

Chapter 3: Old Growth, Disturbance, Forest Succession, and Management in the Area of the Northwest Forest Plan

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Introduction

In this chapter, we examine the scientific basis of the assumptions, management strategies, and goals of the Northwest Forest Plan (NWFP, or Plan) relative to the ecology of old-growth forests, forest successional dynamics, and disturbance processes. Our emphasis is on “coarse-filter” approaches to conservation (i.e., those that are concerned with entire ecosystems, their species and habitats, and the processes that support them) (Hunter 1990, Noss 1990). The recently published 2012 planning rule has increased emphasis on land management rooted in ecological integrity and ecosystem processes, using coarse-filter approaches to conserve biological diversity (Schultz et al. 2013). Fine-filter approaches (e.g., species centric), which are also included in the 2012 planning rule, are discussed in other chapters. We synthesize new findings, characterize scientific disagreements, identify emerging issues (e.g., early-successional habitat and fire suppression effects) and discuss uncertainties and research needs. We also discuss the relevance of our findings for management. Climate change effects on vegetation and disturbance and possible responses (adaptation

and mitigation) are addressed mainly in chapter 2 of this report. Although, our effort is primarily based on published literature, we bring in other sources where peer-reviewed literature is lacking, and we conduct some limited analyses using existing data. We are guided by the NWFP monitoring questions, those from federal managers and our reading of the past three decades of science.

Old-growth forests can be viewed through many ecological and social lenses (Kimmins 2003, Moore 2007, Spies and Duncan 2009, Spies and Franklin 1996). Socially, old growth has powerful spiritual values symbolizing wild nature left to its own devices (Kimmins 2003, Moore 2007), and many people value old growth for its own sake (“intrinsic” values, *sensu* Moore 2007). Old growth also has many “instrumental” or useful functions, including habitat for native plants or animals (e.g., the northern spotted owl [*Strix occidentalis caurina*]), carbon sequestration (Harmon et al. 1990), and other ecosystem services. No single viewpoint fully captures the nature of the old-growth issue as it relates to federal forest management. We focus here on ecological perspectives (Kimmins 2003, Oliver 2009, Ruggiero et al. 1991, Spies 2004, Spies and Franklin 1996), many of which are overlapping conceptually and in common parlance. Old growth is many things at the same time; for example, old growth is:

- An ecosystem “distinguished by old trees and related structural attributes. Old-growth encompasses the later stages of stand development that typically differ from earlier stages in a variety of characteristics including tree size, accumulation of large dead woody material, number of canopy layers, species composition and ecosystem function” (USDA FS 1989).
- An ecological state resulting from interactions among successional, disturbance, and ecosystem processes (e.g., nutrient and carbon cycles, microclimate).
- A biological condition defined in terms of life histories and demographics of forest plant species.
- A habitat for particular fauna, flora, and fungi.

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We define old-growth forests based on live and dead structure and tree species composition (see below). Old-growth forests in the NWFP area differ with age, forest type, environment, and disturbance regime (Reilly and Spies 2015, Spies and Franklin 1991). The variability and complexity of site conditions, forest succession, and disturbance processes make defining old-growth difficult or impossible under a single definition. Under the U.S. Department of Agriculture (USDA), Forest Service (USDA FS 1989) definition (above), the only features distinguishing old-growth from other forests, across all forest types, are the dominance or codominance of old, large, live and dead trees (multiple canopy layers are not necessarily a defining characteristic). For example, in fire-frequent historical forest types, old-growth forests have large old live and dead trees, but amounts of deadwood are low, canopies are generally open, and areas with multiple canopy layers are uncommon (Dunbar-Irwin and Safford 2016, Safford and Stevens 2016, Youngblood et al. 2004) (fig. 3-1).

In the NWFP, “older forests” were defined as “late-successional/old-growth” based largely on stand developmental and successional patterns of Douglas-fir/western hemlock (*Pseudotsuga menziesii*/*Tsuga heterophylla*) forests (Franklin et al. 2002) (fig. 3-2). This multilayered closed-canopy old growth (e.g., canopy cover >80 percent) was the focal point of old-growth conservation during the development of the NWFP, but as we shall argue, old growth is far more diverse than that and functions quite differently across the range of the northern spotted owl. “Older forests” in the original NWFP includes mature forests, 80 to 200 years of age—a pre-old-growth stage, known somewhat confusingly as “late-successional”² in the Plan), and old-growth forests. Old-growth has been defined in the NWFP and elsewhere as forests containing large and old, live and dead trees, a variety of sizes of other trees, and vertical and horizontal heterogeneity in tree clumps, gaps, and canopy layering (see

O’Hara et al. 1996, Spies 2006, and Davis et al. 2015 for more discussion of old-growth or old-forest definitions). According to Spies and Franklin (1988), old-growth is part of a structural and compositional continuum of successional stages that varies by environment. According to O’Hara et al. (1996), speaking of frequently disturbed environments, old forest is a part of the successional continuum that varies by environment and disturbance processes, which have the ability to advance or retard succession.

To operationalize the successional continuum concept of old-forest development, Davis et al. (2015) created an old-growth structure index (OGSI) to characterize the degree of old-growth structure (“old-growthiness” calibrated by potential vegetation type) that occurs in a stand of any age or history, for use in mapping and monitoring in the Plan area. Two definitions for late successional/old growth were then created: OGSI 80 (structural conditions commonly found in forests that are 80 years and older) and OGSI 200 (structural conditions that are representative of forests containing trees that are more than 200 years of age). These classes roughly correspond to the definitions used by FEMAT, the Forest Ecosystem Management Assessment Team (FEMAT 1993), for mature trees (80 to 200 years old) (e.g., “late-successional” in the NWFP) and old growth (>200 years) but have the advantage of being structure based and calibrated to different potential vegetation types. Also, given that this is a continuous index, other age/development thresholds (e.g., 120 years) could be used for mapping and monitoring.

We note that the structure index and definitions used in the monitoring program are based on current forest conditions from forest inventory plots, which means that in fire-frequent dry zone forests, the structure and composition of old growth is a product of 100 years or more of fire exclusion and highly altered forest development processes. Inventory definitions for dry, old forests based on densities of large-diameter fire-tolerant trees have been developed for the eastern Washington Cascade Range (Franklin et al. 2007a). However, definitions and indices of dry, fire-dependent, old-growth forest structure at stand and landscape scales are still needed for the larger NWFP area (see below for further discussion).

² Most of the time in this document, we use the term “late successional” to refer to vegetation that is in the later stages of forest succession where age, height, and biomass are near maximum and shade-tolerant species are the primary understory or overstory tree species. This broad class would include old growth according to classic definitions in textbooks (Barnes et al. 1998).



Tom Iraci

Figure 3-1—Open, old-growth ponderosa pine stand maintained by low-severity fire in central Oregon.



Tom Iraci

Figure 3-2—Multilayered, old-growth Douglas-fir and western hemlock stand in the western Oregon Cascades.

Old growth has been the focal point for forest conservation and restoration on federal lands in the Pacific Northwest. However, the broad goals of forest biodiversity conservation would not be scientifically viable if they focused on only one stage of a dynamic system—all developmental phases and ecological processes must be considered (Spies 2004), including postdisturbance stages (fig. 3-3), nonforest vegetation, and younger forests that constitute the dynamic vegetation mosaics that are driven by disturbance and succession. These other stages and types contribute to biodiversity, and hence, are as important to any discussion of forest conservation or management for ecological integrity as is the discussion of old growth. Indeed, these other successional conditions become future

old growth, so the successional dynamics of the entire landscape ought to be the broader focus of discussions. Consequently, our discussion includes these other stages of forest succession, in addition to old growth.

Guiding Questions

This chapter characterizes the current scientific understanding of old-growth forest conditions and dynamics and other successional stages in the NWFP area, especially as they apply to conservation and restoration of forest ecosystems and landscapes. We give special attention to composition and structure of trees (live and dead) as dominant components of forests but acknowledge that other characteristics are also important, including age (or time since disturbance) and composition, and structure of shrub, herb, and grass communities. Our focus is on the broad landscape, which inherently is a mosaic of vegetation conditions; questions related to conservation and restoration of animal species in terrestrial habitats and riparian and aquatic ecosystems and their habitats are dealt with in other chapters.

We address the following major questions in this chapter, though not directly given their breadth, complexity, and certain degree of overlap. See the conclusions section for bullet statements that are explicitly linked to these questions.

1. What are the structures, dynamics, and ecological histories of mature and old-growth forests in the NWFP area, and how do these features differ from those of other successional stages (e.g., early and mid successional)?
2. How do these characteristics differ by vegetation type, environment, physiographic province, and disturbance regime?
3. What is the scientific understanding about using historical ecology (e.g., historical disturbance regimes and natural range of variation [NRV]) to inform management, including restoration?
4. What are the principal threats to conserving and restoring the diversity of old-growth types and to other important successional stages (e.g., diverse early seral), and to processes leading to old growth?

Thomas Spies



Figure 3-3—Early-successional vegetation 8 years after a high-severity fire in multilayered old growth in southwestern Oregon.

5. What does the competing science say about needs for management, including restoration, especially in dry forests, where fire was historically frequent?
6. How do the ecological effects of treatments to restore old-growth composition and structure differ by stand condition, forest age, forest type, disturbance regime, physiographic province, and spatial scale?
7. What are the roles of successional diversity and dynamics, including early- and mid-seral vegetation, in forest conservation and restoration in the short and long term?
8. What is the current scientific understanding concerning application of reserves in dynamic landscapes?
9. How do recent trends of forests in the NWFP reserve network relate to both original NWFP goals, those of the 2012 planning rule, and climate change adaptation needs?
10. What is the current understanding of postwildfire management options and their effects?
11. What are the key uncertainties associated with vegetation under the NWFP, and how can they be dealt with?

We address these questions using an organization based on major forest regions, disturbance regimes, and potential and existing forest vegetation types.

Key Findings

Vegetation Patterns and Classification

Drivers of regional variation in vegetation—

Forest ecosystems of the vast NWFP region are ecologically diverse and complex and do not lend themselves to simple generalizations (fig. 3-4). In this synthesis, we account for some of that diversity by classifying ecosystems based on potential vegetation types at the zone or series level (Henderson et al. 1989, Lillybridge et al. 1995, Simpson 2007) in a manner similar to Küchler (1964, 1974). Potential vegetation types and disturbance regimes are somewhat correlated, although disturbance regimes can differ significantly within potential vegetation types

(i.e., biological and physical environments) (Hessburg et al. 2007, Kellogg et al. 2007, Wright and Agee 2004,) and differences in potential vegetation types or forest composition do not necessarily mean differences in fire history (Taylor and Skinner 1998).

