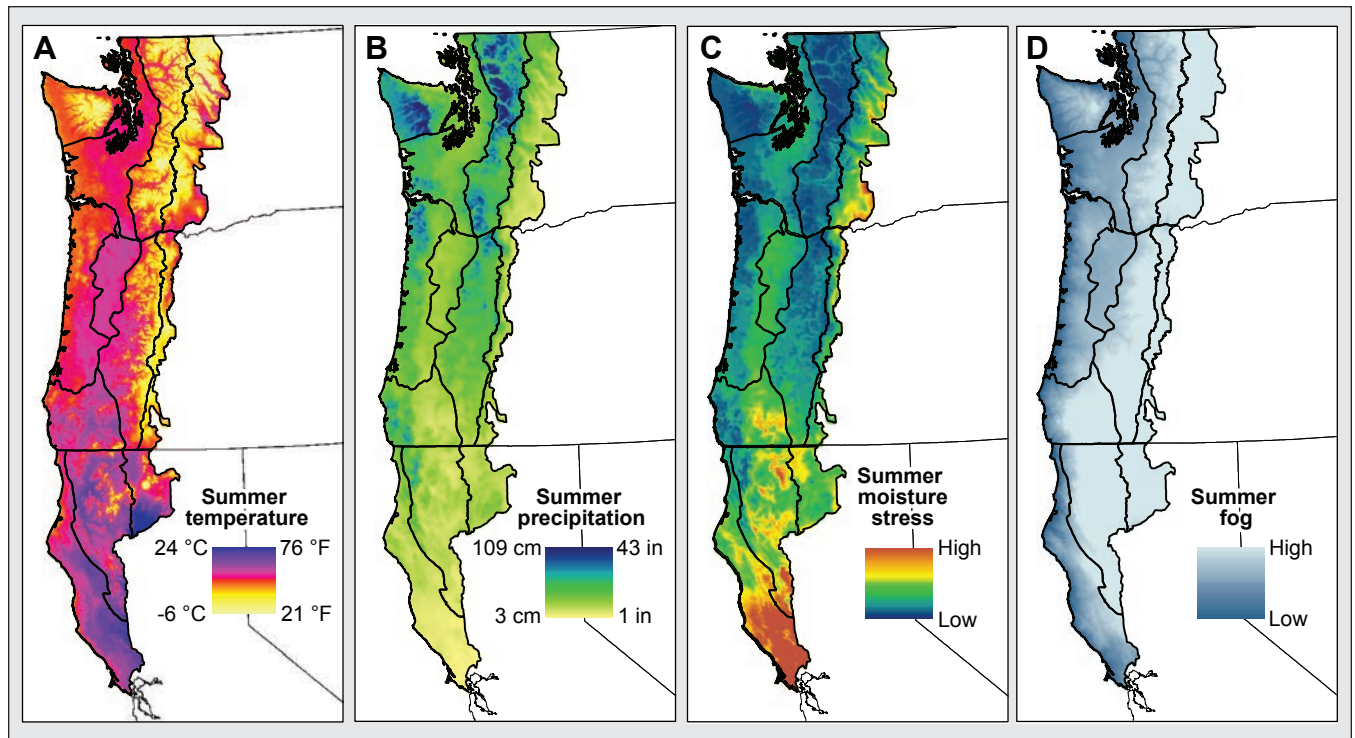


Figure 2-3—Maps of (A) mean summer temperature, (B) total summer precipitation, (C) summer moisture stress, and (D) summer fog in the Northwest Forest Plan area. Temperature and precipitation are derived from 30 arc-second (~800 m) PRISM (parameter-elevation regressions on independent slopes model) (Daly et al. 2008) grids averaged from 1971 to 2000, and were obtained from the Landscape Ecology, Modeling, Mapping and Analysis group at Oregon State University. Summer moisture stress was calculated by dividing summer temperature by summer precipitation for May through September. Summer fog is a proxy based on the optimal path length from coastline representing the easiest path of fog movement given topography and terrain blockage (Daly et al. 2008). Darker lines outline physiographic provinces shown in figure 2-1; lighter black lines show state boundaries.

More information on geographic variability and current vegetation in Oregon and Washington is available at Ecoshare (<https://www.ecoshare.info/publications>) and is discussed further in chapters 1, 3, and 12. Appendix 2-1 provides a crosswalk for linking equivalent vegetation types between the Simpson (2013) vegetation zones and existing vegetation in northern California based on Regional

Dominance 1 in the Pacific Southwest Region (Region 5) CALVEG database. This crosswalk provides a means of interpreting the Simpson vegetation zones in terms of existing vegetation in California. More details on the CALVEG database are available at <https://www.fs.usda.gov/detail/r5/landmanagement/resourcemanagement/?cid=stel-prdb5347192>.

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Figure 2-4—Examples of forests from several vegetation zones illustrating the broad range of environmental and biophysical settings in the Northwest Forest Plan area: (A) western hemlock, (B) redwood, (C) mountain hemlock, (D) subalpine fir, (E) grand fir/white fir, and (F) ponderosa pine.

Key Findings

Past Climate Change in the Northwest Forest Plan Area

The climate and vegetation of the NWFP area went through continuous change over the past 11,700 years during the Holocene. During this time, complex interactions between a fluctuating climate and fire drove vegetation change at millennial scales (Bartlein et al. 1998, Marlon et al. 2009, Walsh et al. 2015, Whitlock 1992, Whitlock et al. 2008). Species responded individually to changes in climate, sometimes forming assemblages that lack contemporary analogs (Whitlock et al. 2003). Species ranges expanded and contracted over time, with some species persisting in refugia where local conditions allowed persistence in regions where climate was generally unsuitable (Gavin et al. 2014). Refugia likely provided an important role in the persistence of populations through the numerous climatic transitions that occurred in the region since the last glacial maximum (Bennett and Provan 2008, Hampe and Jump 2011).

Knowledge of vegetation changes during the Holocene is particularly rich in the NWFP area, and a number of paleoecological studies document change across the region. The Holocene is commonly divided into different periods that can be distinguished by climate and fire activity. We follow the divisions of Walsh et al. (2015) in a recent review, though other studies use different dates to delineate periods, and the timing of changes in climate and vegetation differ across the NWFP area (Whitlock et al. 2003).

Paleoecological studies use charcoal and pollen found in sediment cores from lakes, as proxies for past climatic conditions, and to reconstruct changes in vegetation composition over time (Whitlock et al. 2003). These studies are limited in terms of their spatial and temporal precision, but offer important historical context and insight on climate and vegetation change by broadening our understanding of the historical range of variability at millennial time scales.

The early Holocene—approximately 12,000 to 8,000 years before present (BP)—was a time of rapid vegetation change, with assemblages that include current subalpine and lower elevation species that lack modern analogs

(Whitlock 1992). Increased summer insolation during this period led to higher summer temperatures and drier conditions than the present, while lower winter insolation led to cooler and wetter winters, likely amplifying seasonality and summer drought compared to present-day climate (Bartlein et al. 1998, Whitlock et al. 2001). Fire activity was relatively low at the beginning of the early Holocene, but increased and remained high until approximately 8,000 years BP (Briles et al. 2005, Walsh et al. 2015). Nonforested areas and open woodlands were replaced by forests as glaciers receded early in this period, and xerophytic species increased at many low-elevation sites across western Oregon and Washington as summers warmed (Walsh et al. 2015).

As the climate warmed during the early Holocene, species responded individually and became distributed along elevational and latitudinal gradients (Whitlock et al. 2003). Douglas-fir, red alder (*Alnus rubra*), and oak (*Quercus* spp.) replaced spruce and pine at lower elevations in the Coast Range and western Cascades (Cwynar 1987, Grigg and Whitlock 1998, Long et al. 1998, Sea and Whitlock 1995, Walsh et al. 2008). On the Olympic Peninsula, herbaceous tundra was replaced by subalpine fir (Gavin et al. 2001). Mid-elevations of the eastern Cascades of Oregon were dominated by open pine (*Pinus* spp.) forests, initially with an understory of *Artemisia*, which likely transitioned into a closed-forest environment with a greater abundance of *Abies* spp. Mid-elevations of the Klamath Mountains in Oregon and California were dominated by open woodlands composed of *Pinus* spp., *Quercus* spp., and incense cedar (*Calocedrus decurrens*) (Briles et al. 2005, Daniels et al. 2005, Mohr et al. 2000).

Cooler, wetter conditions were associated with decreasing summer isolation during the middle of the Holocene (~8,000 to 4,000 years BP) (Bartlein et al. 1998). During this time, fire activity decreased (Briles et al. 2005, Walsh et al. 2015), and modern species assemblages were formed in some parts of the region (Whitlock et al. 1992). Redcedar and western hemlock increased during this period across low- and middle-elevation forests of the Coast Range, the Cascade Mountains, and the Puget Trough (Cwynar

1987, Prichard et al. 2009, Walsh et al. 2008). Species composition shifted toward silver fir, mountain hemlock, and Alaska yellow-cedar (*Callitropsis nootkatensis*) on the Olympic Peninsula (Gavin et al. 2001). In the Klamath Mountains, expansion of *Pinus* spp., *Cupressaceae*, and *Abies* spp. also indicated cooler, wetter conditions during this period (Briles et al. 2005, Daniels et al. 2005, Mohr et al. 2000). With the exception of lower elevations, fire activity increased again approximately 5,500 years BP (Walsh et al. 2015).

Fire activity continued to increase during most of the late Holocene (~4,000 years BP to present) despite evidence that this period remained cool and moist (Bartlein et al. 1998, Walsh et al. 2015). There is little evidence in the pollen record to suggest major changes in the composition of vegetation assemblages across most of Oregon and Washington during this time (Walsh et al. 2008, 2015; Whitlock 1992). Modern forest assemblages in the Douglas-fir and white fir zones established approximately 2,000 years ago in the Klamath Mountains, where fire activity also increased during this time despite cool and moist conditions (Briles et al. 2005, 2008; Daniels et al. 2005; Mohr et al. 2000). Climate and fire fluctuated during the past 1,000 years. The warmest temperatures occurred during the Medieval Climate Anomaly (MCA) (900–1250 CE) and the coldest temperatures during the Little Ice Age (LIA) (1450–1850 CE) (Steinman et al. 2012). Precipitation also varied during this time, but there is less consensus about this in the literature. Cook et al. (2004) argued that a period of drought occurred during the MCA, but more recent evidence suggests a wet MCA and dry LIA (Steinman et al. 2014). Fire frequency increased during the MCA in the Klamath Mountains (Daniels et al. 2005, Mohr et al. 2000) as well as the rest of the region in Oregon and Washington (Walsh et al. 2015). Many of the currently existing old-growth forests in moist vegetation zones established at this time (chapter 3).

Climate fluctuations associated with surface temperatures in the Pacific Ocean also became more apparent over the past 1,000 years (Nelson et al. 2011). Warming and cooling of sea surface temperatures in the equatorial Pacific Ocean, referred to as the El Niño Southern Oscil-

lation (ENSO), result in periodic (2 to 7 years) anomalies that affect regional air temperature and precipitation. During the El Niño phase, winter and spring conditions are generally warmer and drier than average (McCabe and Dettinger 1999). During the opposite La Niña phase, winter and spring are generally wetter and cooler, leading to a deeper than average snowpack (Gershunov et al. 1999). The Pacific Decadal Oscillation (PDO) is defined by fluctuations in sea surface temperature in the Pacific Ocean and has longer characteristic periodicity of 20 to 30 years (Mantua et al. 1997), although the PDO is not consistent over time at these frequencies (McAfee 2014) and has exhibited variable regime transitions in the pre-instrumental period (Gedalof and Smith 2001). Newman et al. (2016) pointed out that the PDO is not an independent phenomenon, but a combination of multiple processes that include ENSO. The relationship between ENSO and PDO is weaker in northern California where the respective controls of ENSO and PDO on climate are less predictable (Wise 2010).

Fire History

Regional drought driven by teleconnections with sea surface temperature anomalies (e.g., PDO, ENSO) resulted in synchronous occurrence of fires in the NWFP area (Hessl et al. 2004, Trouet et al. 2006, Weisberg and Swanson 2003, Wright and Agee 2004), as well as elsewhere in the Pacific Northwest and other regions of the Western United States (Heyerdahl et al. 2008, Kitzberger et al. 2007, Schoennagel et al. 2005). Several fire history studies document fire frequency over the past 400 years (table 2-1). Historical fire regimes differed among individual vegetation zones as well as geographically within vegetation zones (see chapter 3 for an indepth discussion). Fire was generally infrequent in most moist vegetation zones but ranged from about 50 years to >200 years, with synchronous, regional fire episodes occurring across the region from the 1400s to the mid 1600s, and again from the early 1800s to approximately 1925 (Weisberg and Swanson 2003). Fire was far more frequent in dry vegetation zones, where return intervals were shorter, generally ranging from 10 to 50 years until the late 19th and early 20th century.

Table 2-1—Fire history studies in the Northwest Forest Plan area by vegetation zone

Vegetation zone	Study	Extent (time period)	Method	Frequency/ return interval	Rotation	Low/moderate/ high	High-severity patch size
		Hectares		Years	Years	Percent	Hectares
Redwood:	Stuart 1987	300 (1898–1940)	Scars	7.8	—	—	—
	Finney and Martin 1989	~600 (1300–1860)	Scars	10.1	—	—	—
	Brown and Swetnam 1994	<1000 (171–1962)	Scars	9.9	—	—	—
	Brown et al. 1999	Unknown	Age, scars	7–13	—	—	—
	Brown and Baxter 2003	20 316 (1550–1930)	Scars	6–20	—	—	—
	Stephens and Fry 2005	~1000 (1615–1884)	Scars	12	—	—	—
Western hemlock:	Means 1982	Unknown	Scars	100	—	—	—
	Fahnestock and Agee 1983	Western Washington (pre-1934)	Age class from historical survey records	—	598	—	—
	Stewart 1986	<1 (~1200–1982)	Age, scars	50 ^a	—	—	—
	Yamaguchi 1986	Unknown (post-1480)	Age, scars	40–150	—	—	—
	Teensma 1987	11 000 (1482–1952)	Age, scars	114	78	—	—
	Agee et al. 1990	3500 (1573–1985)	Age, scars	137	—	—	—
	Morrison and Swanson 1990	1940 (1150–1985)	Age, scars	96	95	0–86/0–60/ 0–100	<110 ha
	Garza 1995	3540 (pre-1910)	Age, scars	93–158	134	24–41/9–23/ 25–54	—
	Impara 1997	~140 000 (1478–1909)	Age, scars	85	271	—	—

Table 2-1—Fire history studies in the Northwest Forest Plan area by vegetation zone (continued)

Vegetation zone	Study	Extent (time period)	Method	Frequency/ return interval	Rotation	Low/moderate/ high	High-severity patch size
		<i>Hectares</i>		<i>Years</i>	<i>Years</i>	<i>Percent</i>	<i>Hectares</i>
Silver fir:	Wetzel and Fonda 2000	2500 (1400–1985)	Age, growth release	21.3 ^b	—	—	—
	Agee and Krusemark 2001	26 000 (pre-1900)	Age, live residual structure from air photos	—	296	7–9/18–31/ 62–90	—
	Robbins 1995	~1562 km ² (1700–1990)	Age, scars	49 (2–191)	—	—	—
	Olson and Agee 2005	~7000 (1650–1900)	Age, scars	2–167	—	—	—
	Weisberg 2009	14 504 (1550–1849)	Age, scars	—	162	—	—
	Wendel and Zabowski 2010	1873 (1568–2007)	Age, scars	127	140	—	—
	Hemstrom and Franklin 1982	~53 000 (1200–1850)	Age	—	465	—	—
	Fahnestock and Agee 1983	Western Washington (pre-1934)	Age class from historical survey records	—	834	—	—
	Agee et al. 1990	3500 (1573–1985)	Age, scars	108–137	—	—	—
	Morrison and Swanson 1990	1940 (1150–1985)	Age, scars	239	149	0–80/0–78/ 0–100	<50
Mountain hemlock:	Garza 1995	3540 (pre-1910)	Age, scars	154–246	—	24–57/ 20–22/45–50	—
	Dickman and Cook 1989	18 000 (post-1400)	Age	—	—	—	>3200
	Fahnestock and Agee 1983	Western Washington (pre-1934)	Age class from historical survey records	—	598	—	—

Table 2-1—Fire history studies in the Northwest Forest Plan area by vegetation zone (continued)

Vegetation zone	Study	Extent (time period)	Method	Frequency/ return interval	Rotation	Low/moderate/ high	High-severity patch size
		<i>Hectares</i>		<i>Years</i>	<i>Years</i>	<i>Percent</i>	<i>Hectares</i>
Subalpine:	Agee et al. 1990	3500 (1573–1985)	Age, scars	137	—	—	—
	Fahenstock and Agee 1983	Western Washington (pre-1934)	Age class from historical survey records	—	800	—	—
	Agee et al. 1990	3500 (1573–1985)	Age, scars	109	—	—	—
Douglas-fir and grand fir/white fir:	Leiberg 1903	Southern Oregon (~1900)	Historical land survey	—	—	—	~14 000
	Weaver 1959	Unknown	Scars	47	—	—	—
	Agee et al. 1990	3500 (1573–1985)	Age, scars	52–93	—	—	—
	Agee 1991	197 (1760–1930)	Age, scars	16	37–64	—	—
	Bork 1984	~100 (pre-1900)	Scars	8	—	—	~400
	Wills and Stuart 1994	~20 (1745–1849)	Age, scars	10.3–17.3	—	—	—
	Taylor and Skinner 1998	1570 (1627–1849)	Age, scars	14.5	19	59/27/14	—
	Van Norman 1998	45 000 (1480–1996)	Age, scars	123	—	—	—
	Brown et al. 1999	2000 (1820–1945)	Age, scars	7.7–13	—	—	—
	Everett et al. 2000	3240–12 757 (~1700–1860)	Scars	6.6–7	11–12.2	—	2.4–40
	Stuart and Salazar 2000	~120 (1614–1944)	Age, scars	27 (12–161)	—	—	—

Table 2-1—Fire history studies in the Northwest Forest Plan area by vegetation zone (continued)

Vegetation zone	Study	Extent (time period)	Method	Frequency/ return interval	Rotation	Low/moderate/ high	High-severity patch size
		Hectares		Years	Years	Percent	Hectares
	Taylor and Skinner 2003	2325 (pre-1905)	Age, scars	11.5–16.5	19	—	—
	Wright and Agee 2004	~30 000 (1562–1995)	Scars	19–24	—	—	10–100
	Hessburg et al. 2007	~72 000 (~1930)	Historical aerial photos	—	—	18/58/24	~10 000
	Baker 2012	140 400 (~1770–1880)	Live structure from historical inventory	—	496 ^c	18/59/23	—
Ponderosa pine:							
	Weaver 1959	Unknown	Scars	11–16	—	—	—
	Soeriaatmadja 1966	(1500–5000) Unknown	Scars	3–36	—	—	—
	West 1969	Unknown	Age	—	—	—	<0.26
	Bork 1984	~100 (pre-1900)	Scars	4–7	—	—	—
	Morrow 1985	2 (pre-1900)	Age	—	—	—	<0.35
	Hessburg et al. 2007	~106 000 (1930–1940)	Live structure from historical aerial photos	—	—	30/58/12	—
	Baker 2012	123 500 (~1770–1880)	Live structure from historical inventory	—	705 ^c	40/44/16	—

^a Stewart noted 15 fires over a 750-year period.^b Estimated at a 200-ha scale.^c Rotation for high severity only.

Note: Most fire history studies are based on fire scars or identification of cohorts of trees with similar establishment dates. Fire frequency or return interval are the most commonly reported metrics of fire activity in fire history studies. Another metric related to fire frequency is fire rotation, or the time it takes to burn an area equal to the size of the area of interest. Relatively few studies report fire severity.

20th-Century Climate Change in the Northwest Forest Plan Area

Increases in temperature and precipitation across the NWFP area during the 20th century exceeded average global increases and vary across the region as well as among seasons (Abatzoglou et al. 2014b, Mote 2003). Most of the research examining 20th-century climate in the Plan area has been aggregated to the scale of individual states (i.e., California, Oregon, and Washington), or summarized for the entire Western United States, and there is less work that focuses specifically on the Plan area. There is evidence supporting both strong human-caused climate change (Abatzoglou et al. 2014a, 2014b) and temperature increases associated with ocean/atmospheric variability (Johnstone and Mantua 2014a, 2014b). However, Abatzoglou et al. (2014a) demonstrated that natural factors alone cannot explain warming in the region.

Average annual temperature in western Oregon and Washington increased by 1.6 °F (0.91 °C) during the 20th century, with the greatest increase of 3.3 °F (1.83 °C) occurring during winter (Abatzoglou et al. 2014b, Mote 2003). Likewise, precipitation during the same period also increased by 13 percent, with the greatest increase of 37 percent during spring (Abatzoglou et al. 2014b, Mote 2003). California also experienced accelerated warming since 1970 (Cordero et al. 2011) and recently experienced the hottest, driest period (2012 to 2014) in the observational record (Mann and Gleick 2015). This same period also includes the lowest precipitation in recorded history (Diffenbaugh et al. 2015) and potentially in the past 1,200 years (Griffin and Anchukaitis 2014). In northwestern California, Rapacciuolo et al. (2014) estimated that mean temperature increased by 0.3 °F (0.18 °C). The same study estimated that minimum temperature increased by 0.9 °F (0.47 °C) and maximum temperature decreased by 0.4 °F (0.24 °C) during the 20th century, although these trends were calculated using temporal differencing rather than traditional slope-based trends, and do not necessarily account for differences in the density of weather stations used in the study (Rapacciuolo et al. 2014). Twentieth-century trends in precipitation differed across northern California with evidence of overall increases (Killam et al. 2014) as well as slight decreases in some parts of the NWFP area (Rapacciuolo et al. 2014).

Climate trends across the region are similar to those reported from studies across the Western United States. These studies indicate changes in several characteristics of weather relevant to forest and vegetation dynamics. Spring (March to May) temperature increased approximately 1.8 °F (1 °C) from 1950 to 1998 (Cayan et al. 2001) and snowpack declined during the latter half of the 20th century (Knowles 2015, Mote et al. 2005). Increases in winter temperature are linked with decreases in snowpack (Mote 2006) and earlier snowmelt, which have altered streamflow timing (Hamlet et al. 2005; Jung and Chang 2011; Stewart et al. 2004, 2005). Decreases in the proportion of annual precipitation falling as snow (Klos et al. 2014), the amount of water contained in spring snowpack (i.e., the depth of water if the snow were to melt) (Hamlet et al. 2005), and increased evapotranspiration from longer growing seasons increased soil water deficits since the 1970s (Abatzoglou et al. 2014b). A longer freeze-free season, an increase in the temperature of the coldest night of the year, and increased potential evapotranspiration during the growing season also occurred during this period (Abatzoglou et al. 2014b). Fog frequency along the coast of northern California declined by 33 percent during the 20th century (Johnstone and Dawson 2010), as has low summertime cloudiness (Schwartz et al. 2014). Most recently, northern California experienced a dramatic shift with extreme drought conditions from 2012 to 2016 followed by extreme precipitation events and severe flooding (Wang et al. 2017). Remote-sensing studies indicate that most vegetation zones across the NWFP area have already experienced moisture stress associated with drought and high temperatures during the early 21st century across the entire NWFP area (Asner et al. 2016, Cohen et al. 2016, Mildrexler et al. 2016).

Projecting Climate Change for the 21st Century

Atmosphere-ocean general circulation models (GCMs) are the primary tools for projecting future climate scenarios (e.g., IPCC 2014). GCMs incorporate interactions among several important components of the Earth's climate system, including atmosphere, land, ice, and ocean to simulate past and future climate at relatively coarse spatial scales (~0.25 to 14 mi² [~0.65 to 36.3 km²]) based on different scenarios of increasing greenhouse gas concentrations in the

atmosphere. Because of differences in model formulation and sensitivity to forcing from physical influences on the atmosphere (e.g., greenhouse gases), GCM projections using the same initial conditions and emissions scenario differ (Lynn et al. 2009), as do projections from the same GCM owing to natural climate variability within a region (Deser et al. 2014).

An ensemble of projections (combinations of projections from multiple GCMs) is commonly used in climate change studies to capture the range and patterns of variability among projections. Ensemble averages appear to provide the best estimates of observed climate (Pierce et al. 2009, Rupp et al. 2013). The range of projections in an ensemble also provides a measure of the amount of uncertainty, which increases as projections extend farther into the future (Tebaldi and Knutti 2007). Uncertainty in climate change projections can be attributed to three main factors: (1) climate change-scenario uncertainty, (2) model-response uncertainty, and (3) natural variability in climate (Hawkins and Sutton 2009). For a given climate change scenario, uncertainty in the warming estimates arises from differences in GCM formulation and parameterization. Natural climate variability presents the greatest uncertainty in the near to mid term for projecting climate change for the first half of the 21st century (Hawkins and Sutton 2009) and poses a major challenge for analyzing and communicating climate change variability within a region (Deser et al. 2012).

For its fifth and most recent assessment (AR5), the Intergovernmental Panel on Climate Change published a set of future scenarios that describe estimated trajectories of greenhouse gas concentrations. These scenarios are called representative concentration pathways (RCP), and each scenario is named after the increase in radiative forcing relative to preindustrial levels. Each pathway is the result of plausible future trends in human population growth, economic and technological development, and energy systems, as well as social beliefs and values that affect human behaviors influencing emissions and climate warming (van Vuuren et al. 2011). Climate change scenarios (e.g., climate changes that are likely given a specific RCP) are considered to be plausible and do not have probability

distributions associated with them (Collins et al. 2014). Current rates of greenhouse gas emissions have exceeded previously anticipated concentrations, thus there is currently insufficient information to rule out any scenario (Manning et al. 2010, van Vuuren et al. 2010). All scenarios project increases in global mean temperatures, but there is a large range among the scenarios bracketing the low and high ends of potential greenhouse gas concentrations. Under the RCP 2.6 scenario, which represents strong mitigation action, global mean temperatures are projected to increase by $2.9^{\circ}\text{F} \pm 0.7^{\circ}\text{F}$ ($1.6^{\circ}\text{C} \pm 0.4^{\circ}\text{C}$) by the end of the century, while under RCP 8.5, the no-mitigation, high-growth scenario, the degree of warming is projected to be $7.7^{\circ}\text{F} \pm 1.3^{\circ}\text{F}$ ($4.3^{\circ}\text{C} \pm 0.7^{\circ}\text{C}$) (Collins et al. 2014). Changes in global precipitation are projected to increase 0.5 to 4 percent/ $^{\circ}\text{C}$ under RCP 2.6 and by 1 to 3 percent/ $^{\circ}\text{C}$ under other scenarios (Collins et al. 2014).

Many relevant studies, especially in northern California, use an earlier generation of climate change scenarios published in the Special Report on Emissions Scenarios (Nakicenovic and Swart 2000). In this set of scenarios, the A2 scenario represents a very heterogeneous world with continuously increasing global population. The B1 scenario represents a convergent world in which population peaks mid-century, then declines, transitioning to resource-efficient technologies. The B2 scenario is intermediate between A2 and B1, with population growth lower than the A2 and a less rapid transition to resource-efficient technologies.

21st-Century Climate Change Projections for the Northwest Forest Plan Area

Analysis of GCM projections for Oregon and Washington (Mote et al. 2014) and northern California (Cayan et al. 2008, 2016; Garfin et al. 2014) depict a future with significant warming by the end of the 21st century, although the magnitude of warming varies at finer scales across the region. In Oregon and Washington, Dalton et al. (2013) projected increases in annual average temperature of 4.3°F (2.4°C) and 5.8°F (3.2°C) by the middle of the century (2041 to 2070) under RCP 4.5 and RCP 8.5 scenarios, respectively. By the end of the century (2070 to 2099),

average annual temperature is projected to warm by 5.9 °F (3.3 °C) to 17.5 °F (9.7 °C), depending on the scenario (Mote et al. 2014). Warming is projected to occur across all seasons, with the greatest temperature increases occurring during summer months (Dalton et al. 2013).

Projected changes in precipitation are more uncertain in Oregon and Washington. Some models project a 10 percent decrease in annual precipitation by the end of the century (2070 to 2099) while others project as much as an 18 percent increase in precipitation (Mote et al. 2014). GCMs generally project wetter winters and drier summers (Dalton et al. 2013). Under the A2 and B2 scenarios, no-analog temperature conditions are projected by 2100 across much of the western Cascades and Klamath Mountains compared with those occurring in the recent past (Saxon et al. 2005). Under RCP 8.5, most of Oregon and Washington are projected to depart from their historical climate regime by 2050, when the mean annual temperature of a given location will exceed the 20th-century range of variability (Kerns et al. 2016).