The major biophysical driving variables (aka “drivers”) of structure, composition, and dynamics of old-growth forests (and forests in general) are climate, topography, soils, succession processes, and disturbance processes (Franklin and Dyrness 1973; Gavin et al. 2007; Hessburg et al. 2000a, 2015; O’Hara et al. 1996; Oliver and Larson 1990; Spies and Franklin 1996). In conjunction with landform and soil conditions, the geographic and historical variability of the regional climate set the stage for somewhat predictable biotic communities, pathways of forest development, levels of ecosystem productivity, and spatial patterns of disturbance regimes (Agee 1993, Gholz 1982, Hessburg et al. 2000a, Reilly and Spies 2015, Weisberg and Swanson 2003, Whitlock 1992). Climatic variation over time and space exerts a strong control over fire frequency (Agee 1993, Gavin et al. 2007, Walsh et al. 2015), and forest dynamics is a product of the self-organizing interactions of climate, topography, disturbance, and plant communities (Scholl and Taylor 2010). Forest succession is the process of change in tree, shrub, and herb species composition, and structure (size, density, and age structure) over time. Disturbances can advance, arrest, or retard succession either slowly and imperceptibly, rapidly and abruptly, steadily, or in other complex and poorly understood ways (O’Hara et al. 1996, Spies and Franklin 1996). In combination, forest succession and disturbance processes can produce a wide range of forest conditions within the NWFP area.

Classification of vegetation—

Ecological classifications of environment and succession are used to promote understanding and implementation of management objectives. One way that Oregon and Washington ecologists account for environmental differences in succession and in old-growth characteristics (Davis et al. 2015, Reilly and Spies 2015) is to use potential vegetation type (fig. 3-4).

Potential vegetation type is named for the native, late-successional (or “climax”) plant community that would

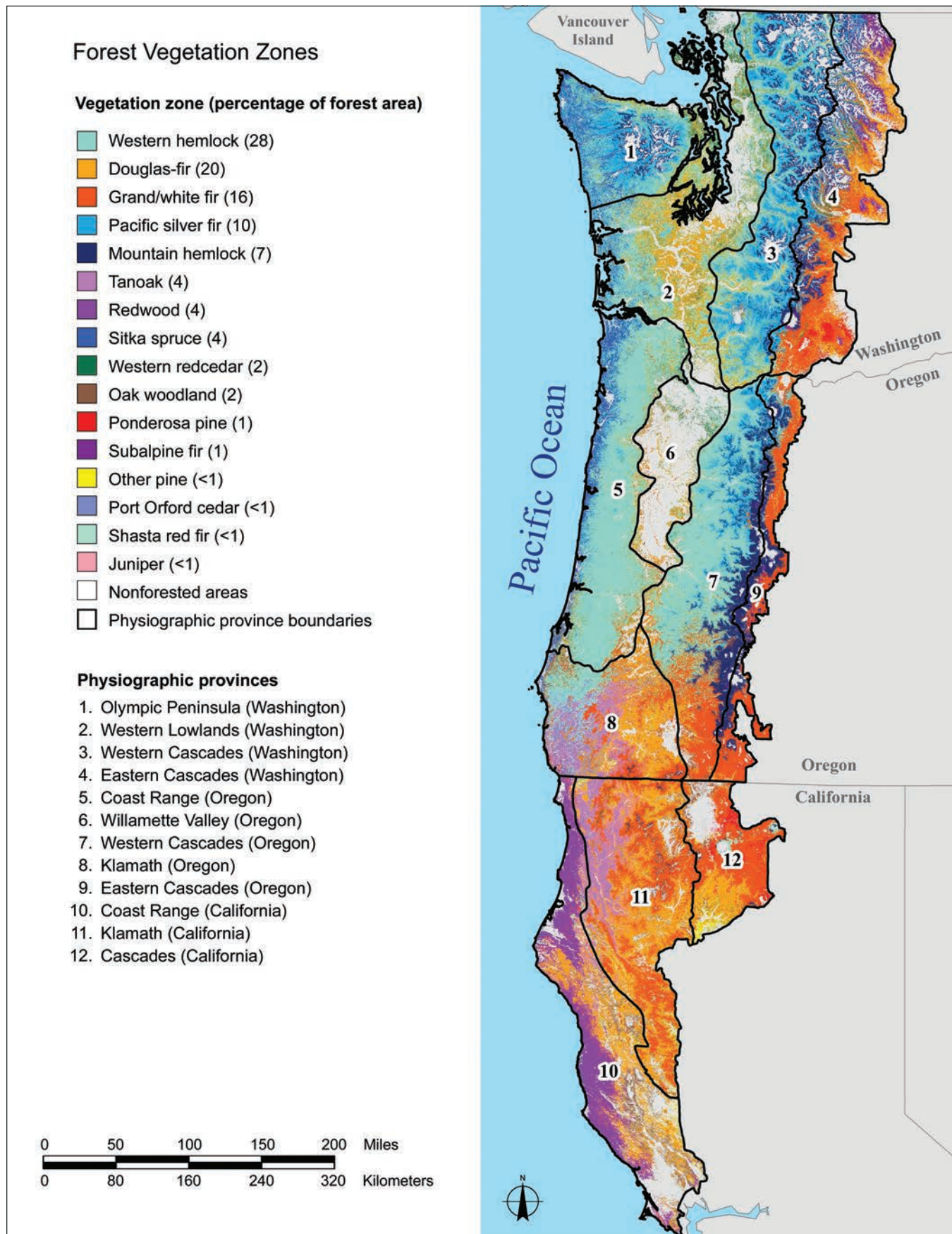


Figure 3-4—Geographic distribution of potential vegetation zones (aka vegetation types) (Simpson 2013) and physiographic provinces across the Northwest Forest Plan area.

occur on a site in the absence of disturbances (i.e., wildfire, bark beetle outbreaks, root disease, weather events), and reflects the biophysical environment (climate, topography, soils, productivity) and composition of overstory and understory species (Pfister and Arno 1980). Stages along the continuum within a potential vegetation type may be binned or categorized into distinct successional stages, which are mileposts for visualizing forest development subjectively given that no clear thresholds in development are known (Franklin et al. 2002, Hunter and White 1997, O'Hara et al. 1996, Oliver and Larson 1990, Reilly and Spies 2015, Spies and Franklin 1988). This classification is often required to enable large-landscape analyses, which cannot efficiently deal with developmental conditions treated as continuous variables.

Not all ecologists and managers use potential vegetation to stratify or map vegetation for management or research purposes. For example, managers in California do not use potential vegetation but use existing or “actual” vegetation cover type instead to classify their forests for management (CALVEG)³ (<http://www.fs.usda.gov/detail/r5/landmanagement/resourcemanagement/?cid=stelprdb5347192>.) To help make our discussion more useful to managers in California, we provide a cross-walk table (app. 1) that links the Pacific Northwest Region (Region 6) potential vegetation types (see chapter 2, fig. 3-1) to Pacific Southwest Region (Region 5) existing vegetation classes. We also note, where appropriate, what the CALVEG classes might be for a given potential vegetation type. Most of our discussions in the text use estimated potential vegetation types for California and the rest of the Plan area based on a provisional map prepared by Michael Simpson (ecologist, Deschutes National Forest) (fig. 3-4).

³ One reason given for doing this is that in California vegetation, historical fire frequencies were quite high and the time since fire exclusion has been too short (e.g., 100 years) to really know what the capacity (potential future vegetation) would have been in the absence of disturbance. For purposes of this document, we use potential vegetation types, because we have a classification and map of these that covers the entire NWFP area (e.g., Simpson 2013), and there is no existing vegetation classification and map for Oregon and Washington. The lack of consistent vegetation data layers between the two regions makes it challenging to apply the findings from one Forest Service region to another.

Moist and dry forests—

At a broad scale, forests of the NWFP area can be classified into moist forests (including the western hemlock, Sitka spruce [*Picea sitchensis*], coastal redwoods, Pacific silver fir [*Abies amabilis*], and mountain hemlock [*Tsuga mertensiana*] potential vegetation zones west of the crest of the Cascade Range in Oregon and Washington), and dry forests (mainly ponderosa pine [*Pinus ponderosa*], Douglas-fir, grand fir [*A. grandis*], and white fir [*A. concolor*] potential vegetation types) east of the Cascade Range and in southwestern Oregon and northern California (Franklin and Johnson 2012). We use this moist forest and dry forest classification to frame much of this chapter.

Disturbance Regimes

Fire regime classification—

For most forest types, fire was and continues to be the major landscape disturbance agent that resets succession or shifts its course to a new pathway (Reilly and Spies 2016). Other disturbance agents are important as well, including wind and biotic agents, but most disturbance regime classifications and maps focus on fire. We characterize the ecology of multiple disturbances for moist and dry forests in sections below. In this section, we focus on approaches to classifying historical fire regimes.

Most of our current understanding of historical fire regimes is based on frequency—empirical studies of severity proportions and spatial patterns at landscape scales are relatively few (Hessburg et al. 2007, Reilly et al. 2017). Fire disturbances occur along a continuum of frequency, severity (e.g., tree mortality), seasonality, spatial heterogeneity, and event sizes. While there is no single classification of disturbance regimes, they are often binned into regime types that are based on fire frequency and severity (Agee 1993, 2003). Average fire frequency interval classes of frequent (<25 years), moderately infrequent (25 to 100 years), infrequent (100 to 300 years), and very infrequent (>300 years) (Agee 1993) are often used, but other frequency classifications exist as well: e.g., ≤35, 35 to 200, and >200 years (Hann and Bunnell 2001, Hann et al. 2004, Rollins 2009, Schmidt et al. 2002).

A widely used classification of fire-severity regimes for vegetation uses three bins of basal area or canopy mortality:

low (<20 percent), mixed or moderate (20 to 70 percent), and high (>70 percent)⁴ (Agee 1993, Hessburg et al. 2016, Perry et al. 2011) (fig. 3-5). Other classifications have been used, often with higher thresholds for canopy cover loss or mortality (e.g., 75 to 95 percent) (Miller et al. 2012, Reilly et al. 2017). The classification of Agee (1993) was initially

⁴ Note that while individual patches can exceed 70 percent mortality, fires typically have such high levels of mortality in only a small fraction of their total area. For example, the high-severity area of the 1988 Yellowstone fires was 56 percent (Turner et al. 1994), and the high-severity percentage of the 2002 Biscuit Fire in the Klamath of Oregon and California was 14 percent with an additional 23 percent at moderate severity based on a sample of inventory plots (Azuma et al. 2004).

developed for the stand or patch scale, but the metric has also been applied to larger regional areas (Agee 1993, Heinselman 1981, Reilly et al. 2017) or entire fire events, which can create confusion about the meaning of fire severity (Hessburg et al. 2016): Is it a fine-grained mix of severities, or coarse-grained mix of high and low severity, or both? Severity can also be characterized in terms of fire-induced changes to soils (i.e., soil burn severity); however, we focus on vegetative effects in this chapter. Soil burn severity is used in Burned Area Emergency Response analyses and is often confused with burn severity to vegetation (Safford et al. 2007).