In northern California, under the mitigation-oriented B1 scenario, annual temperature is projected to increase by 2.7 °F (1.5 °C) by 2100, and, under the high-growth A2 scenario, the increase is projected to be 8.1 °F (4.5 °C) (Cayan et al. 2008). Simulations depict drier futures under the B1 and A2 scenarios, with total annual precipitation decreasing by 18 percent in the more extreme A2 scenario (Cayan et al. 2008). Increases in temperature are projected for all seasons across northern California, with the greatest increases occurring during summer months (Cayan et al. 2008). Projected decreases in summer precipitation range from 4 to 68 percent, whereas projected changes in precipitation during winter months range from a 9 percent decrease to a 4 percent increase. More recent projections of increases in winter precipitation using the RCP 8.5 scenario show a high degree of agreement among models (Neelin et al. 2013). Interannual variability is expected to increase with the occurrence of greater wet and dry extremes during the wet season (October to March) (Berg and Hall 2015). Most of northern California is projected to depart from its 20th-century climate by the year 2040 (Kerns et al. 2016). The projected future climate in the Klamath Mountains

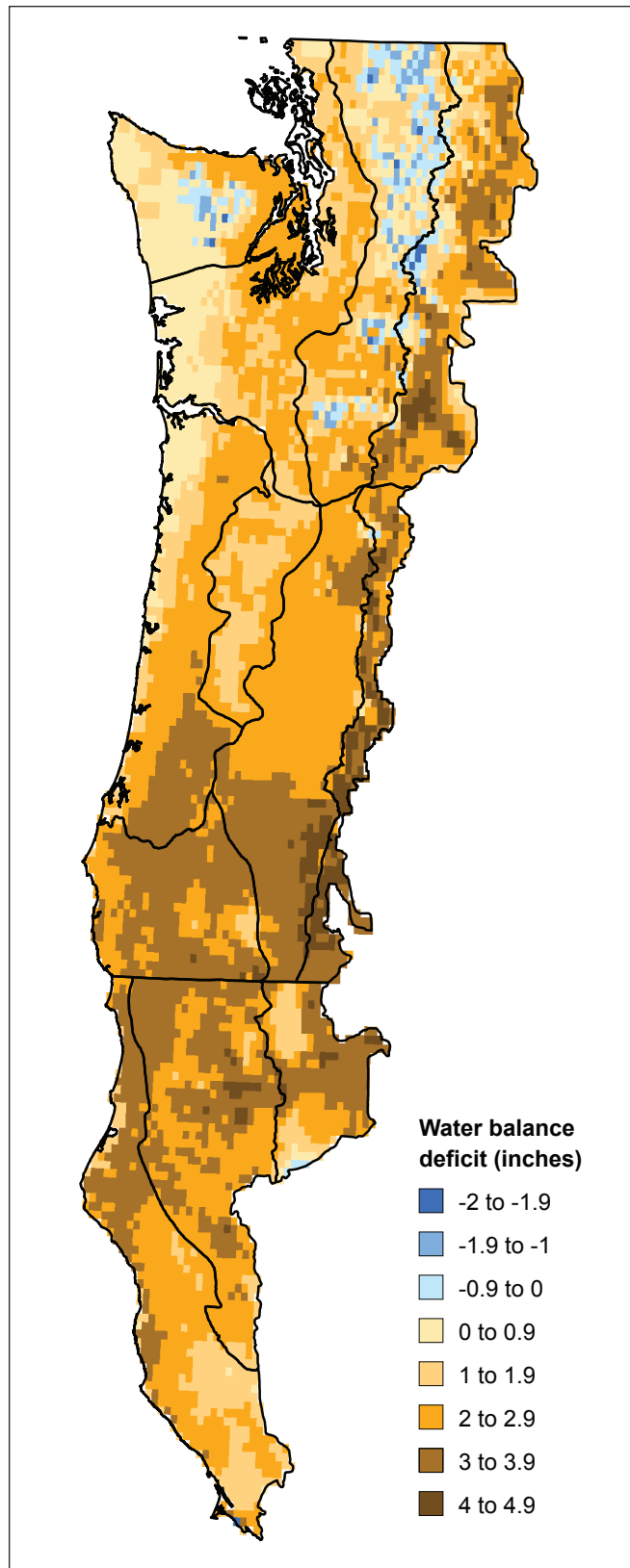
represents conditions of temperature and precipitation not experienced in the recent past by 2100 under the A2 and B2 scenarios (Saxon et al. 2005). Temperature is projected to depart the 20th-century range of variability between 2046 and 2065 under the A2 scenario (Klausmeyer et al. 2011).

Implications of Observed Climate Trends for Water Balance Deficit and Vegetation Change

Changes in the magnitude and seasonality of temperature and precipitation patterns will most likely affect vegetation by altering the availability of water in the soil. Cumulatively, these are expected to be experienced ecologically through hotter periods of drought and greater deficits in water balance. Water-balance deficit for vegetation is defined as the difference between potential evaporation and actual evapotranspiration (Stephenson 1998). Ecologically, the water-balance deficit equates to the difference between the atmospheric demand for water from vegetation and the amount of water that is actually available to use. Even if precipitation remains similar to 20th-century levels, projected increases in temperatures could reduce the amount of soil moisture available for plants.

Projections for changes in water-balance deficits differ among models (Littell et al. 2016) and across the region (fig. 2-5). The majority of the region is projected to experience an increased summer (June, July, August, and September) water-balance deficit during the middle part of the 21st century. The eastern Cascades, Klamath Mountains, and southern portion of the western Cascades in Oregon will likely experience the greatest increases in water-balance deficit, as well as the southeastern portion of the Oregon Coast Range and the northern portion of the California Coast Range. The least amount of change is projected in the northern portions of the Coast Range along the Pacific Ocean. Higher elevations of the Olympic Peninsula and the northern portion of the western Cascades in Washington are projected to experience less summer water-balance deficit in the future.

Although trends in average temperature and precipitation provide some context for vegetation change in the future, individual weather events are also expected to be important drivers of future dynamics (Jentsch et



al. 2007). Climate extremes (e.g., acute drought) related to changes in the variability of temperature and precipitation may have disproportionate effects on vegetation and result in rapid vegetation change (e.g., Allen and Breshears 1998). Increased frequency and intensity of heat waves and extreme temperatures are predicted across North America by the end of the 21st century (Meehl and Tebaldi 2004). Prolonged heat waves (Bell et al. 2004), as well as dry daytime and humid nighttime heat waves, are projected in northern California (Gershunov and Guirguis 2012). Models project increases in the number of both dry days and very heavy precipitation days during the wet season in northern California (October to March) (Berg and Hall 2015). This is consistent with an intensified water cycle characterized by shifts from extreme drought to years with anomalously high precipitation (Wang et al. 2017). Increases in peak flow magnitudes also suggest greater potential for flooding in portions of inland northern California (Das et al. 2013), where floods may be more frequent and severe (Dettinger 2011, Salathé et al. 2014). Heavy precipitation events from warming and shifts in seasonal precipitation patterns may also increase flooding in most of Oregon and Washington (Tohver et al. 2014) and the northern California Coast Range (Kim 2005). Rain-on-snow events may also be more common given warmer winter and spring temperatures, which are also projected to alter the timing of seasonal streamflow (Elsner et al. 2010). The availability of regional climate model outputs provides the climatic basis for better simulating physically consistent extremes relevant to forests processes (e.g., McKenzie et al. 2014, for fires), but these outputs are also subject to the constraints of GCMs used as boundary conditions.

Figure 2-5—Projected changes in summer (June, July, August, and September) water-balance deficit across the Northwest Forest Plan area for 2030–2059 from a composite of the 10 best general circulation model projections based on the CMIP3/AR4 scenarios following Littell et al. (2016). Higher water-balance deficit (browns) means decreased water available for plant uptake. Change is compared to the water-balance deficit from 1916 to 2006. Map boundaries correspond with the physiographic provinces in figure 2-1.

Considering the coarse resolution of climate projections (~ 0.25 to 14 mi^2 [~ 0.65 to 36.3 km^2]), it is important to recognize the potential for landscape-scale variability in future climate and vegetation change. Differences in vegetation structure and topography can drive fine-scale variation in temperature extremes, with differences in maximum and minimum temperatures of similar magnitude to those projected at a broader scale in different climate change scenarios (Suggitt et al. 2011). Spatial variability in bedrock geology also has the potential to mediate seasonal changes in groundwater availability associated with increased temperature (Tague et al. 2008). Complex topography and cold air pooling may decouple climate conditions in

mountain valleys from the surrounding landscape (fig. 2-6) (Daly et al. 2009), and snow may persist later in the season in canopy gaps and topographic depressions (Ford et al. 2013). Temperature is generally lower and soil moisture higher in interior late-successional forests than in clearcuts or edges (Chen et al. 1993), and denser canopies can attenuate warming by providing shade to the forest floor (De Frenne et al. 2013). Recent findings also indicate that dense, old-growth forests in moist vegetation zones of the region have the potential to provide cooling effects at local scales (Frey et al. 2016). Thus, the actual changes in future climate experienced by an organism may differ depending on their tolerances or habitat preference.

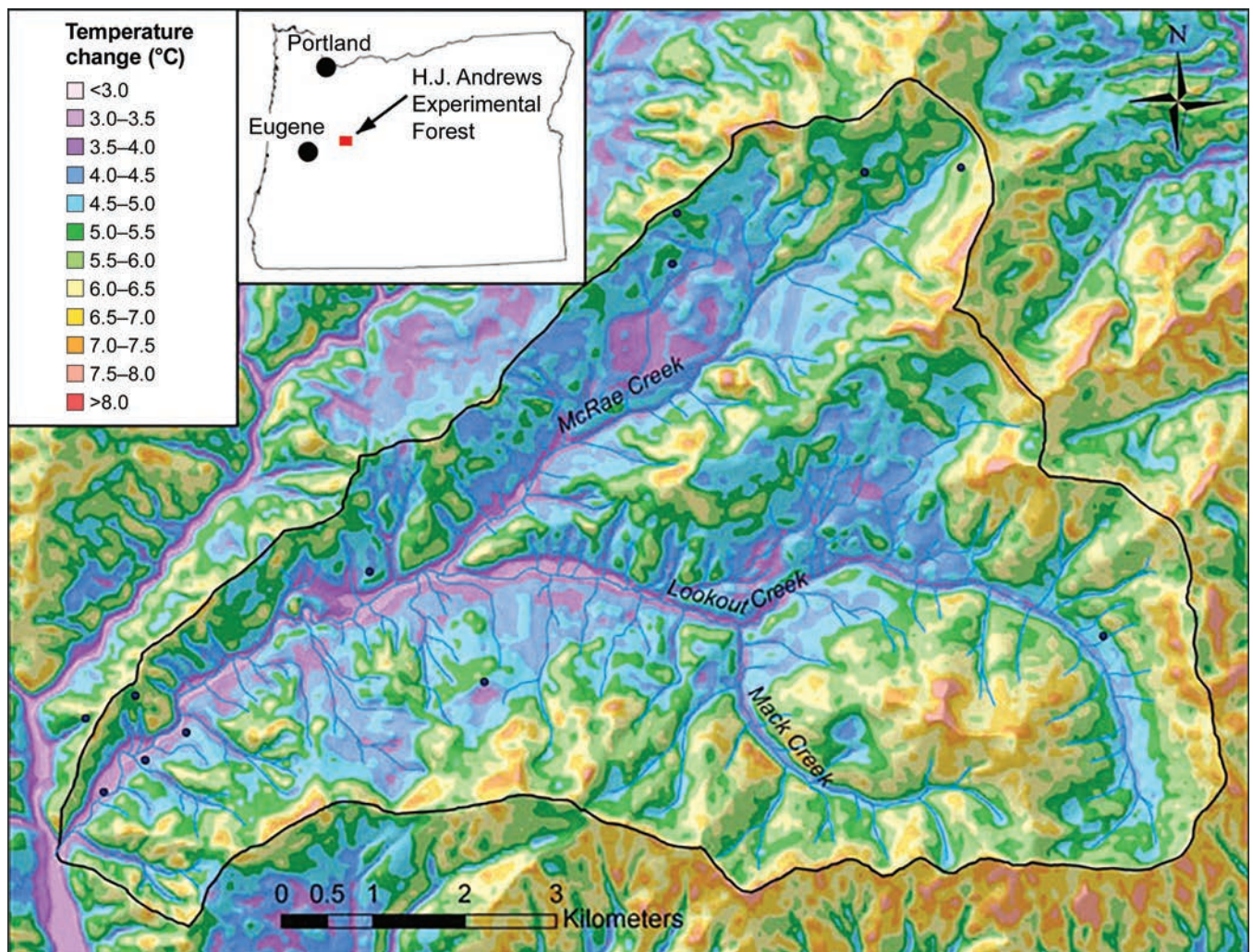


Figure 2-6—Projected changes in maximum December temperatures in response to a 2.5 °C regional temperature increase and changes in atmospheric circulation patterns in the western Cascade Range of Oregon. Source: Daly et al. (2010).

The potential for relatively stable climatic conditions at finer scales in some landscapes (e.g., topographically complex, mountainous terrain) suggests an important role for climatic refugia to contribute to the future persistence of some species (Noss 2001). Despite the conceptual appeal and historical importance of refugia, identification of refugia has proven difficult and has been largely descriptive, and refugia are likely to be species- and process-specific (Keppel et al. 2012). Refugia will most likely be found in topographically complex landscapes where microclimates differ because of differences in aspect, shading and insolation, and cold-air drainages (Dobrowski 2011). These areas may provide potential for species persistence through unfavorable climatic conditions, as well as sources for future recolonization provided that suitable conditions return in the future.

Mechanisms of Vegetation Change

Climate change is expected to alter vegetation through direct effects (e.g., from CO₂ and climate on vegetation processes) and indirect effects (e.g., from disturbance processes). The direct effects of climate change and increasing CO₂ on vegetation are expected to be expressed through changes in mortality, growth, and reproduction, all of which may be sensitive to altered phenology and biotic interactions within and among species (Peterson et al. 2014a). The indirect effects of climate change are expected to be expressed through increases in the frequency, severity, and extent of disturbances, particularly drought, fire, insects, and pathogens. These have the potential for rapid ecological change at landscape scales, and are predicted to be a greater driver of ecological change than direct effects (Dale et al. 2001, Littell et al. 2010). The relative importance of these drivers, however, is likely to vary geographically across the region among species, seral stages, physiographic provinces, and disturbance regimes. Species are expected to respond individually to future changes in climate as they have in the past (Whitlock 1992).

Direct effects of climate change: demographic responses—

Tree mortality from higher temperatures and drought stress has already occurred in many forests of the Western United States, and is expected to increase in the 21st century (Allen et al. 2010, 2015). Warmer temperatures and increased

frequency and duration of droughts projected for the NWFP area are likely to increase climate-induced physiological stress on plants (Adams et al. 2009). Drought-related stress can lead to two separate, but not mutually exclusive, mechanisms of tree mortality including hydraulic failure (irreversible desiccation and collapse of water transport structures) and carbon starvation (McDowell et al. 2008). Although there has been much recent work on the physiological mechanisms associated with tree mortality, a greater understanding of these mechanisms is needed to assess vulnerability among species and enhance our ability to predict mortality (Hartmann et al. 2015). Furthermore, a better understanding of the ecological consequences of mortality in terms of community-level change (i.e., structure and composition) and ecosystem function is needed (Anderegg et al. 2012).

Mortality rates in old-growth forests in the Plan area have increased above most published rates (>1 percent/year) since the mid 1970s (van Mantgem et al. 2009). A regional study on mortality rates on Forest Service lands in Oregon and Washington corroborated the occurrence of elevated mortality rates in old-growth forests across all vegetation zones from the mid 1990s to mid 2000s during regionwide drought (Reilly and Spies 2016). However, Acker et al. (2015) found that mortality rates in old-growth forests on National Park Service lands (Olympic National Park, North Cascades National Park) in western Washington were lower than those reported by van Mantgem et al. (2009) and Reilly and Spies (2016). Lower mortality rates could be due to geographic variation not represented in van Mantgem et al. (2009) and Reilly and Spies (2016), but may also be indicative of decreasing stress-related mortality following a period of elevated mortality. Consistent with this idea, Cohen et al. (2016) found that remotely sensed forest decline peaked in the mid 2000s during the warmest decade in the past 100 years (Abatzoglou et al. 2014b), then decreased.

Increasing tree mortality rates have been documented in young stands of other regions, and some researchers suggest that they may be more vulnerable to changes in climate than old-growth stands (Luo and Chen 2013). However, Reilly and Spies (2016) found that mortality rates in early- and mid-seral stages from the mid 1990s to mid 2000s were lower than rates in young forests in the

western hemlock and silver fir zones of the western Cascades (Larson et al. 2015, Lutz and Halpern 2006). With the exception of old-growth forests, in which increased mortality led to cumulative losses in basal area and density (van Mantgem et al. 2009), there is generally poor understanding of the effects of recent mortality on stand structure and composition, as well as how these effects differ around the region.

The potential response of tree growth to climate change differs substantially among species depending on the factors that limit growth such as water and length of growing season (Littell et al. 2010, Peterson and Peterson 2001). Growth in Douglas-fir is predicted to decrease under climate change where it currently is water limited (Restaino et al. 2016), but growth may increase where Douglas-fir is limited by growing-season length or lower than optimal temperatures (Albright and Peterson 2013; Creutzburg et al. 2017; Littell et al. 2008, 2010). In species of high-elevation forests where growth is limited by temperature and growing-season length (e.g., subalpine fir, mountain hemlock), growth increased during the 20th century because of warmer winter temperatures and longer growing seasons (McKenzie et al. 2001, Nakawatase and Peterson 2006, Peterson et al. 2002). Warmer winters and earlier snowmelt may also increase potential for drought and water stress in higher elevation forests, especially toward the southern portion of their distribution in southern Oregon and northern California. However, these effects are not yet well documented or understood, and increased growth is expected to continue in the future (Albright and Peterson 2013). The effects of projected climate change on ponderosa pine is uncertain as wetter fall seasons may increase growth while drier summers decrease growth (Kusnierczyk and Ettl 2002). These effects may differ across the landscape as ponderosa pine and western juniper may be more sensitive to drought at lower elevations (Knutson and Pyke 2008). The response of these species also depends on the potential for CO₂ to enhance growth by increasing water-use efficiency (Soule and Knapp 2006). However, some evidence suggests that any benefits of CO₂ fertilization will be outweighed in the future as the climate warms and water becomes a more limiting factor (Gedalof and Berg 2010,

Restaino et al. 2016). Increased levels of CO₂ also have the potential to accelerate maturation and increase seed production (LaDeau and Clark 2001, 2006), but little information is available on the effects of climate change on reproduction in species of the region.

The ability of a species to respond to changes in climate (e.g., earlier warming and drying) with shifts in phenology will be an important factor in determining responses to projected climate change. Altered seasonality may affect growth and reproduction in some plant species. A major concern in the NWFP area associated with warmer winters and earlier springs is the requirement for many species (e.g., Douglas-fir, western hemlock, *Pinus* spp., *Abies* spp.) to experience chilling for the emergence of new leaves, or budburst (Harrington and Gould 2015). Douglas-fir may experience earlier budburst in some portions of its range because of warming, but reduced chilling may cause later budburst in the southern portion of its range (Harrington and Gould 2015). Earlier growth in northern and higher elevation portions of Douglas-fir's range may lead to earlier growth initiation, but reduced chilling in the southern and lower elevation portions of its range are likely to lead to delayed growth initiation (Ford et al. 2016).

Climate change may also affect interactions among and within species in complex ways, but the effects are currently poorly understood. However, several recent studies from higher elevation moist forests in the silver fir vegetation zone of Washington provide some insights. For example, the negative effect of competition on growth is likely to be greater for saplings than for adults, and climate change may have less effect on closed-canopy forests at lower elevations than at higher elevations (Ettinger and HilleRisLambers 2013). Individual growth is likely to increase most in lower density stands as trees may show little response to climate at higher density (Ford et al. 2017). Little is known about the effects of climate change on positive species interactions (e.g., facilitation), though they are known to be important in stressful subalpine environments elsewhere in the Western United States (Callaway et al. 2002), and are thought to play a role in early stand development in dry and cold vegetation zones (e.g., ponderosa pine, subalpine, mountain hemlock) in the NWFP area (Reilly and Spies 2015).

Indirect effects of climate change: disturbance—

The indirect effects of climate change will likely be expressed through increases in the frequency, severity, and extent of disturbance, and are predicted to be the primary mechanisms of ecological change in the future (Dale et al. 2001, Littell et al. 2010). Disturbances include discrete events that alter the structure and function of ecosystems (Pickett and White 1985), but may also include prolonged droughts or multi-year epidemics of pathogens and insects. Disturbance agents are commonly characterized as biotic (e.g., pathogens, insects) or abiotic (e.g., fire, wind, volcanoes), and differ considerably in terms of their prevalence and severity (i.e., tree mortality) across the region and among vegetation zones (Reilly and Spies 2016) (chapter 3). There is great concern that interactions among climate change, forests, and disturbance regimes may result in disturbance effects outside of the natural range of variation (Dale et al. 2000).

Of particular concern are multiple, successive, or compound disturbances (e.g., Paine et al. 1998). Interactions among multiple disturbances may result in multiplicative effects on the structure and function of ecosystems that differ from the cumulative effects of both individual disturbances. The effects of compound disturbances are difficult to predict, but may amplify disturbance severity, cause changes between ecological states (e.g., forest to nonforest transitions), and decrease forest resilience (Buma 2015). However, despite growing recognition and interest in interactions among disturbances, the effects of compound disturbances remain poorly characterized and difficult to predict (Buma 2015, Seidl et al. 2017).

Biotic disturbances—

Biotic disturbances (e.g., insects and pathogens) elevate stand-scale mortality above what are considered normal “background mortality rates” associated with competition and stand development, but may also erupt into epidemic outbreaks that result in high levels of tree mortality (e.g., Raffa et al. 2008). Insects and pathogens do not always result in immediate tree mortality. However, the resulting decline in tree growth and vigor (Hansen and Goheen 2000, Marias et al. 2014) may initiate a long process of mortality (Manion 1981), making trees less resistant to wind disturbance and predisposing them to stem breakage (Larson and

Franklin 2010). Although mortality rates associated with insects are generally much lower than those associated with fire in this region (Reilly and Spies 2016), insects resulted in greater loss of live carbon (Berner et al. 2017) and greater canopy mortality (Hicke et al. 2016) than fire in recent years at the regional scale.

Native insects and pathogen activity is expected to increase as trees experience more stress associated with growing-season drought; however, the implications and magnitude of their effects are likely to be variable and differ geographically as well as among species (Chmura et al. 2011, Kolb et al. 2016a, Sturrock et al. 2011). In addition to affecting host species, climate change will also affect population dynamics and geographic distributions of pathogen and insect species. Pathogen activity is likely to increase in areas where they typically infect drought-stressed host species, while the effects of climate change on pathogens that proliferate under moist conditions may be more variable and difficult to predict (Sturrock et al. 2011). Warmer winters and hotter droughts are expected to enable insects to move into previously unsuitable habitat (Bentz et al. 2010, 2016), and some regions in the Western United States experienced what are considered unprecedented outbreaks of insects in the past few decades (e.g., Raffa et al. 2008). Drought and insects may also interact to further stress trees and predispose them to mortality, but these dynamics are complex and are just beginning to be understood (Anderegg et al. 2015).

Native pathogens play a prominent but variable role in the disturbance regimes of both moist and dry vegetation zones of the region (Goheen and Willhite 2006, Hansen and Goheen 2000) (see Shaw et al. 2009 and chapter 3 for more information on insects and pathogens). Most native pathogens affect small, localized areas at low levels of tree mortality, but are pervasive and generally widespread across the region (Reilly and Spies 2016). Pathogens often initiate forest canopy gaps and can accelerate successional dynamics in old-growth Douglas-fir-dominated forests of the western hemlock vegetation zone (Holah et al. 1997). Laminated root rot (*Phellinus sulphurascens*) (formerly *weirii*) affects Douglas-fir, true firs (*Abies* spp.), and mountain hemlock. Armillaria (*Armillaria ostoyae*) affects Douglas-fir, hemlocks (*Tsuga* spp.), pines (*Pinus* spp.), and

Engelmann spruce. Annosus root disease (*Heterobasidion annosum*) affects firs, pines, hemlocks, and Engelmann spruce. Black stain root disease (*Leptographium wageneri*) affects Douglas-fir and ponderosa pine. Several other types of pathogens are also present, including rusts (*Cronartium* spp.) and mistletoes (*Arceuthobium* spp., *Phoradenron* spp.).

In the Coast Range, Swiss needle cast (*Phaeocryptopus gaeumannii*) is a disease specific to Douglas-fir that has increased since the early 1990s (Hansen et al. 2000b). Ritóková et al. (2016) found that the area affected by Swiss needle cast more than tripled between 1996 and 2015, with growth reductions of 23 percent in the Oregon Coast Range. Swiss needle cast is predicted to increase in the Oregon Coast Range in response to warmer and wetter conditions in the future (Stone et al. 2008), although an increase in drought conditions may inhibit spread of the disease (Rosso and Hansen 2003). High-density Douglas-fir plantations near the coast, where Sitka spruce and western hemlock were historically dominant, are thought to be particularly vulnerable to Swiss needle cast (Black et al. 2010, Hansen et al. 2000, Manter et al. 2003, Rosso and Hansen 2003). An extensive list of research studies of Swiss needle cast is available at <http://sncc.forestry.oregonstate.edu/publications>.

Several species of insects, including bark beetles and defoliators, are also native to the NWFP area. Insects are more prevalent in drier vegetation zones and affected large areas east of the Cascade Range in recent decades (Hicke et al. 2016, Meigs et al. 2015). In Oregon and Washington, recent mountain pine beetle outbreaks were positively associated with warmer winter temperatures and negatively associated with drought stress and precipitation in the current and previous year of outbreak (Preisler et al. 2012). Mountain pine beetle has the potential to cause extensive mortality in lodgepole pine (*Pinus contorta*) and also affect other species of pines, including ponderosa pine, sugar pine (*Pinus lambertiana*), western white pine (*Pinus monticola*), and whitebark pine. Defoliating insects are also common, and though they often do not result in mortality, they may reduce growth and make trees more susceptible to other insect infestations. Several species of pine are susceptible to outbreaks of pandora moth (*Coloradia pandora*), and ponderosa pine is also susceptible to pine butterfly (*Neophasia menapia*). Spruce

budworm (*Choristoneura occidentalis*) is a major concern east of the Cascade Range and affects Douglas-fir and true firs. Williams and Liebhold (1995) projected decreases in the area defoliated by spruce budworm with increased temperature alone, but the area increased with increases in temperature and precipitation. Douglas-fir is also susceptible to Douglas-fir beetle (*Dendroctonus pseudotsugae*), which operates on small patches of trees, especially after blowdown from wind events (Powers et al. 1999).

Several nonnative pathogens and insects are of particular concern in the NWFP area. White pine blister rust (*Cronartium ribicola*) is a major threat to whitebark pine (Goheen et al. 2002, Ward et al. 2006) as well as both western white pine and sugar pine (Goheen and Goheen 2014). Decline of Pacific madrone (*Arbutus menziesii*) related to multiple fungal diseases has been reported over the past 30 years, with larger older trees experiencing the most mortality (Elliott et al. 2002). Balsam woolly adelgid (*Adelges piceae*) has affected subalpine fir and especially grand fir at lower elevations west of the Cascades (Mitchell and Buffam 2001). In southwest Oregon and northwest California, sudden oak death (caused by *Phytophthora ramorum*) has the potential to spread through air, water, and infected plant material (Peterson et al. 2014b, Rizzo and Garbelloto 2003) and may affect tanoak, various species of oak (e.g., California black oak [*Quercus kelloggii*]), other hardwood species (e.g., Pacific madrone and bigleaf maple [*Acer macrophyllum*]), and several species of shrubs (e.g., *Rhododendron* spp.) (see chapter 3). Warmer, wetter winters intensify risk of infection (Haas et al. 2015), and the area affected by sudden oak death is predicted to increase tenfold by the 2030s under projected warmer and wetter conditions (Meentemeyer et al. 2011). Sudden oak death is also associated with increased fire severity on soils in northwest California (Metz et al. 2011). Port Orford cedar (*Chamaecyparis lawsoniana*) is susceptible to a lethal, nonnative root pathogen (*Phytophthora lateralis*) that can be spread over long distances via organic matter carried on boots, vehicles, and animal hooves, and by water (Hansen et al. 2000a, Jules et al. 2002). Recent work suggests that despite rapid initial spread and colonization of *Phytophthora lateralis*, the rate of spread has slowed greatly since 2000 (Jules et al. 2014).

Abiotic Disturbances

Abiotic agents of disturbance in the NWFP area include windstorms, fire, volcanic eruptions, landslides, and avalanches. These disturbances result in much higher levels of tree mortality than biotic disturbances, and are the primary natural agents of stand-replacing disturbance (Reilly and Spies 2016). Abiotic disturbances can create forest gaps and patches of mortality that range in size depending on the disturbance agent (Spies and Franklin 1989). Smaller gaps created by abiotic disturbances may increase stand and landscape heterogeneity, while large, infrequent disturbances may have effects on landscape composition and structure that may persist for centuries (Foster et al. 1998) and are qualitatively different from smaller disturbances (Romme et al. 1998). More details on abiotic agents of disturbance can be found in chapter 3.

Windstorms arising from extratropical cyclones off the Pacific Ocean have the potential to produce hurricane-force winds and extensive damage to forested ecosystems, and large storms affected parts of the NWFP area several times in recorded history (Mass and Dotson 2010). These events are generally characterized by southwesterly winds and occur during the winter when soils are saturated. Coastal areas, particularly the Coast Range in Oregon and Washington, as well as the Olympic Peninsula, were subject to multiple synoptic winds events during the 20th century. Some of these storms also affected inland areas and caused substantial tree mortality in portions of the western Cascades, particularly near the Columbia River Gorge (Sinton and Jones 2002). The most intense of these events, the Columbus Day Storm of 1962 (Lynott and Cramer 1966), killed approximately 11 million board feet of timber in Oregon and Washington (Teensma et al. 1991). High-wind events are positively associated with neutral to warm PDO conditions, and their influence has shifted northward over the past 120 years (Knapp and Hadley 2012), but we are currently unaware of any published literature including future projections of the frequency or intensity of windstorms in the region.