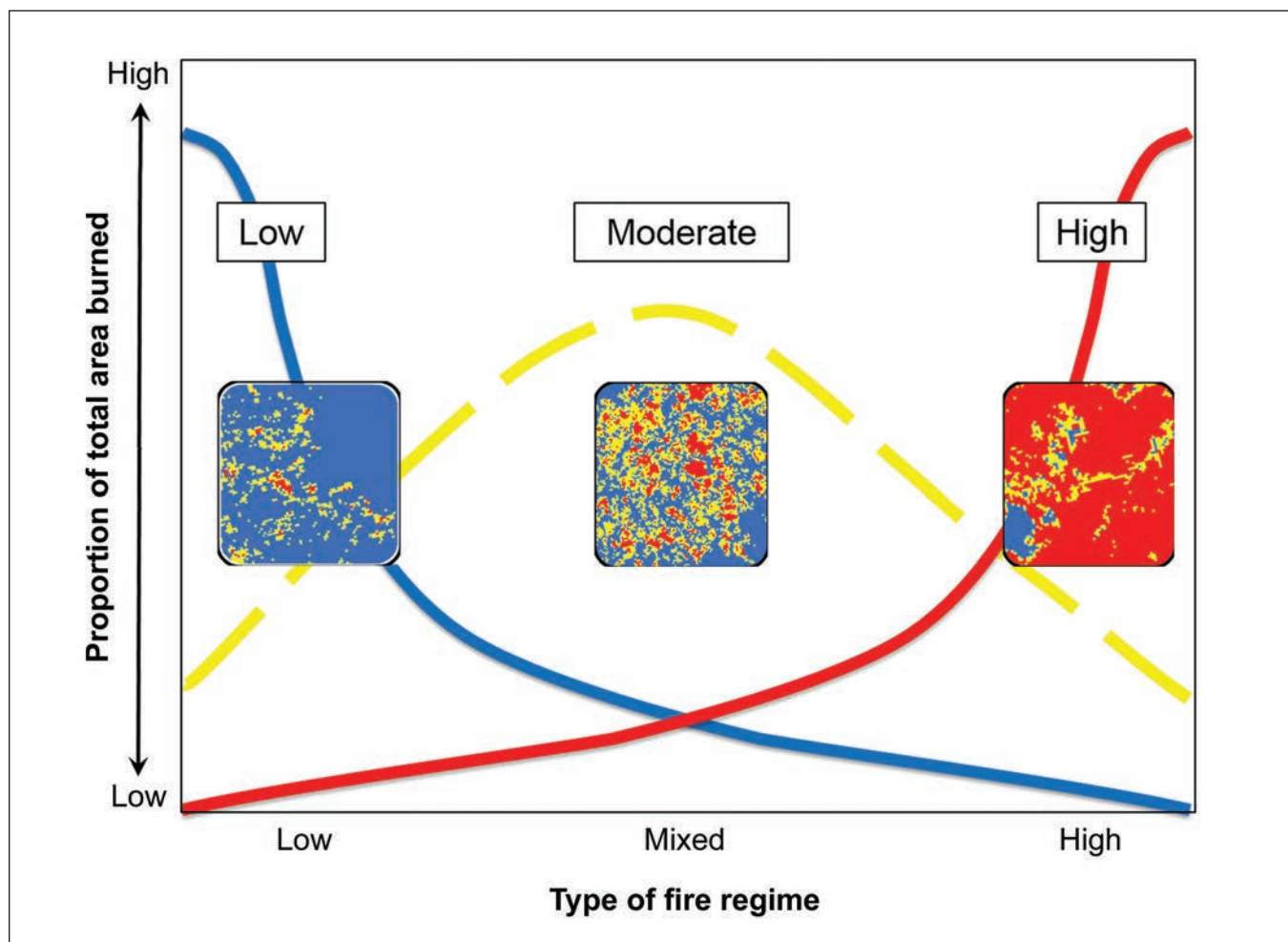


Figure 3-5—Conceptual diagram characterizing the proportions of low-, moderate-, and high-severity fires in three major fire regime classes. Inset panels represent idealized landscape dynamics associated with each regime based on proportions and size class distributions of patches at each of the three levels of severity. From Reilly et al. 2017, who modified it slightly from Agee (1993, 1998).

For management applications and regional planning, broad-scale regime classifications are typically used (Haugo et al. 2015), but fire history studies indicate that fire regimes can be relatively distinctive at topographic and landform scales (10^0 to 10^3 ac) (e.g., Taylor and Skinner 1998, Tepley et al. 2013). At landscape scales (ca. 10^3 to 10^6 ac), most fires occur as a mix of low, moderate, and high severity, driven by variation in topography, land forms, microclimate, surface and canopy fuels, soils, and vegetation, as we explore in later sections.

Combining fire regimes into broad average frequency and severity types is useful for regional planning (e.g., Rollins 2009, USDA and USDI 1994), but it oversimplifies variability that exists at finer scales, which is important for landscape planning and management. In general, simplifying fire into a few regime classes can obscure ecological diversity associated with fire effects (Hutto et al. 2016). Note that fire-severity proportions for any particular landscape or landform is often more restricted than implied by the broad ranges used to define broad regime classes. For example, for some landscapes in the very high frequency, low-severity regime (see below), the historical range of high-severity fire may be in the low end of the 0 to 20 percent⁵ range used to define this class.

A new fire regime classification—

For national and regional planning and management purposes, managers often use the LANDFIRE (Rollins 2009) fire regime classification. Our review of recent science in the NWFP region suggests that the national-scale product oversimplifies the fire history within the NWFP area. Thus, we developed a new classification and map (table 3-1, fig. 3-6) by synthesizing existing data on climate, lightning, and potential vegetation types (see app. 2 for methods) and fire history studies (app. 3).

This classification and map are meant to be a rough guide for understanding and visualizing ecological variation at regional scales and for framing a discussion about forest conservation and restoration science in the NWFP area (figs. 3-4 and 3-5). They reflect current understanding of fire ecology and geographic variability in the region. This typology is different from that used in the record of decision (USDA and USDI 1994) and FEMAT (1993) documents, which divided the NWFP region into moist and dry physiographic provinces but did not characterize variability in regimes within them. The physiographic provinces explained much of the variation in the physical environment, but they contain considerable subregional variations in vegetation types and fire regimes that are important to understanding the ecology of the forests in NWFP area. The potential vegetation types differ in distributions of fire regimes that occur within them (fig. 3-7), and the distribution of potential vegetation types differs between fire regimes, though the differences are relatively small between regimes within the moist or dry forests (fig. 3-8). Almost all fires in these regimes have mixed-severity effects, but they typically differ in the proportion and distribution of the high-severity effects. The very frequent low-severity regime, for instance, contains some area in high-severity fire patches at the scale of acres to tens of acres. The recognition of a drier, more fire-frequent mixed-severity zone on the west side of the Cascade Range in Oregon (fig. 3-6) is based on a number of studies (Agee and Edmunds 1992; Dunn 2015; Impara 1997; Reilly and Spies 2016; Tepley et al. 2013; Weisberg 2004, 2009). This regime, which typically burns with mixed severity and includes medium to large patches of high-severity fire, was first identified by Agee (1993), based in part on the fire history work of Morrison and Swanson (1990) from the western Cascades in Oregon.

Our classification also recognizes that the California portion of the NWFP area cannot be simply divided into a moist (Coastal province) and dry (Klamath and Cascades provinces) province for understanding succession and disturbance regimes. In fact, that area has relatively little of the “moist” forest that is characterized by historically

⁵ Odion et al. (2014) called for restricting definitions of historical low- and mixed-severity fires to regimes where crown fires and active or passive torching are generally absent. However, this classification would not be useful, as crown fires can occur in all fire regimes including low-severity regimes (Agee 1993), particularly when the regimes are intermixed, as they often are, where large landscape contain a range of topography, environmental, or vegetation conditions.

Table 3-1—Characteristics of major historical fire regimes used in this report and in figure 3-6

NWFP forest zone	Regime and landfire group	PVTs and cover types	Spatial characteristics
Moist	Infrequent (>200-year return intervals), stand replacing; LANDFIRE group V	PVT: wetter/colder parts of western hemlock, Pacific silver fir, mountain hemlock Cover types: Douglas-fir, western hemlock, Pacific silver fir, noble fir, mountain hemlock	Area dominated by large to very large patches (10^3 to 10^6) of high-severity fire; low- and moderate-severity fire also occurs. Small- to medium-size patches were most frequent.
	Moderately frequent to somewhat infrequent (50- to 200-year return intervals), mixed severity; LANDFIRE regime group III	PVTs: drier/warmer parts of western hemlock, Pacific silver fir and others Cover types: Douglas-fir, western hemlock, Pacific silver fir, noble fir	Mixed severity in space and time, typically including large (10^3 to 10^4 ac) patches of high-severity fire and areas of low- and moderate-severity fire. Small patches of high-severity would be common within lower severity areas.
Dry	Frequent (15- to 50-year return intervals) mixed severity; LANDFIRE regime group I and III	PVTs: Douglas-fir, grand fir, white fir, tanoak Cover types: Douglas-fir, white fir, red/noble fir, western white pine	Mixed-severity fire with medium to large (10^2 - to 10^3 -ac) patches of high-severity fire.
	Very frequent (5- to 25-year return intervals) low severity; LANDFIRE regime group I	PVTs: ponderosa pine, dry to moist grand fir, white fir Cover types: ponderosa pine, Douglas-fir, mixed pine, oak	Dominated by low-severity fire with fine-grained pattern (< 10^2 to 10^2 ac) of high-severity fire effects; large patches of high-severity fire rare in forests except in earlier seral stages (e.g., shrub fields).

NWFP = Northwest Forest Plan, PVT = potential vegetation type/zone used in the Pacific Northwest Region. Cover type = current vegetation classification used in the Pacific Southwest Region. LANDFIRE regime groups follow Rollins (2009).

infrequent, high-severity fires. Rather, forests in the California Coastal province were dominated by frequent, mixed-severity regimes, while the eastern Klamath and California Cascades were dominated by historical regimes of very frequent, low-severity fire.