Fire played an important role in the historical dynamics of the region (Agee 1993), but a long period of fire exclusion reduced fire activity during the mid-20th century (Littell et

al. 2009). However, increases in the frequency and extent of fire across the Western United States since the mid-1980s have been attributed to longer fire seasons associated with earlier snowmelt and warmer spring and summer temperatures (Jolly et al. 2015, Westerling et al. 2006) as well as drought (Gedalof et al. 2005, Littell et al. 2009). A recent study also linked increasing fire activity to human-driven climate change, which is contributing to a more conducive fire environment by increasing fuel aridity (Abatzoglou and Williams 2016). Annual area burned has increased since the mid 1980s (Miller et al. 2012, Reilly et al. 2017). However, recent fire activity differs substantially depending on spatial scale and geographic location across the region (Davis et al. 2015, Reilly et al. 2017), and there is growing consensus that the region experienced less fire than would be expected under historical conditions (Marlon et al. 2012, Miller et al. 2012, Parks et al. 2015, Reilly et al. 2017).

The effects of recent fires have been extremely variable across the region, with most recent fire activity occurring in the Klamath Mountains, eastern Cascades, and western Cascades of Oregon (fig. 2-7). The annual area burned increased in most vegetation zones since the mid-1980s, but dry vegetation zones, including ponderosa pine, Douglas-fir, and grand fir/white fir, experienced less fire than they would have during presettlement times because of fire suppression (Miller et al. 2012, Reilly et al. 2017) (see chapter 3 for more discussion). Mean and maximum fire size from 1910 to 2008 increased in northwest California (Miller et al. 2012). Cold and moist vegetation zones (silver fir, mountain hemlock, and subalpine zones, but with the exception of western hemlock) experienced the greatest proportions of high-severity in recent fires, and most of the area burned in the previously mentioned dry vegetation zones has been at low and moderate severity (Miller et al. 2012, Reilly and Spies 2016, Reilly et al. 2017, Whittier and Gray 2016). Fire severity has been related to climate and drought at broad spatial scales since the mid 1980s (Abatzoglou et al. 2017, Keyser and Westerling 2017, Reilly et al. 2017). Although the area burned has increased in all major vegetation zones during this time, there is little evidence that the proportion burning at high severity has increased across the region (Law and Waring 2015, Miller et al. 2012, Reilly et al.

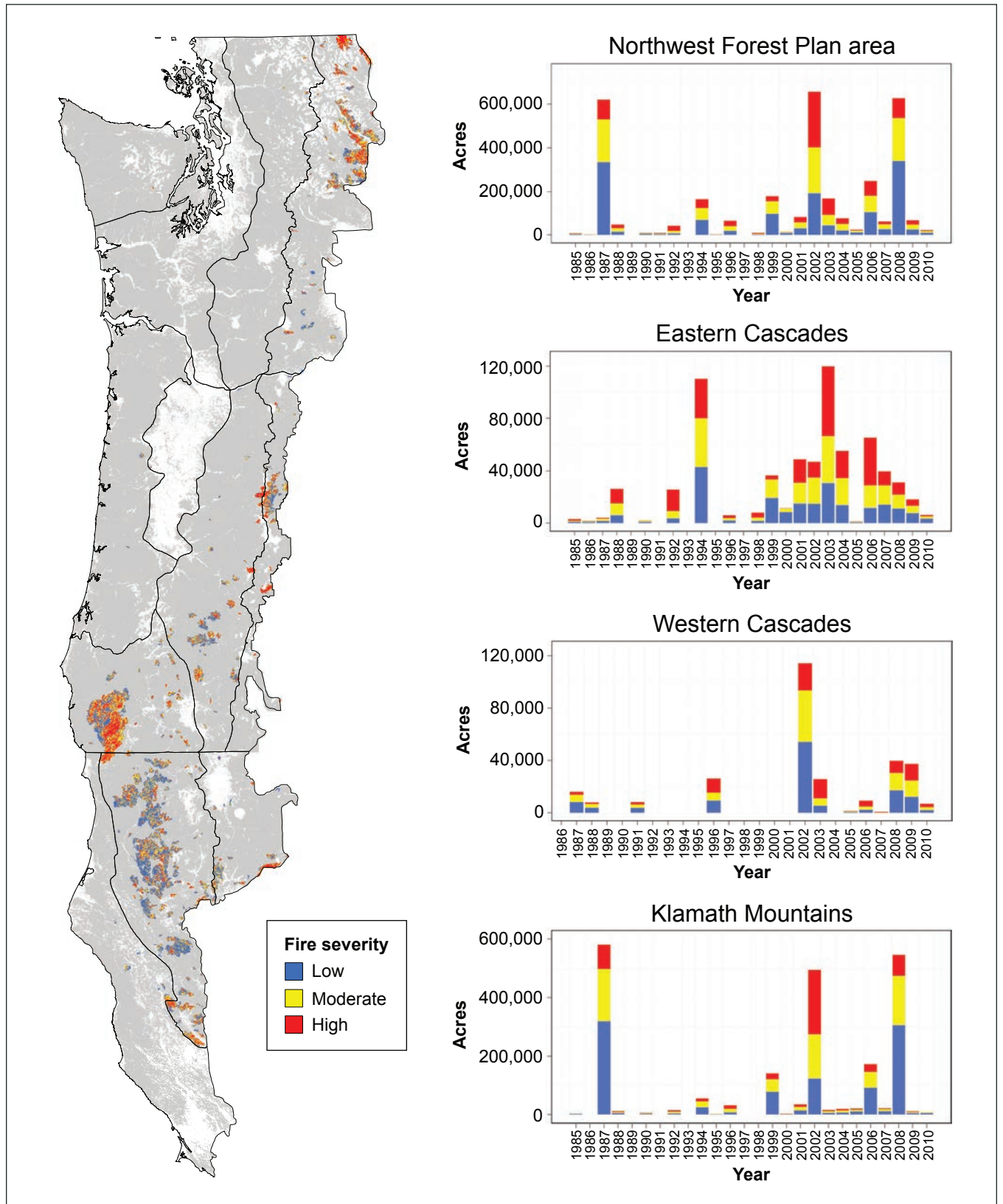


Figure 2-7—Geographic patterns of burn severity from 1985 to 2010 in the Northwest Forest Plan area. Burn severity is derived from the relativized version of the difference in the normalized burn ratio and is based on the percentage of basal area mortality as follows: low (<25 percent), moderate (25 to 75 percent), and high (>75 percent) (Reilly et al. 2017). Map boundaries correspond with the physiographic provinces in figure 2-1.

2017). Although they found no increase in the proportion of high-severity fire, Reilly et al. (2017) found that increases in high-severity patch size during this time were associated with more area burned during drought years in all major vegetation zones.

Despite concern that insect outbreaks may exacerbate fire effects by altering fuel structure (Hicke et al. 2012), there is a growing body of literature within the region and across the Western United States indicating that the two disturbances are not positively linked (Hart et al. 2015, Meigs et al. 2015), and that prefire insect activity does not make fires more severe (Agne et al. 2016, Meigs et al. 2016, Reilly and Spies 2016). These findings are also consistent with several other studies in other regions of the Western United States (Black et al. 2013, Bond et al. 2009, Donato et al. 2013, Harvey et al. 2013, Simard et al. 2011).

Hessl (2011) outlined a framework proposing three major pathways through which future fire activity may respond to climate change. Most studies to date have assumed that the major pathway to change will be based on alteration of fuel conditions as the relationships among weather, fuel moisture, and fire activity are well established. Fewer studies have focused on changes in the second pathway, alteration of fuel amount, though this may be of particular concern given its relation with severity. The least is known about the third pathway, changes in sources of ignition. This pathway will be subject to changes in lightning frequency as well as changes in human ignitions and fire-suppression efforts.

A number of studies using different techniques project increases in a variety of metrics of fire activity (i.e., area burned, fire size, fire severity, fire interval) during the 21st century, although projections differ considerably across the NWFP area (table 2-2). Most studies report coarse-scale projections (i.e., individual states), and few include details at geographic variability within study areas (i.e., east vs. west). Stavros et al. (2014) found that the probability of very large fires will increase based on climate projections for Oregon and Washington, but increases will be minor in northern California. McKenzie et al. (2004) used statistical models and found that an increase in temperature of 3.6 °F (2 °C) will increase fire extent by 1.4 to 5 times for many Western

states, including Oregon, Washington, and California. Using a similar statistical approach, Littell et al. (2010) found that area burned is likely to increase by 2 to 3 times across Washington by the end of the 2040s. They also found that area burned in the western Cascades of Washington is expected to increase by more than eight times, but on average will still affect only a small extent (9,100 ac) of the ecoregion by the 2080s. Liu et al. (2013) projected increases in fire potential associated with warming and drought from 2014 to 2070. Turner et al. (2015) projected an increase in area burned by 3 to 9 times in a portion of the central western Cascades of Oregon. Krawchuk et al. (2009) also predicted increases in fire probability in the western Cascades. Barr et al. (2010) projected an increase in annual fire extent of 11 to 22 percent in the Klamath River basin by 2100. Davis et al. (2017) projected increases in fire suitability across multiple provinces in Oregon and Washington during the 21st century (under RCP 4.5 and 8.5, respectively), including the Klamath Mountains (18 to 48–58 percent), the western Cascades (1 to 13–18 percent), and the eastern Cascades (11 to 40–45 percent). Although projections differ geographically, all studies predict increased fire activity during the 21st century.

There are few statistical predictions for moist maritime forests (i.e., Sitka spruce, redwood, western hemlock) because there has been very little area burned near the coast in the past several decades (Littell et al. 2010). Davis et al. (2017) found no increase in fire suitability in the Puget Trough and only minor increases (<1 to 2 percent) in the Coast Range. Creutzburg et al. (2017) projected very little increase in area burned by 2100 compared to the period from 1959 to 2009 in the Oregon Coast Range. Fried et al. (2004) suggested a decrease of 8 percent in area burned by fires along the north coast of California over the 21st century under continued fire-suppression efforts. Liu et al. (2013), however, predicted an increase in fire potential (measured as Keetch-Byram Drought Index) from 2.5 to 5 times owing to changes in fire weather in coastal forests by 2070. Westerling et al. (2011) projected 300 percent increases in area burned in northwest California. Krawchuk et al. (2009) projected little change in fire potential in coastal forests, but increased potential across the rest of the region. Rogers et

Table 2-2—Projections for future fire activity in the Northwest Forest Plan area from published studies

Study	Method	Geographic extent	Scenario	Time period	Projected change from current	Suppression effects	Variable
					<i>Percent</i>		
Stavros et al. 2014	Statistical	Oregon, Washington, northern California	RCP 4.5, RCP 8.5	2031–2060	+	No	Very large fire occurrence ^a
McKenzie et al. 2004	Statistical	Oregon, Washington, northern California	A2, B2	2070–2100	+	No	Area burned
Littell et al. 2010	Statistical	Washington	A1B	2020–2080	+200 to 300	No	Area burned
Turner et al. 2015	Process	Willamette Valley, Oregon	RCP 4.5, RCP 8.5	2100	+300 to 900	No	Area burned
Krawchuk et al. 2009	Statistical	Global	A2, B1	2070–2090	+	No	Fire probability ^b
Fried et al. 2004	Statistical	Northern California	2× CO ₂	N/A ^c	-8	Yes	Area burned
Spracklen et al. 2009	Statistical	Oregon, Washington, northern California	A1B	2050	+78	No	Area burned
Barr et al. 2010	Process	Klamath Basin, Oregon, and northern California	A2	2075–2085	+11 to 22	No	Area burned
Liu et al. 2013	Statistical	Continental United States	A2	2041–2070	No	No	Fire potential ^d
Westerling et al. 2011	Statistical	Northern California	A2	2085	+100	No	Area burned
Rogers et al. 2011	Process	Oregon, Washington	A2	2070–2099	+76 to 310/29–41	Yes	Area burned/burn severity ^e
Sheehan et al. 2015	Process	Oregon, Washington	RCP 4.5, RCP 8.5	2071–2099	-82 to 14	Yes	Mean fire interval
Creutzburg et al. 2017	Statistical	Oregon	RCP 8.5	2100	Negligible	Yes	Area burned
Parks et al. 2016	Statistical	Western United States	RCP 8.5	2040–2069	No change to decrease	No	Fire severity ^f
Davis et al. 2017	Statistical	Oregon, Washington		2071–2100	No change to increase	No	Suitability for large wildfires ^g

Note: Most studies project area burned or other variables associated with increased area burned (fire suitability, large fire occurrence), and there are relatively few projections for fire severity.

^a Very large fires are defined as those >50,000 ac.

^b Fire probability is the probability of fire occurrence.

^c This study does not project to an explicit time period in the future, but rather conditions based on 2× current CO₂ scenario.

^d Fire potential is measured by the Keetch-Byram Drought Index y.

^e Burn severity is based on combustion of biomass.

^f Burn severity is based on a postfire composite burn index (CBI) based on changes in multiple strata, including soil and rock, litter and surface fuels, low herbs and shrubs, tall shrubs, and trees.

^g Large wildfires are defined in this study as >40 ha.

N/A = not available.

al. (2011) used a mechanistic vegetation model (MC1) that integrates fire and suppression efforts, and found increases in area burned in Oregon and Washington from 76 to 310 percent by 2070 to 2099. Although this increase may seem high, it is important to note that the recent extent of fire in moist forest is very low, and a tripling of fire may still be a relatively small amount in absolute terms.

Although several studies have projected future increase in fire activity, far less work has been done on future fire severity. This component of fire regimes is less well studied and understood (Hessl 2011, Parks et al. 2016), potentially because of the complexities of incorporating feedbacks from fire and climate on fuel structure and arrangement at stand and landscape scales. Previous fires have the potential to inhibit the spread of subsequent fires occurring within a limited time window (Parks et al. 2014), and increased area burned in the future may provide a feedback related to decreased fuel availability. Rogers et al. (2011) used a process model (MC1) and suggested increases in burn severity of 29 to 41 percent that related to increases in productivity and biomass during non-summer months. However, a recent study incorporating changes in vegetation type, fuel load, and fire frequency predicted either no change or potential reductions in fire severity across the entire NWFP area for 2040–2069 under the most extreme climate change scenario (RCP 8.5) (Parks et al. 2016). The authors attributed decreases in fire severity to greater water deficits, decreased productivity, and less available fuel.

The wide range of projections of climate change effects on fire within the NWFP area are likely the result of several factors. These factors include differences in emissions scenarios, spatial and temporal scale, model structure (e.g., statistical vs. process), and variability in how models project precipitation. In addition, McKenzie and Littell (2017) showed that differences in climate-fire relationships among physiographic provinces are likely to be substantial, and further analysis is required to put differences in methodological and regional future projections of fire into context. At coarser regional scales, dynamical and statistical approaches to projecting future fire activity may agree, but the mechanisms operating at more local scales require careful interpretation.