Historical maps of high-severity burned forest patches from Washington and Oregon (data not available from California) (Plummer et al. 1902, Thompson and Johnson 1900) provide an independent source of primary data to evaluate the regional regime map. These maps support the hypothesis that the largest patches and percentage of forest burned by high-severity fire occurred in the infrequent high-severity regime; whereas the smallest patches and lowest area of forest burned by high-severity fire occurred in the very frequent/low-severity regime (fig.

3-9).⁶ The relatively high percentage of area burned in the infrequent fire regime may reflect elevated ignitions from Euro-American settlement activities, because lightning densities in these areas are low (fig. 3-10) and these forests are not typically fuel limited (Agee 1993). American Indian burning practices would have also been a historical component in some parts of the region, but the importance would have varied considerably among regimes (see chapter 11). For example, several studies (app. 3) have

⁶ These early 20th century maps are our best snapshots of this time period but do not necessarily represent the range of variability in fire sizes that would occur in these regimes over time. This is especially true for the infrequent, high-severity regime where sample of historical fires is small and extremely large patches of fire may have occurred in past centuries.

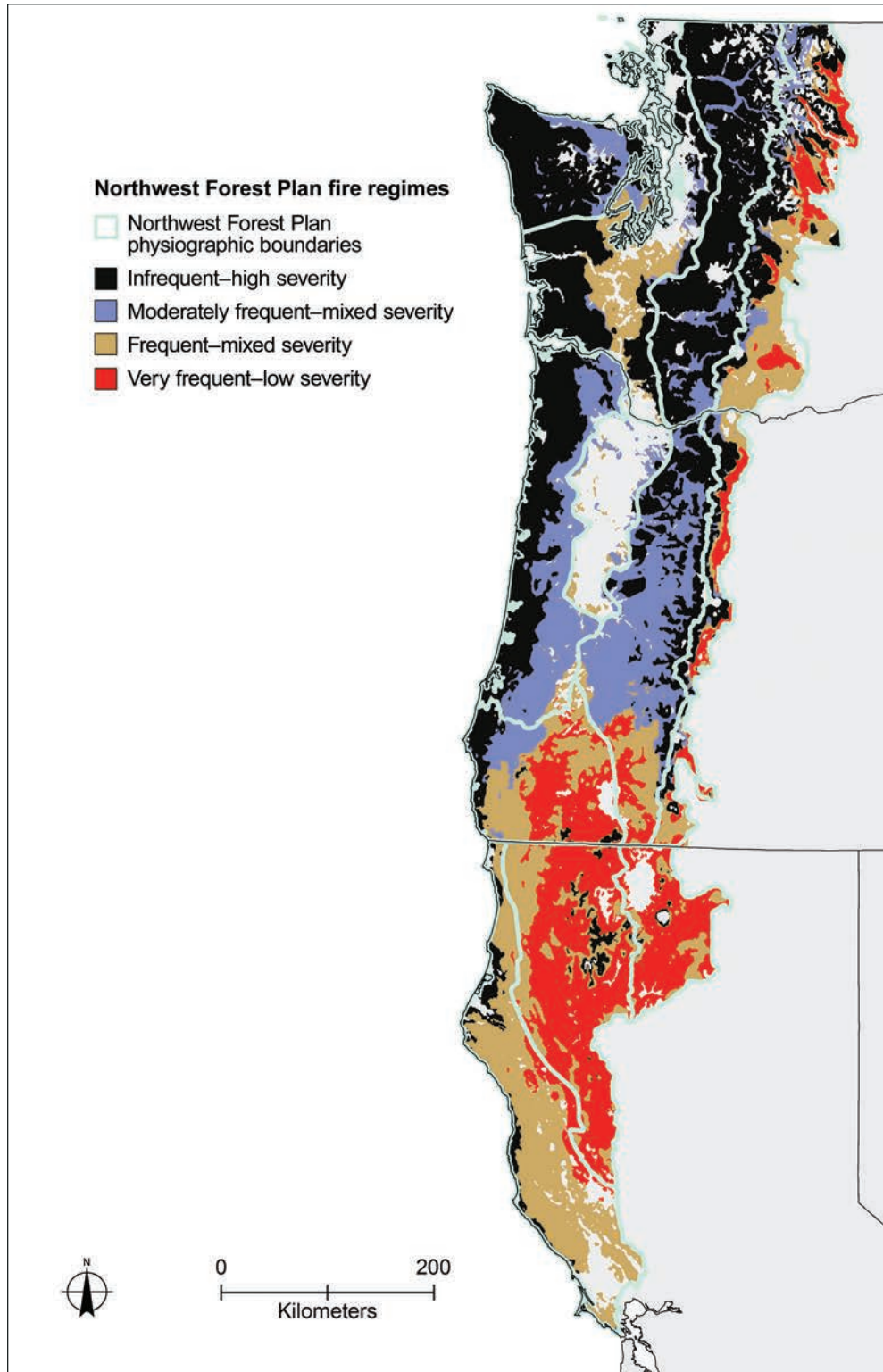


Figure 3-6—Generalized fire regimes for the Northwest Forest Plan (NWFP) area based on climate and lighting density. Fire frequency, particularly in coastal areas of California, may be underestimated because historical ignitions by American Indians are not included in the model. See table 3-1 for more information about the regimes and appendix 2 for methods. Moist forests are typically associated with the infrequent and moderately frequent regimes, while dry forests typically are associated with the frequent and very frequent regimes.

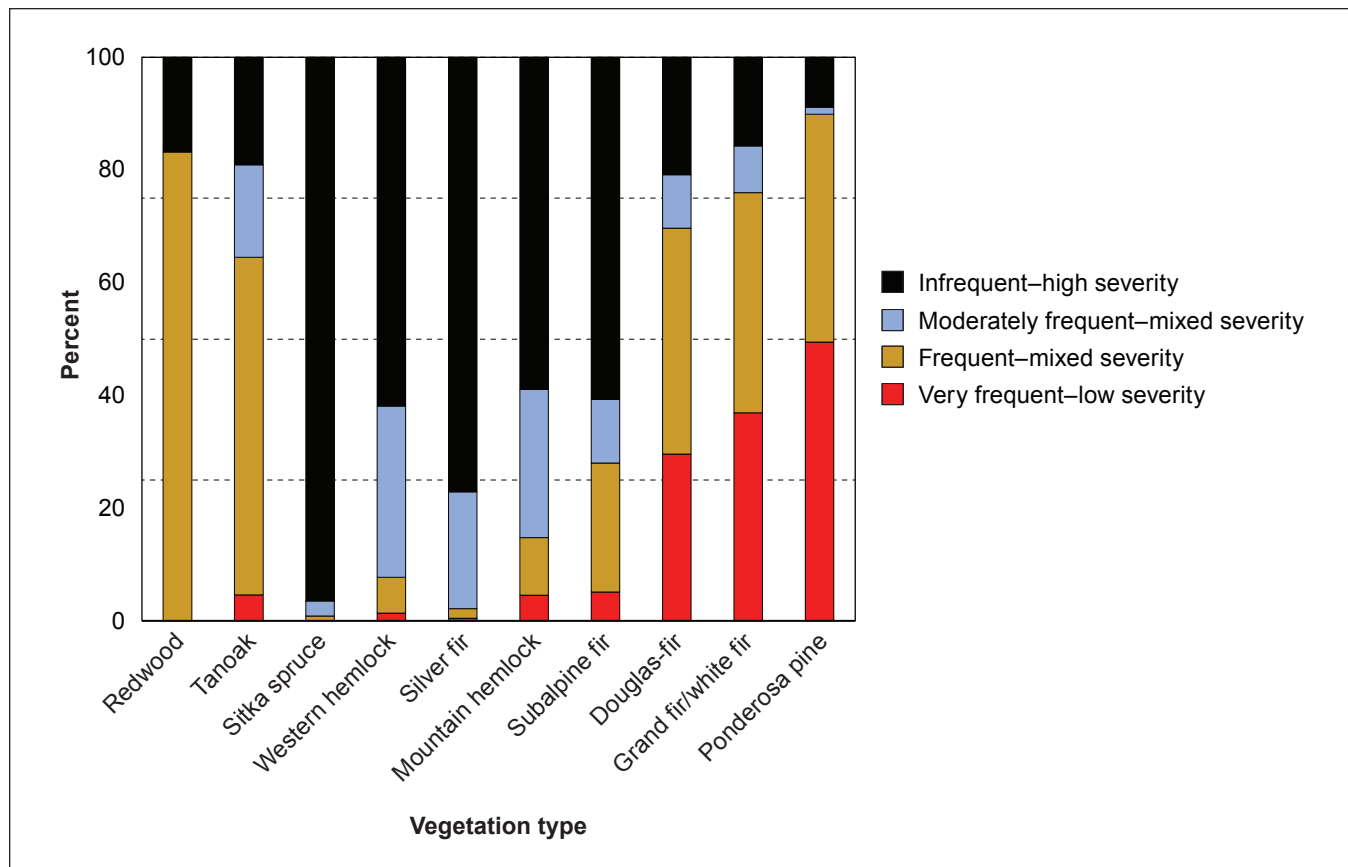


Figure 3-7—Percentage of major potential vegetation types (PVTs) in the four different fire regimes. Small percentages of a fire regime within a PVT may be a result of errors in the PVT maps, fire regime maps, or both.

noted that burning by American Indians likely caused fires to be very frequent (<29 years) (app. 3) in the redwood (*Sequoia sempervirens*) forests of northern California, although the map based upon climate and incidence of lightning classifies those areas as moderate frequency, mixed-severity fire regimes.

The lack of close correspondence of fire regime with major potential vegetation type or climate zone (figs. 3-4 and 3-6) indicates that vegetation type at the zone (series) level (at climax) and fire regime do not necessarily respond in the same way or at the same scale to variation in the environment (Kellogg et al. 2007) (see discussion of the regimes for more information). If disturbance regime variation within subregions and landscapes is not taken into account, efforts to retain or restore biological diversity based on historical fire regimes may not be effective or may have undesirable effects.

Disturbance regimes of moist forests—

Moist forests occur primarily west of the crest of the Cascades in Washington and Oregon, including the Coast Range forests, and on the west slope of the Cascades, they extend into high-elevation wet and cool forests (fig. 3-4). Potential vegetation types are dominated by western hemlock, Pacific silver fir, and mountain hemlock (fig. 3-8). Sources of stand-replacement disturbance in this region included fire, wind, and volcanic eruptions. Insects and diseases, especially root diseases, typically created finer grained disturbances such as canopy gaps (e.g., 0.1 ac [0.04 ha]) to several acres in size) (Dickman and Cook 1989, Spies et al. 1989). In California, moist forests with infrequent fire regimes are confined to relatively small areas along the coast and in some higher elevations.

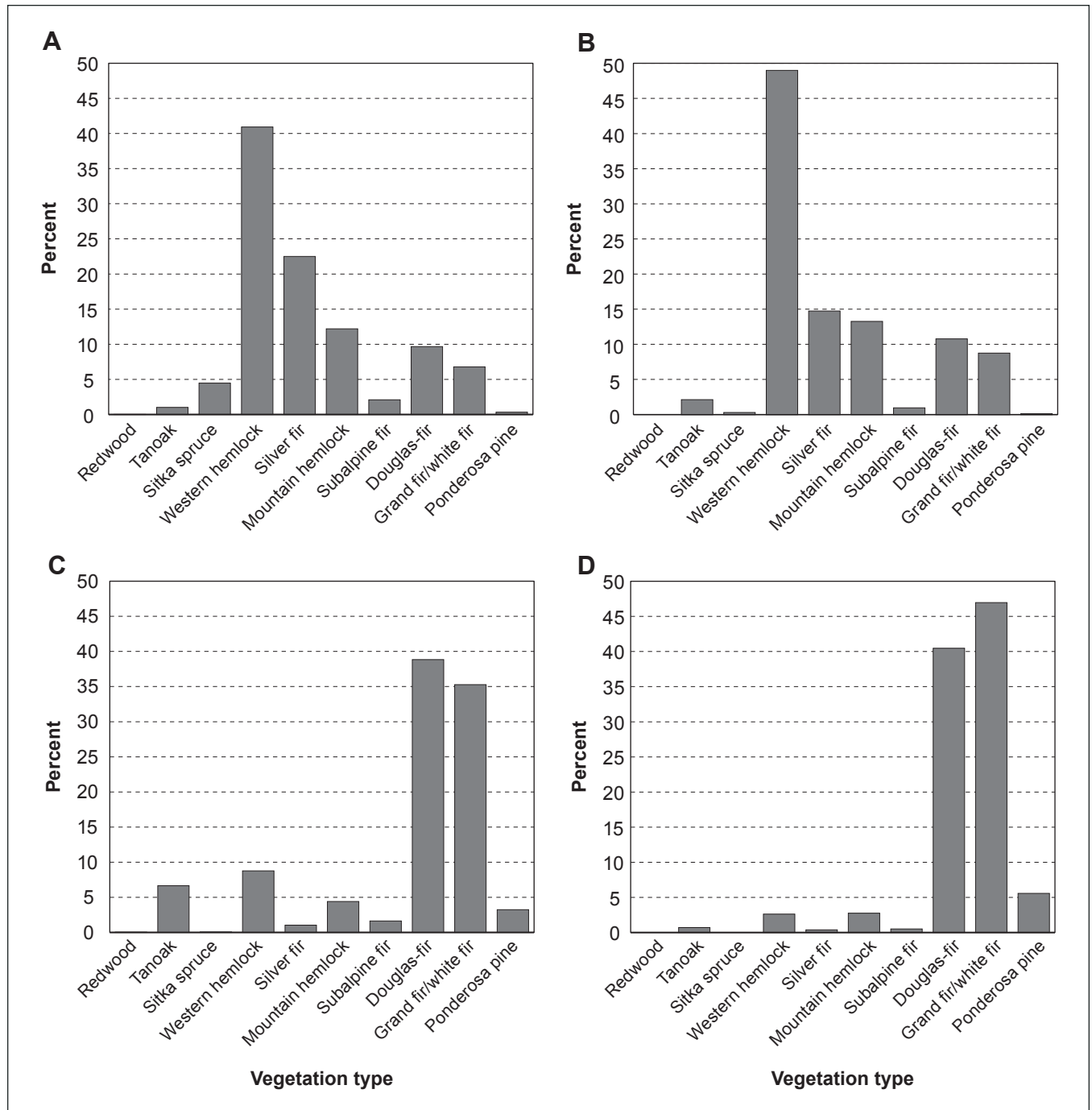


Figure 3-8—Distribution of major potential vegetation types (PVTs) within the (A) infrequent, high-severity regime; (B) moderately frequent, mixed-severity regimes of the moist forests; (C) frequent, mixed-severity regime; and (D) very frequent, low-severity regimes of the dry forests. Only major PVTs are shown. See appendix 1 for crosswalk to California vegetation types. Forests currently dominated by ponderosa pine would occur within the Douglas-fir, grand fir, and white fir PVTs.

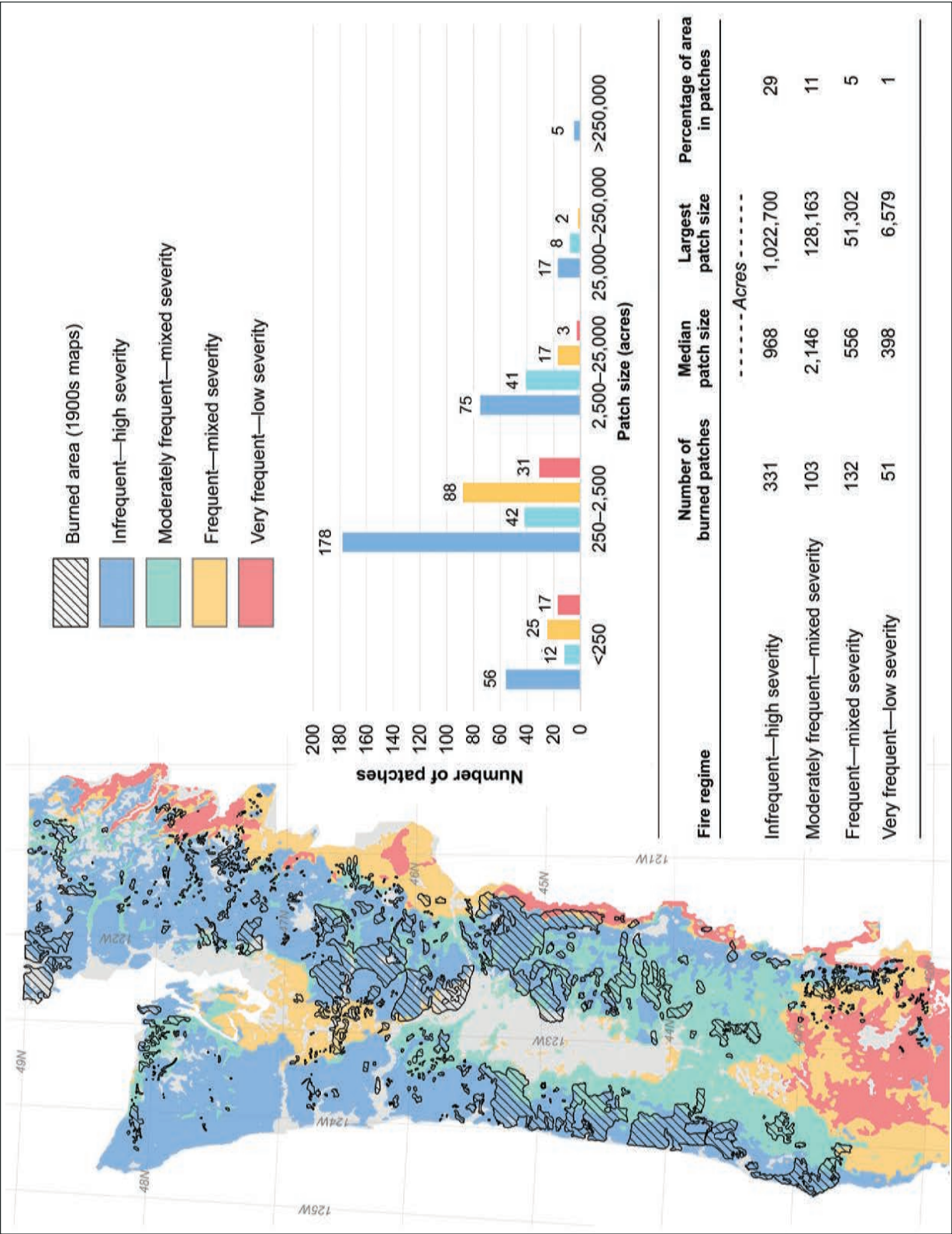


Figure 3-9— Historical (1900–1902) patterns of early-successional postfire patches (where “destruction of timber was nearly or quite complete... areas... with only a partial destruction are not here represented”) (Plummer 1902). Note how patches were fewer and smaller in the high-frequency/low-severity regime compared to the other regimes. Many of these fires would have been ignited by settlement and logging activities but would have burned before fire suppression was effective in most cases. Burned forest patches were digitized from Thompson and Johnson (1900) and Plummer et al. (1902). Data were not available for California.

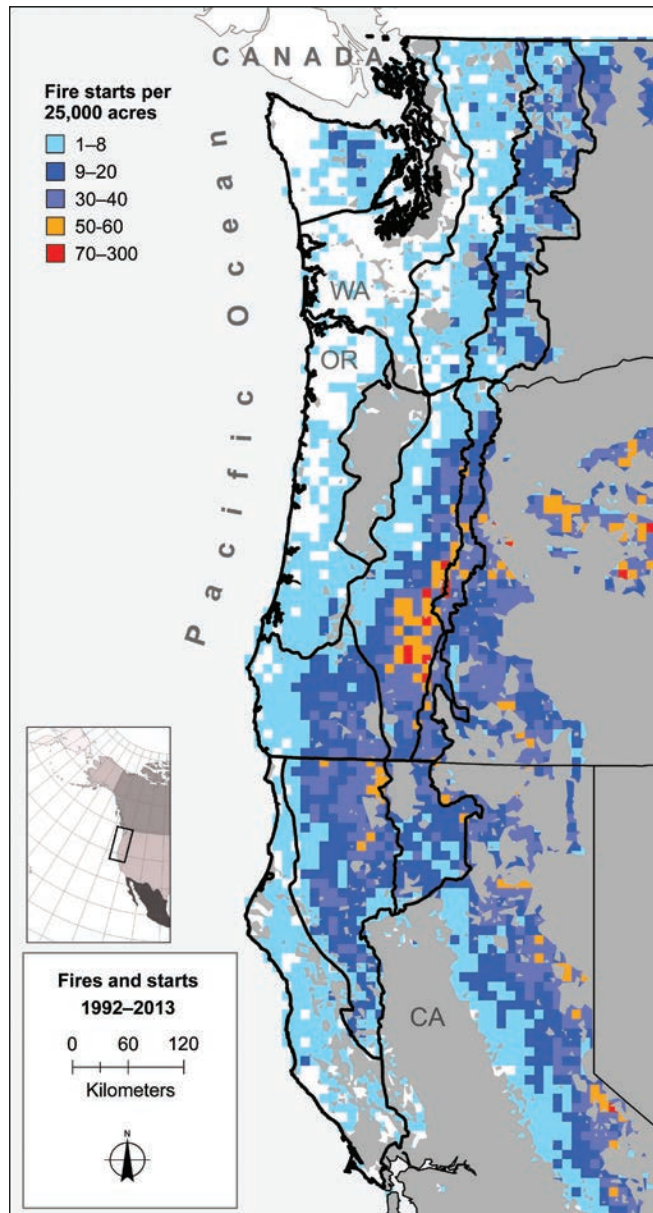


Figure 3-10—Density of lightning-ignited fires per 25,000 ac on forest lands in the Northwest Forest Plan area for the period 1992–2013. Black lines are physiographic provinces as delineated in figure 3-4.