Cumulative effects of climate change on tree species distributions and range shifts—

The cumulative effects of changes in mortality, growth, and recruitment will ultimately be manifest in shifts in species distributions and ranges. These effects will also depend on the size and degree of connectivity within populations. Range expansion occurs through migration and colonization at the outer limits, or “leading edge,” of a species’ distribution where climate is becoming more favorable. Range expansion at the leading edge is controlled by fecundity and dispersal (Thuiller et al. 2008). More vagile species that produce greater amounts of seeds and have a greater ability to disperse will have more potential to track climate change than those with poor dispersal ability. At the lower limits or “trailing edge” of a species’ distribution where climate is becoming less favorable, range contraction and progressive isolation will occur through local extirpation. Range contraction is related to the ability of a species to persist in refugia that experience less change than the surrounding landscape. Individuals at the trailing edge may thus play an important role in the maintenance of genetic diversity for some species (Hampe and Petit 2005). Although local extirpation may occur throughout the range of species, small, isolated populations at the trailing edge may be particularly vulnerable as the climate changes rapidly (Davis and Shaw 2001).

It is likely that species that are more adapted to cold environments will be more sensitive to warming at their lower limits of elevation or latitude, while expansion of species adapted to warmer conditions is expected at upper range limits at high elevation or latitude (HilleRisLambers et al. 2015). Range limits may also be altered at the eastern limits of the range of some species as a result of increasing aridity. Warmer temperatures are likely to lead to range expansion at the leading edge for some species at the upper tree line, but not necessarily for species in closed-canopy forests at lower elevations (Ettinger and HilleRisLambers 2013, Ettinger et al. 2011). However, expansion at upper range limits may be limited by dispersal and low abundance of adult trees that produce seed (Kroiss and HilleRisLambers 2015). Warmer temperatures may increase germination and survival of seedlings provided adequate water, as well

as increase sapling growth rates (Ettinger and HilleRis-Lambers 2013, Ettinger et al. 2011, HilleRisLambers et al. 2015), but many tree species are long lived and may exhibit lagged responses to climate change in terms of range shifts (Kroiss and HilleRisLambers 2015).

A common approach to detecting range shifts is comparing current distributions of mature trees and seedlings. Juveniles (and seedlings specifically) with limited root systems and smaller reserves of carbon are more vulnerable to mortality from drought and temperature extremes (Jackson et al. 2009). Monleon and Lintz (2015) provided evidence of range shifts for common tree species in California, Oregon, and Washington where the range of seedlings extended to temperatures 0.22 °F (0.12 °C) colder than that of adult trees, and seedlings were found at higher mean elevations and latitudes than mature trees for most species during the period from 2001 to 2010. Results also suggested that overall distributions of individual species remained relatively stable, but most species were more abundant toward the colder edge of their range and distributions changed the least at the warm end of their range. Some of the more common tree species with seedlings found at significantly colder temperatures included western redcedar, silver fir, western hemlock, grand fir, and mountain hemlock.

Thus far, individual tree species have shown differential responses to recent warming, and it is likely that tree species will respond differently to projected future changes in climate. Lintz et al. (2016) examined recent changes in basal area and density of 22 tree species on unburned Forest Service lands in Oregon and Washington from the mid 1990s to mid 2000s. Several species had stable populations in terms of density and basal area, including noble fir (*Abies procera*), western redcedar, western hemlock, ponderosa pine, and Douglas-fir. These findings are consistent with HilleRisLambers et al. (2015), who suggested that compositional change in the near term will be slow in higher elevation forests of the silver fir vegetation zone. The greatest levels of mortality in Lintz et al. (2016) occurred in western white pine, whitebark pine, Pacific madrone, subalpine fir, lodgepole pine, grand fir, Engelmann spruce, and western yew (*Taxus brevifolia*). Although this study suggested only slight mortality-related declines of Alaska yellow-cedar

(*Callitropsis nootkatensis*), this species has experienced recent mortality across large areas in southeast Alaska associated with a warming climate (Krapek and Buma 2015).

Recent work from the Klamath Mountains and eastern Cascades in northern California suggests that multiple species, including red fir, Jeffrey pine, lodgepole pine, and white fir, experienced recent increases in mortality (Mortenson et al. 2015). Results from this study indicated that mortality rates for all species were generally higher in smaller size classes. Despite increases in the number of recently dead red fir associated with dwarf mistletoe and drought, the population structure of this species was stable.

Vegetation Models and Potential Future Vulnerability

Several climate change vulnerabilities have been identified either explicitly in the literature, or may be inferred based on knowledge of long-term vegetation change in the region, distribution and dynamics of current vegetation, and projected changes across the region. Increases in temperature, as well as altered precipitation and disturbance regimes, are expected to alter vegetation across the region (see “Summary of Vulnerabilities to Climate Change” on next page). Several types of simulation models are commonly used to predict vegetation responses to potential future climate scenarios, each with their own unique set of assumptions, strengths, and weaknesses (see Peterson et al. 2014a for a more indepth review). Models simplify the complexity of ecological processes by making assumptions that are ideally based on empirical measurements. However, because empirical data are often only available for a few species at a few geographic locations, models are most often based on applications of theory on how species interact and respond to environmental gradients. As a result, the best use of models may be for understanding variability in the magnitude of effects as opposed to predicting specific outcomes (Jackson et al. 2009, Littell et al. 2011). Some of the most common models used to project the effects of climate change can be generally characterized as species distribution models (SDM), dynamic global vegetation models (DGVM), and landscape models. These models have their own unique assumptions and relative strengths and weaknesses, which should be carefully considered when interpreting results.

Summary of Vulnerabilities to Climate Change

General vulnerabilities to climate change include increased wildfire and insect activity driven by drought and extreme weather events, ongoing and new invasions of nonnative species, and loss of some high-elevation species. Fragmented populations at range margins (e.g., Alaska yellow-cedar), as well as narrowly distributed species and species with poor dispersal, are vulnerable to declines from losses of climate-suitable habitat, especially in areas that lack topographic conditions that foster the potential for long-term persistence in relatively climate-stable refugia.

The greatest vulnerability to climate change exists in the drier and colder portions of the region in the eastern Cascades, southern portion of the western Cascades of Oregon, coastal and inland areas of the Klamath Mountains, and the California Coast Range. In dry vegetation zones of these regions, increases in area burned during drought conditions may result in larger patches of high-severity fire and drive landscape-scale change. In general, there is good model agreement that subalpine forests are likely to be reduced everywhere except in the northern portion of the eastern Cascades. Several tree species in both wet and dry vegetation zones

are vulnerable to nonnative pathogens whose effects may be exacerbated by climate change. These include whitebark pine, subalpine fir, sugar pine, western white pine, Port Orford cedar, tanoak, and multiple species of oak. Old-growth forests may also be vulnerable to periods of elevated mortality rates associated with insects and pathogens during drought. Along the coast, decreases in summer fog may substantially reduce suitable climate for redwood and other coastal species that depend on it to mitigate summer drought.

Much of the coastal and inland area toward the central and northern part of the region show either less potential increase or decreases in water-balance deficit during the summer months. However, high-elevation areas may see reduced snowpacks with more precipitation falling as rain. Warmer, wetter conditions may also promote native and nonnative pathogen activity, especially Swiss needle cast on Douglas-fir near the coast. Some of these areas may be vulnerable to a continued northward shift of high-wind events, particularly near the coast in Washington. Although they have been rare in the past century, these areas have historically experienced large fires driven by synoptic warm, dry wind events from the east during drought conditions projected for the future.

Species distribution models are statistical models based on empirical observations of the relationship between a species occurrence and the observed range of environmental or bioclimatic conditions. SDMs are commonly used due to their simplicity, but generally do not represent ecological processes (e.g., biotic interactions, dispersal, adaptation) that constrain species distributions (Ibáñez et al. 2006), and are problematic when extrapolating to future climates that have no modern analogs (Bell and Schlaepfer 2016). Despite these limitations, SDMs provide a basic understanding of how suitable bioclimatic conditions constrain the current distribution of a species, as well as how this distribution might change under any number of different climate change scenarios.

DGVMs are a type of process model that predict ecosystem processes along with the distribution of specific biomes or plant function groups. These models (e.g., MC1) incorporate biogeography and ecophysiology of vegetation types (e.g., coniferous forests, grasslands, woodlands) as well as climate and disturbance to project broad-scale vegetation changes. Biogeochemistry models are also process models, but focus more specifically on carbon, water, and nutrient cycles and are often used to investigate the effects of climate change on productivity and carbon storage. Both types of models are capable of incorporating some of the important ecological processes affecting vegetation response to climate change (e.g., disturbance, CO₂, site water balance), but have generally been applied at broad regional scales with coarse spatial resolution.

Landscape models (e.g., LANDIS-II) (Scheller et al. 2007) generally focus explicitly on simulating processes (e.g., dispersal, growth, mortality) and can represent interactions among vegetation, disturbance, climate change, and management scenarios at a variety of different spatial and temporal scales. Landscapes are represented as gridded cells in which individual cohorts of trees compete for resources, grow, and die. Although some ecological processes are represented in landscape models, many processes that will be sensitive to climate change (e.g., CO₂ fertilization, phenology, biotic disturbances) are not incorporated in these or other models for projecting vegetation change.

Model projections—

DGVMs generally project persistence of cool, maritime forests in the western hemlock and Sitka spruce vegetation zones of the Coast Range in western Oregon and Washington (Creutzburg et al. 2017, Rogers et al. 2011, Shafer et al. 2015, Turner et al. 2015). SDMs project persistence of western redcedar, Sitka spruce, and western hemlock across 55 to 82 percent of their current distributions by 2080 (DellaSala et al. 2015). However, most species in lower elevation, moist vegetation zones are predicted to have less suitable climatic conditions than currently by the mid-21st century (Saxon et al. 2005). One DGVM-based study projected losses of conifer forest across much of the Coast Range in Oregon with increases in cool mixed forests under the RCP 4.5 scenario, and increases in warm mixed forests under the RCP 8.5 scenario (Sheehan et al. 2015). Although western redcedar is thought to be moderately vulnerable to climate change, bigleaf maple is considered to be one of the least vulnerable species in the region (Case et al. 2016). Consistent with a potential decrease in summer fog (Johnstone and Dawson 2010), DellaSala et al. (2015) projected a decrease in suitable climate for redwood of almost 25 percent by 2080.

SDMs project some of the greatest changes for the southern and southwestern part of the NWFP area, with less change in the north and in the western Cascades (Crookston et al. 2010; DellaSala et al. 2015; Hargrove and

Hoffman 2004; McKenney et al. 2007, 2011; Rehfeldt et al. 2006). Using a DGVM, Turner et al. (2015) projected the dominant vegetation type in a portion of the central western Cascades of Oregon to remain forest by 2100, but that the forest would transition from evergreen needleleaf forest to a mixture of broadleaf and needleleaf growth forms. An SDM-based study by Latta et al. (2010) suggests annual growth increases of 2 to 7 percent in moist vegetation zones west of the Cascade Mountains depending on scenario. However, projections from mechanistic models differ, with some projecting moderate to extreme decreases owing to increases in fire activity (Rogers et al. 2011), and others projecting slight to small decreases in growth (Coops and Waring 2011b). Shafer et al. (2015) suggested that growth will decrease in the southwestern part of the region based on projections from a DGVM.

All types of models project that high-elevation forests will experience the greatest change within the region, with moderate to total reductions in suitable climate by the end of the 21st century (Crookston et al. 2010; Halofsky et al. 2013; Hargrove and Hoffman 2004; Mathys et al. 2016; McKenney et al. 2007, 2011; Rehfeldt et al. 2006; Shafer et al. 2015). Case et al. (2016) suggested that western white pine and whitebark pine have relatively high vulnerability to climate change, while noble fir and silver are moderately vulnerable. Mechanistic models project that suitable climate for subalpine fir will be available only in the northern Cascade Range (Coops and Waring 2011b, Rogers et al. 2011), although climate suitability may increase for mountain hemlock in Oregon (Coops and Waring 2011a). Two additional studies also using mechanistic models also predicted large decreases in the distribution of lodgepole pine by the 2100s (Coops and Waring 2011a, Mathys et al. 2016). SDMs project reduction of 15 to 39 percent by 2080 for several species occurring in high-elevation wet vegetation zones, including silver fir, grand fir, Alaska yellow-cedar, and mountain hemlock (DellaSala et al. 2015). In general, there is more model agreement for subalpine forests than for other vegetation zones, and most suggest that suitable climate is likely to be reduced everywhere except in the northern portion of the eastern Cascades.

Model projections for vegetation change in dry coniferous forests in the southern and eastern parts of the region show little agreement. Species distribution models suggest decreases in suitable climate for ponderosa pine, while some DGVMs project increases or only slight changes in temperate coniferous forests (Coops et al. 2005, Halofsky et al. 2013, Rogers et al. 2011, Sheehan et al. 2015) and others projected decreases (Coops and Waring 2011a). Halofsky et al. (2014) projected that while the area of dry mixed-conifer forest is expected to increase from 21 to 26 percent by 2100, the area of moist mixed-conifer forest is expected to decrease 36 to 60 percent in the grand fir/white fir vegetation zone of the central eastern Cascades. Shafer et al. (2015) projected expansion of woodland vegetation during the 21st century. Case et al. (2016) suggested that grand fir will only be moderately sensitive to climate change. Given the lack of agreement among model projections for vegetation change in dry coniferous forests, these results should be used cautiously in planning and management (Peterson et al. 2014a).

In northern California, the projected changes in most scenarios include losses of evergreen conifer forests and increases in mixed evergreen forest primarily because of increased fire activity (Lenihan et al. 2008). A mechanistic model projects that Douglas-fir will be stressed across almost all of northern California (Mathys et al. 2016). Increases are projected in the hardwood component, shrublands, and grasslands, particularly throughout the eastern and drier areas, while maritime evergreen needleleaf forests are expected to contract (DellaSala et al. 2015). Barr et al. (2010) projected that the upper Klamath River basin will support primarily grassland in place of sagebrush and juniper by 2100. In the lower Klamath River basin (California), conditions suitable for hardwood forests (oaks, tanoak, madrone, etc.) are projected to expand, while those suitable for conifer-dominated forests are projected to contract. Results from Kueppers et al. (2005) primarily suggest range expansion and persistence of currently existing populations of valley oak (*Quercus lobata*). Expansion and persistence of blue oak (*Q. douglasii*) is projected in the northern part of its range, but projections primarily suggest range contraction toward the southern portion of northern California.