Two major fire regimes can be recognized within moist forests: infrequent (>200 -year return interval) and dominated by high severity; and moderately frequent to somewhat infrequent (50- to 200-year return interval) fire with mixed-severity patterns (table 3-1). The infrequent regime is characterized by relatively long fire-return intervals and dominance of high-severity fire in medium to very large

patches. Historically, mean fire-free intervals averaged greater than 200 years with some areas not experiencing fire for more than 1,000 years (Agee 1998). Although most of the area in high-severity patches is contained within larger patches in this regime, individual fires could have high-severity (>70 percent mortality) patches ranging from quite small (1 to 25 ac [0.04 to 20 ha] to very large ($>10^6$ ac [~ 400 000 ha]) (Agee 1993, 1998). Given the historical infrequency of such fires and the tendency for high-severity fire to erase information about previous fires, there are few empirical studies based on actual fire occurrence (using fire scars), and most of our collective knowledge is derived from studies that used age-class data to reconstruct large-scale fire rotations (Hemstrom and Franklin 1982) and maps of historical fires (fig. 3-6). Climate variation at century scales controlled fire frequency and successional dynamics (Gavin et al. 2007, Long et al. 1998, Walsh et al. 2015). Fire frequency, for instance, was relatively high during the Medieval Warm climate anomaly about 1,000 years ago, but declined during the Little Ice Age between 1400 and 1850 BP. The low fire frequency in these systems was due to chronically high fuel moistures and infrequent lightning ignitions (Agee 1993) (fig. 3-10). Large high-severity fires would typically occur during unusually dry periods when synoptic weather patterns created strong hot and dry east or north winds (Agee 1993; Morrison and Swanson 1990; Weisberg 1998; Weisberg and Swanson 2001, 2003), but even those fires typically left patches with surviving live trees, which would contribute to regeneration and habitat diversity. As in other settings, the frequency-size distribution of fires followed a negative exponential distribution; i.e., the smallest fires were the most numerous, and the largest fires accounted for the majority of area burned (e.g., see Moritz et al. 2011).

Humans have played a role in fire occurrence in these forests. American Indian use of fire would have contributed to fire regimes, especially in drier regions and in local areas near Indian settlements in western valleys and coastal areas (Agee 1993, Walsh et al. 2015) (see chapter 11). We did not adjust the mapping of fire regimes for potential effects of Indian burning. Scientific opinions differ regarding the contribution of Indian burning to these forests over evolutionarily relevant time scales. Clearly, the contribution of such

burning was locally important in many areas. Euro-American influence began around the time of settlement (early 1800s) and coincided with warming and progressively drier weather patterns as the Little Ice Age began winding down, potentially exacerbating fire activity (see Weisberg and Swanson 2003).

In the drier parts of the moist forest subregion, fires were more frequent and mixed in severity, although medium to large patches of high mortality were present (table 3-1). The moderately frequent to somewhat infrequent regime (Morrison and Swanson 1990, Van Norman 1998) occurred across a range of potential vegetation types (fig. 3-8), along the eastern slopes of the Olympic Mountains

and Coast Ranges, and the interior valleys extending to the western slopes of the Cascades in Oregon (fig. 3-6). The climate there is warmer and drier than in the infrequent fire regime, and lightning ignitions are more frequent (fig. 3-10). Patches of high-severity fire could be highly variable and were probably somewhat smaller than in the infrequent high-severity regime (Morrison and Swanson 1990) (fig. 3-9). Mixed-severity fires likely affected many older forests (Weisberg 2004). For example, many of the existing old-growth trees in the southern western Cascades of Oregon and interior parts of the Coast Range in Oregon showed evidence of low-severity fire occurrence (fig. 3-11). Severe windstorms also played a role in forest dynamics

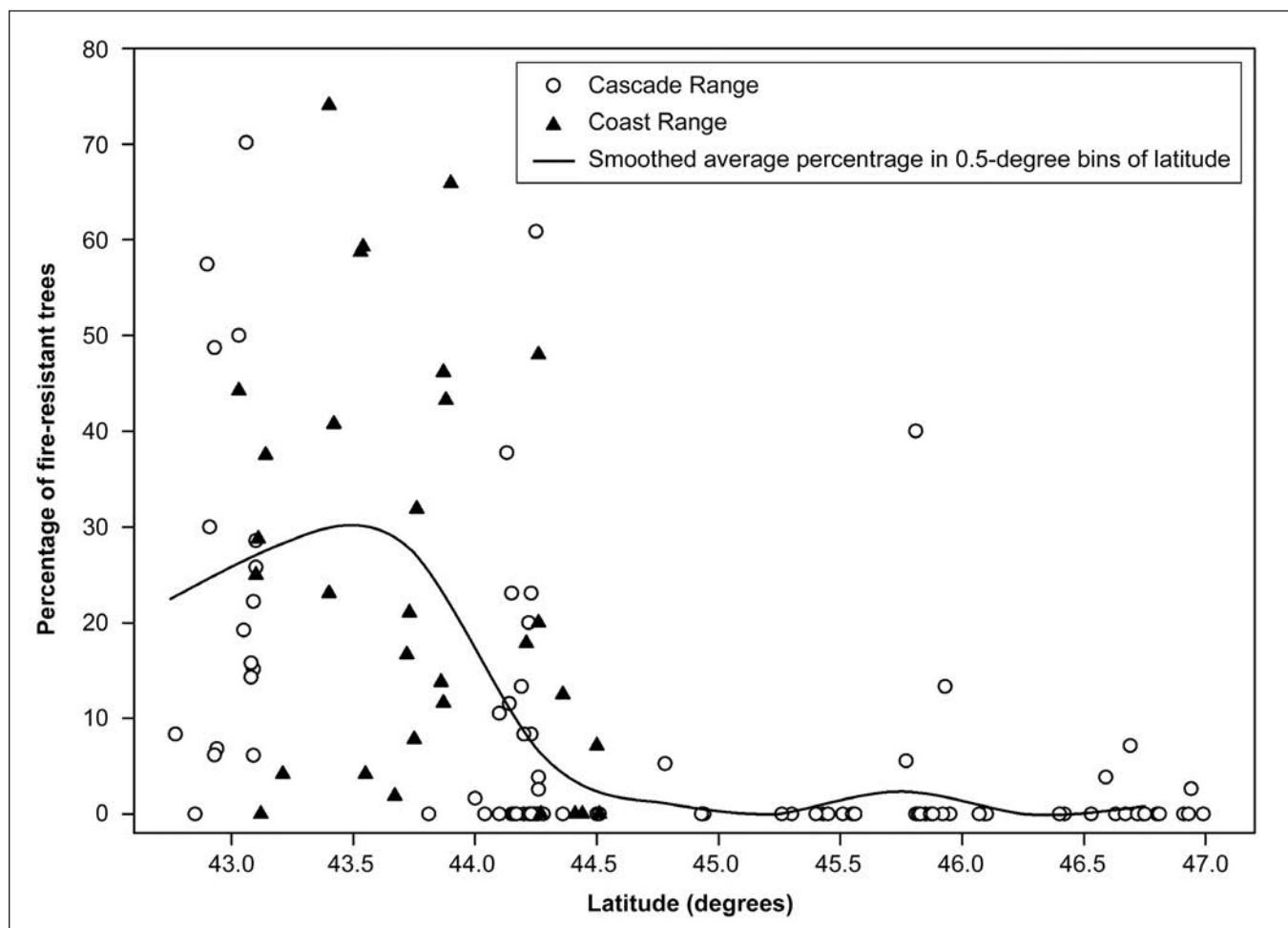


Figure 3-11—Percentage of fire-resistant mature and old trees with evidence of fire (scars or charred bark) in the western Cascades and Oregon Coast Range in relation to latitude. Line is smoothed running average in 0.5° bins. The increase in evidence of fire on tree boles around latitude 44.5° N in Oregon (about the latitude of Corvallis) indicates a shift from infrequent, high-severity to moderately frequent, mixed-severity fire regimes moving from north to south (right to left). Data source: Spies and Franklin (1991).

west of the Cascade crest (Knapp and Hadley 2012). Wind occasionally created large stand-replacement patches and frequently small gap disturbances across all forest types in the region. While the frequency of wind disturbance is greatest near the coast (Harcombe et al. 2004) and in the Columbia Gorge (Sinton et al. 2000), infrequent large regional-scale wind events, such as the 1805 “perfect storm” experienced by Lewis and Clark (Knapp and Hadley 2011), the 1962 Columbus Day windstorm (Lynott and Cramer 1966), and the 1981 Big Blow of November 14th can affect forests across the west side of Oregon and Washington. The 1962 storm may be the largest natural disturbance event in regional forest history, blowing down 11 billion board feet of timber across Washington and Oregon, in concentrations of over 80 ac/mi² (12.5 ha/km²) in some areas (Teensma et al. 1991). The frequent occurrences of large windstorms in coastal areas control tree growth, forest structure, and successional patterns (Knapp and Hadley 2012). More frequently, windthrow disturbances are typically related to patterns of topographic exposure, which can concentrate windflow (Harcombe et al. 2004, Sinton et al. 2000, Wimberly and Spies 2001), root disease, or edges of older and younger patches of forests (Franklin and Forman 1987, Sinton et al. 2000) created by clearcutting or other stand-replacement disturbances.

Biotic disturbance agents play important roles in succession, and in ecosystem processes and patterns of moist forests (table 3-2). They also play important roles in producing dead and damaged trees that serve as wildlife habitat (Bull 2002). These agents primarily include root diseases and bark beetles, although foliage diseases, defoliators, heart rots, rust diseases, and dwarf mistletoes can also be quite important. Root disease fungi and related organisms cause root death, heart rot of large roots and tree butts, reduced tree productivity, top dieback, and tree mortality, while interacting with bark beetles or other mortality agents to influence gap dynamics and stand structure (Hansen and Goheen 2000, Lockman and Kearns 2016). *Phellinus sulphurescens* (syn *Poria weirii* or *P. weirii* in the older literature) clones are thought to occur on about 5 to 16 percent of the landscape in the moist forests (Lockman and Kearns 2016, Washington

State Academy of Sciences 2013), for example. Root rot diseases are often called, “diseases of the site” in the sense that once established in a stand, the fungi can persist for decades on belowground wood depending on management or compositional changes (Hadfield et al. 1986, Shaw et al. 2009).

Foliage disease fungi can be major disturbance agents that influence competitive relationships and tree productivity potentially throughout a climatic region (Bednářová et al. 2013). However, foliage diseases in Pacific Northwest forests are best known in young plantation forests, and are poorly studied in natural, or especially, older forests (Shaw et al. 2011). Swiss needle cast, caused by the native fungus *Phaeocryptopus gaeumannii*, is currently causing an epidemic in managed Douglas-fir coastal forests of Oregon and Washington state, within about 35 mi (56.3 km) of the Pacific Ocean, reducing plantation productivity an average of 23 percent within a study area of the northwest Coast Range of Oregon (Maguire et al. 2002, 2011, Navarro and Norlander 2016, Ramsey et al. 2016, Ritóková et al. 2016). The disease is particularly associated with lower elevations of the infrequent–high-severity fire regime (fig. 3-6). The role of foliage diseases in the development of forest stands, and in particular, old-tree crown dynamics, remains elusive. It is generally thought that maintaining tree species diversity, canopy complexity, and adherence to site compatible seed zones reduces the threat of foliage diseases to forest health (Shaw et al. 2009).

Bark beetles and wood borers are diverse, but major disturbance from mortality is mostly associated with climatic events such as drought, ice/snow breakage, and windthrow (Furniss and Carolyn 1977). Two particularly important species are the fir engraver (*Scolytus ventralis* (LeConte)) in true firs (Ferrell 1986) and the Douglas-fir beetle (*Dendroctonus pseudotsugae* (Hopkins)) in Douglas-fir (Furniss and Kegley 2014). Mortality from both insects is associated with root diseases and drought, and, in the case of the Douglas-fir beetle, with windthrow events (Furniss 2014a, 2014b; Goheen and Willhite 2006). Typically, flareups of mortality from this beetle persist for a few years and then abruptly subside (Furniss and Carolyn 1977, Goheen and Willhite 2006).

Table 3-2—Major biotic disturbance groups, effect on trees, and ecological influences in forests of the Northwest Forest Plan area

Disturbance group^a	Tree effects	Ecological influences
Root diseases	Major mortality agent Growth reduction Root death Root/butt heart trots	Alters stand composition/structure Creates snags, down wood Wildlife cavities Creates ant/termite habitat Attracts bark beetle mass attack Increases surface fuels
Live tree decays	Wood volume reduction Increased windsnap	Wildlife cavity creation Reduced carbon sequestration Creates ant/termite habitat
Foliage diseases	Reduce foliage retention Reduced growth Carbon starvation	Less competitive in stands Reduced carbon sequestration Alters stand composition/structure
Cankers and rusts	Branch, top, tree death Foliage loss Tree deformation	Reduced carbon sequestration Reduce host species abundance Wildlife habitat
Dwarf mistletoe	Growth reduction Top, branch, and tree death Branch and tree deformation Increased susceptibility to other agents	Alters forest structure/composition Encourages passive crown fire Wildlife habitat platforms Influence with fire
Bark beetles	Major mortality agent Patch attacks on bole Top and branch death	Alters composition/structure Increases forest fuels Wildlife habitat
Defoliators	Growth loss Top dieback Mortality	Alters composition/structure Reduces canopy density Wildlife habitat impacts
Aphids, adelgids and scale insects	Growth loss Leaf, branch, and tree death	Alters forest structure Reduced carbon sequestration
Terminal and branch insects and pitch moths	Tree leader death Stunted growth Tree deformation	Forest structure Reduced competitive ability

^a Groups from Shaw et al. (2009).

Source: Furniss and Carolin 1977, Goheen and Willhite 2006, Scharpf 1993, Shaw et al. 2009, Wood et al. 2003.

Other important biotic agents include the hemlock dwarf mistletoe (*Arceuthobium tsugense* Rosendahl), which is the only known moist forest dwarf mistletoe, and can dramatically influence forest structure (Muir and Hennon 2007). The plant occurs localized in western hemlock-dominated forests, where it is estimated to infect 10.8 percent of the western hemlock trees in Oregon (Dunham 2008). Hemlock dwarf mistletoe has a strong connection to fire history (Shaw and Agne 2017); more frequent fires favor less mistletoe.

Disturbance regimes of dry forests—

This region includes the mid to lower elevations of the eastern Cascades from Washington to California, southwestern Oregon, in the Klamath region, and inland portions of the California Coast Range. It spans a range of dry forest potential and current vegetation types, including ponderosa

pine, Douglas-fir, and white fir (figs. 3-4 and 3-6; table 3-1). Fire is the major stand-replacement disturbance in this region followed by outbreaks of major forest insects.

The more moist and productive part of this region experienced a frequent, mixed-severity regime with fire-return intervals of 15 to 50 years (Agee 1991, Agee et al. 1990b, Stuart and Salazar 2000, Taylor and Skinner 1998, Van Norman 1998, Whitlock et al. 2004, Wright and Agee 2004). Fire events contained medium to large patches of high-mortality and extensive areas of low- and moderate-severity fire. The 2002 Biscuit Fire is an example of such a fire (Halofsky et al. 2011, Thompson and Spies 2009) (fig. 3-12). The occurrence of mixed-severity fire even at short fire-return intervals (e.g., <25 years) probably reflects the higher moisture conditions and site productivities in parts of this regime in comparison to the very frequent, low-severity dominated regime in



Thomas Spies

Figure 3-12—Mosaic of high-severity burn patches in a portion of the 2002 Biscuit Fire in southwest Oregon in an area classified as historically supporting a frequent, mixed-severity fire regime (fig. 3-6). A large portion of the area with surviving tree canopies experienced low-severity surface fire.

California or the eastern Cascades. Patterns of mixed-severity patches were historically shaped by prevailing topographic features (Beaty and Taylor 2001; Hessburg et al. 2015, 2016; Taylor and Skinner 1998; Weatherspoon and Skinner 1995) with variable proportions of both surface and crown fires accounting in part for tree mortality in mixed-severity fire regimes (Perry et al. 2011, Stephens and Finney 2002).

The very frequent (<25 years) low-severity regime occurs in the driest forests⁷ of the NWFP area in a variety of pine, dry Douglas-fir, dry grand or white fir, and oak potential and current vegetation types (figs. 3-4 and 3-6, table 3-1, app. 1). Historically, fires burned very frequently, with average fire intervals between 5 and 25 years (Bork 1984; Everett et al. 2000; Sensenig et al. 2013; Soeriaatmadja 1965; Taylor and Skinner 1998, 2003; Weaver 1959), although for many forests the range was much narrower. Overall, tree mortality from fire was low, with typically <20 percent of the trees killed in fires, and most high-severity effects occurring in very small patches (<1 ac [<0.40 ha]). Fire severity was primarily influenced by fine-scale patterns of surface fuels and topography (Churchill et al. 2013, Larson and Churchill 2012). Fuels were reduced frequently enough that active crown fire was infrequent. Frequent fires often created multicohort stands with low tree density and canopy cover (Hagmann et al. 2013, 2014; Sensenig et al. 2013). Larger patches (>250 ac [>101 ha]) of high severity could occur but were uncommon in most areas (Agee 1993, Rollins 2009; Skinner 1995; Taylor and Skinner 1998, 2003) and were linked to topography (Taylor and Skinner 1998, 2003). The forested landscape was dominated by open forests with islands of denser vegetation, including clumps of trees of various sizes (Churchill et al. 2013, Hessburg et al. 2007, Larson and Churchill 2012, Lydersen et al. 2013, Perry et al. 2011). Some scientists (e.g., Baker 2012) dispute the idea that these dry forests experienced a regime dominated by frequent, low-severity fire, and argue instead that they commonly experienced larger patches of high-severity fire (see section on alternative viewpoints below for more discussion of this).

⁷ In the Klamath and southern Cascades of California, these regimes occur where the climate is characterized by long warm/dry seasons but relatively high precipitation, which is concentrated in the winter months.

Wind is not a major disturbance agent in drier forests of the region that are typically inland from coastal areas, and south of areas where the strongest windstorms occur. Coastal California is south of most of the mid-latitude cyclones that affect the Oregon and Washington coast (Lorimer et al. 2009). Coastal redwood forests experience winter storms and high winds, but effects appear to be limited to canopy damage and scattered blowdown of trees on high ridges (Hunter and Parker 1993, Lorimer et al. 2009). Drier ponderosa pine, Douglas-fir, and mixed-conifer forests experience scattered windthrow that creates canopy gaps and fine-scale pit and mound microtopography (Weaver 1943), but we are not aware of studies that document occurrence of larger patches of windthrow. Reilly and Spies (2016) report that between the 1990s and mid 2000s, wind was a very small component of all natural sources of mortality in dry forests of the Pacific Northwest. Agee (1994) reported similar results for the dry interior forests.

Major biotic disturbance agents in dry forests include several root diseases and host specialized dwarf mistletoes as chronic long-term stand influences that are associated with creating complexity in forest patches by killing and deforming trees, creating snags and gaps, and influencing fuels and fire (Goheen and Willhite 2006, Hadfield et al. 1986, Hawksworth and Wiens 1996, Lockman and Kearns 2016, Shaw and Agne 2017) (table 3-2). Major bark beetle and defoliator disturbances tend to be episodic, although individual old-tree death caused by bark beetles is chronic in some forests. Large outbreaks are more common in the eastern slope of the Cascades than in northern California, where tree species diversity, complex terrain, geological diversity, and contrasting site microenvironments may reduce the potential for widespread outbreaks. Heart rots, rust diseases, cankers, as well as foliage and tip diseases and insects may be locally significant, especially heart rots, which create cavities for wildlife (Bunnell 2013).

Root diseases are widespread in dry forests (Filip and Goheen 1984, Hadfield et al. 1986, Lockman and Kearns 2016), where they play an integral part in forest stand dynamics and canopy gap formation. In northwestern California, Hawkins and Henkel (2011) found that root diseases caused more mortality and gap formation in white

fir than Douglas-fir, which in the absence of fire, allowed Douglas-fir to better persist in forest stands. This is not always the case in the dry forests.