Other Vulnerabilities

Invasions of nonnative plant species have the potential to alter vegetation dynamics, soil properties (Caldwell 2006, Slesak et al. 2016), and disturbance regimes (Brooks et al. 2004) (see also chapter 3). Most nonnative plant species were initially introduced for horticultural uses and erosion control, or as contaminated crop seed (Reichard and White 2001). Gray (2008) used a systematic inventory of forest health monitoring plots and found that more than 50 percent of plots in almost all physiographic provinces in the NWFP area had nonnative species present. Most common nonnative plants are associated with management (e.g., clearcuts, thinning), though there is potential for the spread of some nonnative, shade-tolerant shrubs in undisturbed forests (Gray 2005). There is also evidence from the region that roads facilitate the spread of nonnative plants (Parendes and Jones 2000, Rubenstein and Dechaine 2015). Little information is available on temporal trends in the abundance of nonnative plants, but increasing temperatures may favor exotic species, especially grasses in California (Sandel and Dangremond 2012). Warm, dry sites with increased topographic exposure may be particularly vulnerable to exotic species, especially annual grasses, following high-severity wildfire (Dodson and Root 2015). Gray et al. (2011) provided a field guide and prioritized list of nonnative plants along with range maps that cover the entire Plan area. More information on management of nonnative species is also available in Harrington and Reichard (2007).

Many species that depend on climate-sensitive habitats will also likely be sensitive to climate change (Case et al. 2015). Narrowly distributed species (e.g., rare and threatened, endemics) that specialize in uncommon or sparsely distributed habitats (e.g., serpentine soils, montane meadows) are expected to have difficulty responding to changing climatic conditions. Increases in Alaska yellow-cedar mortality in southeast Alaska associated with warmer climatic conditions and projections of future decreases in habitat suitability (DellaSala et al. 2015) suggest that this species may be particularly vulnerable to loss. Damschen et al. (2010) found decreases in the

richness and cover of endemics on serpentine soils in southwest Oregon from the 1950s to early 2000s that were consistent with a warming climate. Harrison et al. (2010) found changes in forest herb communities in the Klamath Mountains of Oregon that were also consistent with expectations of a drier climate during the second half of the 20th century, including lower cover of species with northern affinities and greater compositional similarity to communities on southerly aspects. Loarie et al. (2008) projected decreases in the richness of endemic plant species by 2100 for those that cannot disperse, but potential increases if plants can disperse to suitable areas. If dryer growing season conditions accompany projected warming trends, cool, mesic topographic refugia are likely to become increasingly important for species persistence (Dobrowski 2011, Olson et al. 2012, van Mantgem and Sarr 2015). Montane wetlands may be especially at risk from reductions in water levels, shorter hydroperiods, and increased probability of drying out (Lee et al. 2015).

Adaptation to and Mitigation of Climate Change

Adaptation and mitigation are essential to strategic planning for the effects of climate change (Millar et al. 2007). Adaptation options include management actions at stand and landscape scales to reduce vulnerabilities to climate change. Mitigation includes efforts to increase carbon sequestration in forest ecosystems and provide new energy-efficient products and technologies for society. Halofsky and Peterson (2016) provided a summary of an extensive list of vulnerabilities and corresponding strategies and tactics that were identified and developed through a series of science-management partnerships across the northwestern United States (<http://adaptationpartners.org/library.php>). Strategies for adaptation and mitigation have been identified for forests in the Pacific Northwest, including drier forests of southwest Oregon (Halofsky and Peterson 2016; Halofsky et al. 2016, 2017). Here, we highlight general management actions that could promote adaptation to climate change. We summarize these options in table 2-3. For a broader discussion of conservation options (including reserves) in a period of climate and other landscape changes and their specific relevance to NWFP goals, see chapter 12.

Table 2-3—Summary of adaptation options for climate change vulnerabilities in the Northwest Forest Plan area

Vulnerability	Strategy	Tactics
Increased drought stress	Increase resilience	Thinning Favor drought-resistant species/genotypes
	Foster genetic and phenotypic diversity	Protect trees adapted to water stress Collect seed for future Maintain connectivity for natural species migration
Increasing area affected by fire, insects, and pathogens	Increase stand resilience	Thinning and prescribed fire Increase stand heterogeneity Favor fire-tolerant species
	Increase landscape resilience	Increase landscape heterogeneity Increase diversity of patch sizes Use topography to guide treatments
Loss of forest cover	Monitoring of change	Use existing data and add more where needed Planting/assisted migration Maintain connectivity for natural species migration
Exotic species	Increase control efforts	Early detection/rapid response/frequent inventory Interagency coordination

Source: Halofsky and Peterson 2016.

Adaptation—

Several adaptation options to reduce climate change vulnerability are available (table 2-3). These range from manipulation of stand and landscape structure to foster resistance and resilience to future disturbance, to protection of intact areas and climate change refugia that provide connectivity, and facilitate species migration to more favorable habitats. In the case of disturbance, managers may choose to take actions prior to and in anticipation of disturbance to reduce vulnerability, or after a disturbance to affect the ongoing process of recovery (Dale et al. 1998).

Manipulation of stand and landscape structure with management tools (i.e., thinning, prescribed fire) is thought to increase resistance and resilience to future vulnerabilities associated with drought and disturbance (e.g., fire, insects) in drier forests that may be subject to moisture stress and fire (Hessburg et al. 2015, Spies et al. 2010). Findings from dry forests in other regions support the use of thinning as an option to increase soil water availability, reduce growing-season moisture stress, and improve vigor in older trees (Bradford and Bell 2017; McDowell et al. 2003, 2006), but the NWFP area is lacking specific studies on this topic. Prescribed fire has also been found to increase resistance to drought in dry forests of the Sierra Nevada of California (van Mantgem et al. 2016). Thinning has effectively been used and reduced fire severity in dry Douglas-fir of Washington's eastern Cascades (Prichard et al. 2010), and other regions in the Western United States (Wimberly et al. 2009). Fuel treatment may be effective at reducing fire behavior and burn severity during moderate burning conditions; however, treatments may not be effective during large, weather-driven fires (Lydersen et al. 2014, Reinhardt et al. 2008).

A general principle for thinning to reduce fire severity at the stand scale includes maintaining older trees of fire-tolerant species, reducing understory density, and increasing height to live crowns (Agee and Skinner 2005). Given that these actions will likely increase surface fuels, thinning followed by prescribed fire may help reduce surface fuels. Landscape-scale treatments that restore structural heterogeneity in places where historical fire regimes have been interrupted are proposed as a way to reduce vulnerability to high-severity fire and extensive pathogen

and insect outbreaks in the future (Hessburg et al. 2015). Topography can provide a physical template to consider when designing and implementing landscape-scale treatments (e.g., thinning on dry ridges). Increasing landscape heterogeneity is thought to impede the spread of contagious disturbances (e.g., fire, insects), but empirical evidence supporting this is currently lacking.

There is relatively little research on the use of thinning in moist forests as a climate change adaptation strategy. These forests were relatively dense historically. Thinning, specifically variable-density thinning, can help the growth and survival of the residual trees, as well as improve the adaptive capacity and ecological diversity of stands (Neill and Puettmann 2013) (see chapter 3). In drier parts of moist vegetation zones, where fire was more frequent, thinning and prescribed fire could be used to mimic low- and moderate-severity fire and promote landscape diversity, which in turn could promote landscape-scale resilience to climate change (chapter 3). The use of thinning in moist forests is generally focused on plantations and younger forests and would have to be balanced against landscape-level goals for maintaining high canopy cover in older forests, which can buffer climatic changes as described above (Frey et al. 2016).

Assisted migration of genotypes and species that are adapted to future climate scenarios may improve resilience of species that are not able to migrate, but this option is controversial and poorly understood (Marris 2009). Coastal Douglas-fir populations in particular are considered genetically “maladapted” to future climates in Oregon and Washington (St. Clair and Howe 2007). Bansal et al. (2015) found that populations of Douglas-fir from cooler climates had greater resistance to drought than those from warmer climates, contrary to expectations. Populations from areas with relatively cool winters and dry summers were more tolerant to drought and cold and may be the best adapted to warmer future climate conditions (Bansal et al. 2016). There is little information available from other species from the NWFP area, though a study from Arizona found that ponderosa pine seedlings that originated from low-elevation, drier sites survived the longest during drought (Kolb et al. 2016b).

An alternative to assisted migration involves increasing connectivity by establishing large blocks of forest managed

for biodiversity and resilience to climate change. Where forests are more fragmented by land use and past management, corridors can facilitate the flow of organisms through the matrix of unsuitable habitat (Krosby et al. 2010, Nuñez et al. 2013). Linking contemporary climates with future climate analogs is one approach to promote connectivity in the future and facilitate movement of species in the future (Littlefield et al. 2017). Vos et al. (2008) suggested the following to mitigate projected climate changes: (1) linking isolated habitats to nearby climate-proof reserves, (2) increasing colonization capacity of reserve networks that are projected to remain suitable in the future, and (3) optimizing reserve networks in which climate remains relatively stable (e.g., refugia). In the only biodiversity-climate resiliency study of the NWFP area, Carroll et al. (2010) found that reserves based on spotted owl conservation criteria overlapped areas of high localized-species richness, but poorly captured core areas of localized species' distributions. They found that resilience to climate

change was improved when refugial areas were incorporated into the reserve design of the NWFP.

Protection of climate change refugia based on physiography, soils, and vegetation are a key part of climate change adaptation strategies (fig. 2-8), but identification of refugia has proven difficult (Keppel et al. 2012, Morelli et al. 2016). Most studies of refugia have been ad hoc or descriptive and primarily conceptual, and multiple lines of evidence using different approaches from across disciplines (e.g., SDMs, downscaled climate models, genetics) may be necessary to further understanding of refugia (Keppel et al. 2012). Refugia will most likely be found in topographically complex landscapes where microclimates vary from differences in aspect, shading and insolation, and cold-air drainages (Dobrowski 2011). McRae et al. (2016) mapped potential landscape resilience based on topoclimate diversity and regional connectivity for the Pacific Northwest and northern California. Many of the areas of highest resilience occurred in mountainous areas of federal

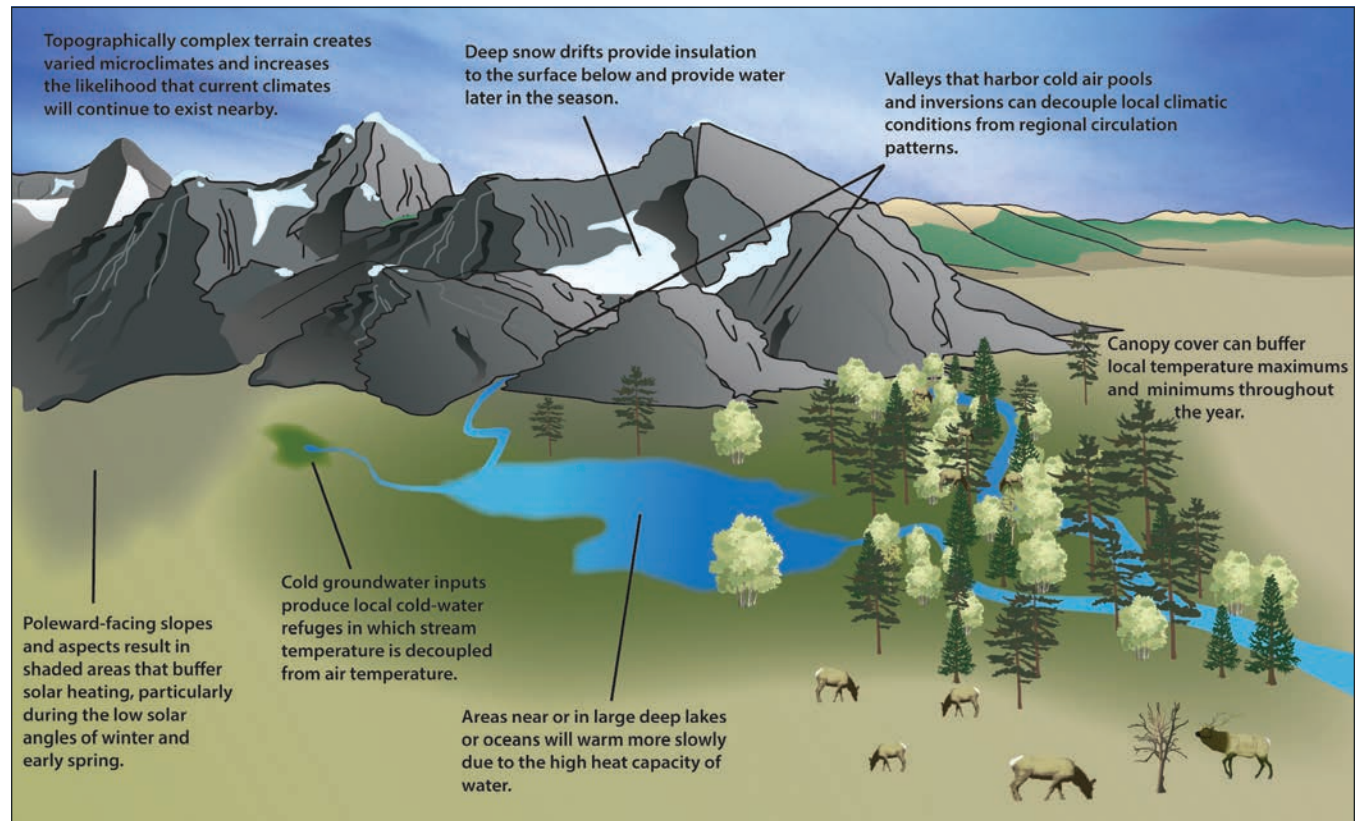


Figure 2-8—Examples of the physiographic and vegetation-based refugia that may experience reduced rates of climate change. Source: Morelli et al. (2016).

lands (e.g., Olympic Peninsula and the Klamath Mountains region). Morelli et al. (2016) presented a synthesis and review of literature pertaining to climate change refugia for climate adaptation. They provided a framework for identifying, mapping, and conserving climate change refugia to meet management objectives. This involves consideration of valued resources and vulnerabilities, identification of climate change refugia, and prioritization of refugial areas.

Increasing connectivity may be insufficient for those species that are unable to migrate as rapidly as the climate changes (Dobrowski et al. 2013). Connectivity considerations would likely need to be species-specific because each species experiences the same landscape in different ways (Betts et al. 2014). Refugia should also be large enough to support populations they are aimed at conserving (Stewart et al. 2010). Planning and monitoring are also essential for adaptation and can help identify microclimatic settings that may provide suitable refugia in the future, coordinate planning across jurisdictions and ownerships, and revise management goals and objectives to be consistent with the uncertainty that accompanies climate change (Spies et al. 2010). For a broader discussion of refugia and connectivity related to the reserve network of the NWFP, see chapters 3 and 12.

Mitigation—

Mitigation includes efforts to increase carbon sequestration in forest ecosystems and provide new energy-efficient products and technologies for society. Of these, we focus on the former, which has been proposed as a means of climate mitigation (Depro et al. 2008, Law and Harmon 2011, Ryan et al. 2010), and then discuss how management practices have the potential to affect carbon sequestration in the NWFP area.

Forests in the NWFP area have great potential to store large amounts of carbon in both live and dead biomass (Smithwick et al. 2002). Total carbon storage levels differ among physiographic provinces (fig. 2-9) as a result of

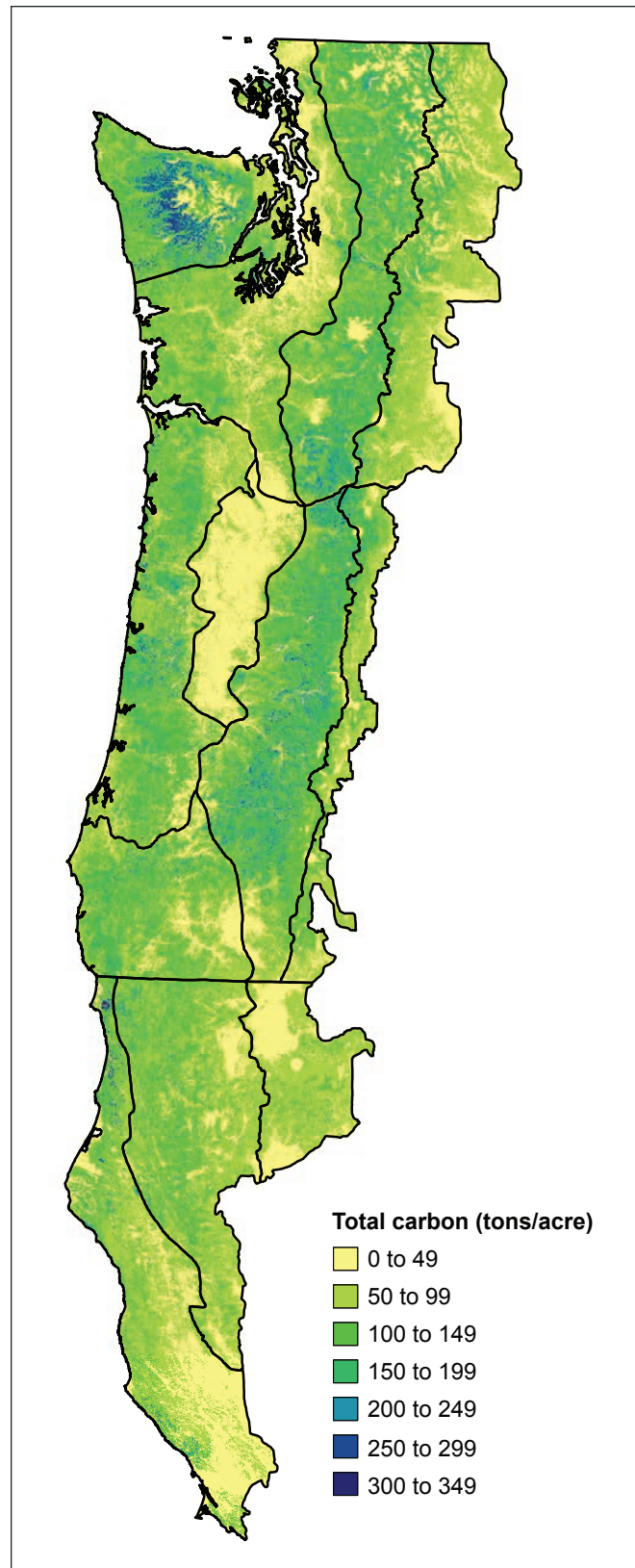


Figure 2-9—Total forest carbon density in the Northwest Forest Plan area (2000–2009). Carbon estimates are from Wilson et al. (2013). Map boundaries correspond with the physiographic provinces in figure 2-1.

productivity and disturbance (Law et al. 2004). Recent findings suggest that forests on Forest Service lands in Oregon and Washington currently store about 63 percent of their potential maximum carbon (Gray et al. 2016). At current rates, harvest and disturbance have little overall impact on carbon sequestration on federal lands in Oregon and Washington as a whole, but this differs at smaller scales among geographic areas (Gray and Whittier 2014). This is particularly true in areas in which dry forests have experienced substantial landscape change in recent fires. In the Oregon Coast Range, projected increases in productivity are associated with projections of increased carbon storage (Creutzburg et al. 2017), but gains could be offset by losses depending on harvest intensity (Creutzburg et al. 2016). Projections suggest future decreases in carbon storage from increases in fire activity in the eastern and western Cascade Range of Washington (Raymond and McKenzie 2012). In forests west of the Cascades where fire is less frequent, decreasing harvesting, increasing rotation age, and maintaining and increasing the extent of late-successional and old-growth forests are strategies to increase carbon storage toward theoretical maximum limits (Creutzburg et al. 2016, 2017; Hudiburg et al. 2009). Maintaining and increasing the area of dense old-growth forests with high biomass also has the potential to mitigate temperature changes in topographically complex mountainous environments (Frey et al. 2016).

Carbon stores in the more fire-prone drier eastern and southwestern part of the region are more unstable and less predictable owing to recent increases and future projections of increased fire activity (Restaino and Peterson 2013). Some studies from other regions in the Western United States (i.e., the Southwest and Sierra Nevada) suggest that thinning and fuel reduction can mitigate carbon loss from fire. Fuel reduction may reduce losses of carbon at stand levels compared with the consequences of high-severity wildfire burning in stands with high fuel loads (Finkral and Evans 2008; Hurteau and North 2009; Hurteau et al. 2008, 2011, 2016; North and Hurteau 2011; North et al. 2009, Stephens et al. 2009). However, because the probability of treated areas burning is generally low (Barnett et al. 2016), and most biomass is not consumed by fire, slight differences

in losses resulting from combustion in fire compared with losses from fuel reduction are unlikely to make fuel reduction a viable mitigation strategy (Ager et al. 2010, Campbell et al. 2012, Kline et al. 2016, Mitchell et al. 2009, Restaino and Peterson 2013, Spies et al. 2017). As the amount of fire on the landscape increases, the difference in carbon sequestration between untreated and treated landscapes declines and the likelihood that thinning will pay off in respect to the overall carbon balance increases (Loudermilk et al. 2014).

Research Needs, Uncertainties, Information Gaps, and Limitations

Despite the accumulating scientific information that supports increased warming, considerable uncertainty surrounding the effects of climate change on precipitation, vegetation response, and disturbance remains a significant challenge to forest management (Halofsky and Peterson 2016, Millar et al. 2007). Many of these research needs are mentioned throughout this chapter, but we identify several specific information gaps here.

1. Future role of climate extremes and weather events as disturbances (e.g., heat waves, floods, windstorms).
2. Clarification of the effects of future changes in CO₂, temperature, and water deficit on growth and mortality, and how these effects differ geographically across the region within and among species and seral stages.
3. Effects of recent tree mortality on composition and structural development across seral stages in all vegetation zones.
4. Role of drought on future patterns of disturbance occurrence and severity (e.g., fire, insects, pathogens) in all vegetation zones.
5. Role of interactions among multiple disturbances (e.g., compound and linked disturbances, including insects and fire).
6. Effects of climate change on demographic processes related to migration (e.g., fecundity, dispersal) and how these differ among species in different vegetation zones.

7. Limited understanding of the role of biotic interactions (e.g., competition with and among species) on vegetation response to climate change.
8. Reducing uncertainty surrounding projections in the amount and geographic distribution of species in dry vegetation zones (e.g., ponderosa pine).
9. Response of high-elevation forests to increased summer drought.
10. Effects of thinning on resilience to drought in all vegetation zones.
11. Effects of increasing landscape heterogeneity from fuel treatments (e.g., thinning and prescribed fire) and recent wildfires on future fire and insect activity.
12. Phenotypic responses of individual species to drought and warmer winter temperatures.
13. The potential role and identification of climate and disturbance refugia in all vegetation zones.
14. Multiscale assessment (i.e., stand to landscape) of fuel treatment effects on carbon mitigation under increasing fire activity.
15. Potential of the current NWFP reserve network and management standards and guidelines to provide climate refugia, connectivity to facilitate migration of different species, and stand and landscape conditions that promote resilience to drought, fire, insects, pathogens, and nonnative species.

Conclusions and Management Considerations

Despite the uncertainty surrounding projections of future climate, disturbance and vegetation change, several key vulnerabilities have been identified and are supported by a large body of scientific evidence (see box on page 56). Most models agree and project that the region will experience warmer, drier summers and potentially warmer and wetter winters. Conditions are projected to exceed the 20th-century range of variability around the 2050s, particularly in the Klamath and southern Cascade Mountains. Potential impacts in lower elevation, moist vegetation zones (i.e., western hemlock) include decreased growth and productivity, especially where species are already water limited

during the growing season. The greatest vulnerability to climate change is in higher elevation forests, specifically in the subalpine vegetation zone. These forests are likely to experience large decreases in area and may potentially be limited to refugia in the Northern Cascade Range (Mote et al. 2014). Although a great deal of uncertainty surrounds future vegetation change in dry forests, most models consistently agree on an increased role of fire in the 21st century, which is likely to include more area burned and larger patches of high-severity fire. However, most models do not project fire severity or include fire/climate/fuel feedbacks that could be used to project severity.

Projections for climate and vegetation change represent a range of outcomes that can be used to estimate the potential magnitude of effects across the region, but they do not predict specific outcomes. Recent scientific findings suggest several important management considerations for mitigation and adaptation in the face of ongoing climate change across the NWFP area. It is important to consider the potential variability in projections among physiographic provinces and even among landscapes and topographic settings **within** a physiographic province when planning management activities.

1. Considering a variety of approaches may be helpful when managing in the face of uncertainty. “Bet hedging” strategies and multiple courses of action may help to minimize risk and enable further learning. One strategy for dealing with this uncertainty in a planning context is to use scenarios and risk analysis (Acosta and Corral 2017, Bizikova and Kremer 2015, Pasalodos-Tato et al. 2013) (see also chapter 12).

Maintaining dense late-successional forests may help mitigate effects of climate change and have the potential to buffer warming at finer scales in moist vegetation zones where fires are infrequent. In addition to storing large amounts of carbon, late-successional forests may also provide refugia for species that depend on cooler, mesic habitats. In dry forest landscapes, maintaining large areas of dense, multilayered older forests would be inconsistent with a strategy for increasing resilience to drought and fire (chapter 3).

2. Landscape-scale treatments to reduce fuels with thinning, prescribed fire, and managed wildfire may promote heterogeneity in dry forests where historical fire regimes were interrupted during the 20th century. These activities can also reduce vulnerability to high-severity fire during moderate weather conditions, as well as to extensive pathogen and insect outbreaks. Topography can provide a physical template to consider when designing and implementing landscape-scale treatments (e.g., thinning on dry ridges and around sheltered refugia).

Maintaining and increasing connectivity may facilitate migration of species experiencing unsuitable climatic conditions. However, connectivity needs are likely to differ among species, and generic connectivity measures may not be adequate for focal species. In situations in which species' climatic envelopes are changing more rapidly than species are migrating, assisted migration can promote genetic and phenotypic diversity and may help maintain forest cover, although the net benefits of this practice are uncertain and controversial in the scientific literature.

3. Monitoring of populations, species distributions, forest conditions, and disturbance are essential to inform management decisions and help prioritize objectives for adaptive management in response to changes. Most species are expected to respond individually to projected changes in climate and disturbance regimes, and future forest communities may not have contemporary analogs. Understanding the responses of an individual species and how they differ across its range can assist in developing strategies to promote species persistence and prioritize management efforts.

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Appendix: Crosswalk of Simpson (2013) Potential Vegetation Zones With Existing Vegetation From the Classification and Assessment With Landsat of Visible Ecological Groupings (CALVEG) System

Values indicate the percentage of the potential vegetation zone that falls into the CALVEG class. Existing vegetation comes from the Regional Dominance Type 1 field in the CALVEG database and indicates the primary, dominant vegetation alliance. The listed existing vegetation alliances comprise 95

percent of each potential vegetation zone in northern California. Current vegetation types with less than 2 percent cover in a potential vegetation zone are not shown. For information on CALVEG, see <http://www.fs.usda.gov/detail/r5/landmanagement/resourcemanagement/?cid=stelprdb5347192>.

Potential vegetation zone	CALVEG regional dominance 1
Western hemlock	Douglas-fir (40.3%), white fir (18.5%), Jeffrey pine (15.5%), tanoak (madrone) (9%), black oak (3.9%), ultramafic mixed conifer (3.7%), California bay (2.9%), red fir (2.4%)
Tanoak	Douglas-fir (40.3%), tanoak (madrone) (11.3%), Oregon white oak (6.2%), California bay (5%)
Shasta red fir	Red fir (33.2%), white fir (10.1%), Jeffrey pine (10.1%), barren (10%), mixed conifer–fir (8.1%), alpine grasses and forbs (5.1%), pinemat manzanita (5%), subalpine conifers (4.9%), upper montane mixed chaparral (2.9%), perennial grasses and forbs (2.1%)
Port Orford cedar	Douglas-fir (46.6%), ultramafic mixed conifer (24.8%), Douglas-fir–white fir (7.9%), tanoak (madrone) (2.9%), Douglas-fir–ponderosa pine (2.9%), mixed conifer–pine (2.2%), Oregon white oak (2%)
Other pine	Lower montane mixed chaparral (16.5%), gray pine (10.1%), chamise (8%), Oregon white oak (7.1%), interior mixed hardwood (6.6%), canyon live oak (5.6%), blue oak (5.6%), annual grasses and forbs (4.8%), Douglas-fir–ponderosa pine (4.4%), scrub oak (3.6%), Douglas-fir (3.5%), mixed conifer–pine (3.3%), Sargent cypress (3.2%), black oak (2.5%), knobcone pine (2.2%), ponderosa pine (2%)
Grand fir/white fir	Mixed pine conifer (27.1%), white fir (19%), Douglas-fir–white fir (14%), Douglas-fir (10.6%), Douglas-fir–ponderosa pine (6.3%), red fir (5.9%), mixed conifer–fir (2.5%), upper montane mixed chaparral (2%)
Douglas-fir	Douglas-fir (29.3%), Douglas-fir–ponderosa pine (13.3%), Oregon white oak (12.7%), mixed conifer–pine (7.8%), lower montane mixed chaparral (5.3%), canyon live oak (4.6%), black oak (4%), interior mixed hardwood (3.8%), ponderosa pine (3.2%), annual grasses and forbs (2%)
Juniper	Annual grasses and forbs (45.3%), mixed conifer–pine (17.2%), barren (8.3%), Douglas-fir–ponderosa pine (7%), upper montane mixed chaparral (4.3%), perennial grasses and forbs (2.9%), manzanita chaparral (2.8%), ponderosa pine–white fir (2.3%), Jeffrey pine (2%)

Map available from <http://www.ecoshare.info/category/gis-data-vegzones>.

Source: Simpson 2013.



Old-growth forest, Oswald West State Park, Oregon.
Photo by David Patte, U.S. Fish and Wildlife Service.