Dwarf mistletoes are host specialized parasitic seed plants that are a major influence on dry forest structure. Host-specialized mistletoes infest nearly all species, where they create structures such as witch's brooms, dead tops, dead branches, and fuel ladders (Hawksworth and Wiens 1996, Mathiasen and Marshall 1999, Shaw et al. 2004). A key ecological function of dwarf mistletoes is the creation of wildlife habitat structures via their large witch's brooms, which provide nesting and roosting platforms for a variety of forest birds and other small mammals (Shaw et al. 2004). Douglas-fir dwarf mistletoes can provide the majority of nesting sites for the spotted owl in dry interior forests (Buchanan et al. 1995, Forsman et al. 1984). Dwarf mistletoe distribution and abundance is related to fire history; with more regular fire there is less dwarf mistletoe because heavily infested trees are prone to torching or passive crown fire initiation (Shaw and Agne 2017). Although fire influences dwarf mistletoe, dwarf mistletoe also influences fire behavior by creating complex fuels structures, contributing to surface fuels, increasing ladder fuels, decreasing canopy base height, and increasing canopy bulk density.

Bark beetles are associated with most mortality events in dry forests, however, determining whether the beetles are to blame for individual tree mortality can be a challenge. Drought, dwarf mistletoe, root diseases, defoliators, and other biotic or abiotic factors can all predispose weakened trees to bark beetle mass attack. Bark beetle outbreaks can also be initiated by long-term drought events, and these outbreaks can last well over a decade. Bark beetles are also host specialized, and they influence forest stand structure and development by killing specific tree species. In the aftermath, tree mortality associated with beetle outbreaks can contribute significantly to forest fuels, but it can take more than a decade or two for the snags of the former forest structure to fall down and accumulate on the forest floor. Major bark beetle outbreaks typically occur in dry forests east of the Cascade crest where expansive stands of lodgepole pine (*Pinus contorta*) have been hit very hard by mountain pine beetle (MBP) (*Dendroctonus ponderosae*) (Gibson et al.

2009). Recent large bark beetle mortality events associated with periods of extended drought in the southern and central Sierra Nevada of California suggest that the potential for major climate change-driven outbreaks is ongoing and may result in species conversion in some areas (Moore et al. 2017). The interaction of fire with prior MPB events has become a significant research emphasis following large outbreaks throughout western North America. Following MPB mortality, canopy fuels decrease drastically within a few years, and depending on composition of the stand, surface fuels will significantly increase with time (Hicke et al. 2012).

Defoliators on the east side of the Cascade Range are a major disturbance agent in forest stands, with the western spruce budworm, Douglas-fir tussock moth (*Orgyia pseudotsugata*), pine butterfly (*Neophasia menapia*), and Pandora moth (*Coloradia pandora*) potentially able to defoliate large regions (Furniss and Carolin 1977, Goheen and Willhite 2006). Outbreaks of the western spruce budworm (*Choristoneura occidentalis*) have not occurred in dry forests of California and southwestern Oregon (Brookes et al. 1987), although the Douglas-fir tussock moth may defoliate true firs, and the Pandora Moth may affect ponderosa pine (Brookes et al. 1978, Wood et al. 2003). Defoliators have the potential to shift composition of stands to nonhosts owing to reduced growth and mortality effects, as well as increased potential for bark beetle infestation in defoliated trees (Brookes et al. 1978, 1987). The interactions of fire with forest defoliators suggest a negative association of fire and defoliated stands (Meigs et al. 2015).

Forest Succession and Landscape Dynamics

Moist forests—

Succession—Our synthesis of this regime is primarily based on studies from Douglas-fir and western hemlock forests (i.e., the western hemlock potential vegetation type) (Franklin et al. 2002, Oliver and Larson 1996, Reilly and Spies 2015, Spies et al. 1988). Patterns of postfire and postwind stand-replacement succession for other potential vegetation types in this fire regime, which have received less study (e.g., mountain hemlock in Oregon and Washington, Pacific silver fir potential vegetation types) may have been generally similar, but they differ in a number of ways, including species composition, varied

pathogen and insect associations, and slower rates of structural and compositional development. These potential vegetation types also likely have lower levels of total biomass relative to Douglas-fir/western hemlock forests in late-successional stages, owing to shorter and cooler growing seasons.

The archetypal or standard model of forest succession in this forest region and under these disturbance regimes has been characterized in many papers but is developed in greatest depth by Franklin et al. (2002), and most recently by Franklin and Johnson (2017) and Franklin et al. (2018) (fig. 3-13). Simply stated, after a stand-replacement disturbance such as wildfire or windstorm (1) considerable dead and live legacies of the disturbance remain for decades; (2) new shade-intolerant and tolerant plants and early-seral associated wildlife colonize a site; and (3) a dynamic mix of nonforest and forest plant species develops and persists until conifer canopy closure, which may take between 30 and 100 years. The forest then goes through a process of structural and compositional changes and stages driven by growth, competition, immigration of shade-tolerant species, and fine- to moderate-scale mortality events that create canopy gaps of various sizes (Bradshaw and Spies 1992, Spies et al. 1990). These canopy gaps can promote growth of shade-tolerant trees growing in the understories of densely shaded forests. This is not the only successional pathway that forests followed in this large and ecologically diverse region, but it is a common one, especially in wetter

and northern parts of the western hemlock potential vegetation type in cover types characterized by Douglas-fir and western hemlock (Winter et al. 2002a, 2002b), and a lack of fire between stand-replacement events. We characterize this model of succession further below and describe its variations and other successional pathways that can occur.

Early post-stand-replacement fire vegetation in the western hemlock–Douglas-fir forests of the western hemlock zone typically occurred as heterogeneous mosaics of grasses, herbs and shrubs, and hardwoods often with high levels of dead snags and down wood, and high species richness (Donato et al. 2011, Reilly and Spies 2015, Swanson et al. 2011) (fig. 3-3). Species compositional change, which can be rapid over the first 20 years as a function of the relative importance of invading and residual plant species groups, differs with time, the availability of propagules, disturbance characteristics, and properties of the environment (Halpern 1988, 1989). Standing dead tree structure and decay states are also dynamic within western conifer forests during the first decade or two following fire (Russell et al. 2006). Studies of post-wildfire conifer forests in the Western United States indicate that wildlife use of early-seral vegetation following fire and logging can change rapidly with time—since disturbance, with some species appearing in the first few years before disappearing later and others increasing in abundance as snag conditions and plant species composition changes (Saab et al. 2007, Smucker et al. 2005). Gashwiler

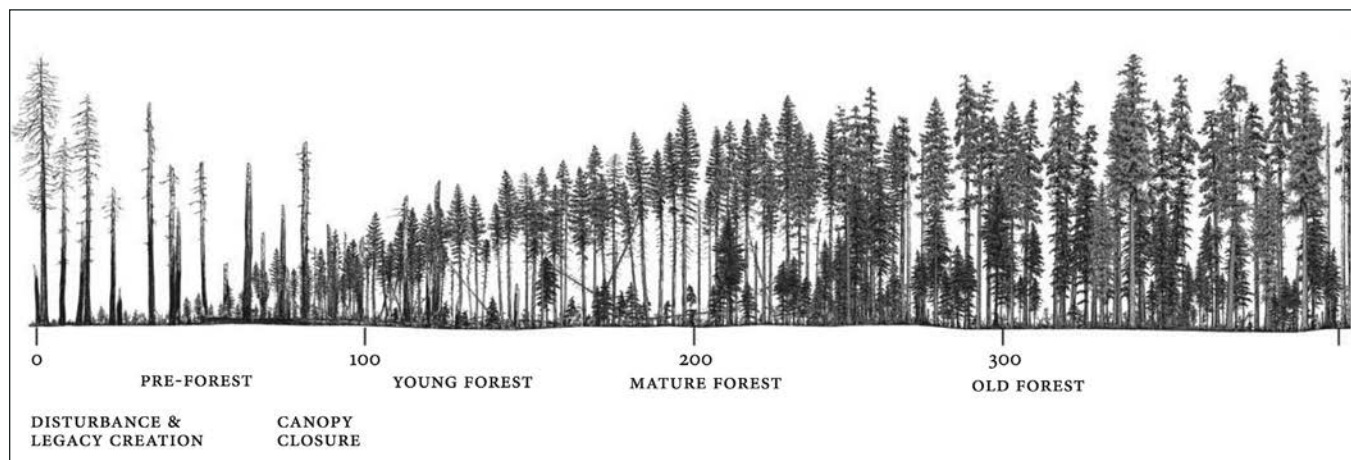


Figure 3-13—A common stand developmental pathway for a Douglas-fir and western hemlock forest following stand-replacement wildfire (from Franklin et al. 2018).

(1970) found that small mammal communities were quite dynamic in the first 10 years following clearcutting of an old-growth forest in the western Cascades of Oregon. The general pattern seems to be that while the “pre-forest” or early-seral stage can persist for many decades, the plant and animal communities are dynamic within that stage, and some species and communities are ephemeral.

Dead wood levels were especially high where prefire forests were late successional or old growth (Spies et al.

1988). Where fires burned early-successional and younger forest stand conditions, dead wood legacies were typically few and composed of smaller down logs (Nonaka et al. 2007, Spies et al. 1988). In contrast, where fires burned in forests containing large trees, levels of down wood were high, and individual pieces of large down wood may have persisted for several centuries while undergoing decomposition. Charcoal deposits from fires lasted in soil for up to one or more millennia (DeLuca and Aplet 2008).

Scientific and conservation interest in early-successional vegetation has increased in recent years as scientists learned about ecosystem responses to severe disturbance from studies of the eruption of Mount St. Helens (Dale et al. 2005) and high-severity wildfires that have occurred in the Western United States in recent decades (e.g., Donato et al. 2011; Hessburg et al. 1999a, 1999b; Hutto et al. 2016). Post-high-severity and mixed-severity disturbance ecosystems are generally understood to support unique biodiversity and ecosystem functions (Donato et al. 2011; Franklin et al. 2017; Hessburg et al. 2016; Swanson et al. 2011, 2014) relative to closed-canopy forests. This understanding is based largely on studies of clearcuts (e.g., Halpern 1988, Harr 1986) and volcanic eruptions (Dale et al. 2005) in the Northwest Forest Plan area, and few studies have been conducted in early-seral vegetation following wildfire or windstorms (e.g., Fontaine et al. 2009, Larson and Franklin 2005). Early-successional stages following natural disturbances are rich in biological legacies that include surviving organisms and organic matter such as dead trees. With tree canopies gone or greatly reduced, other life forms, including shrubs, grasses, and herbs often dominate the site, taking advantage of higher resource levels in light, water, and nutrients. These legacies clearly influence postdisturbance succession, stand development, and ecosystem function, though the variability in these relationships over time is not well understood. Variation in disturbance severity and predisturbance forest conditions has strong influence on legacy patterns, and subsequent forest succession that can persist for hundreds of years

(Donato et al. 2011, Dunn and Bailey 2016, Spies et al. 1988). In sum, early-seral stages are important when managing for conservation of native biodiversity and resilience in forested ecosystems and landscapes.

Given new scientific perspectives on early-seral vegetation, some have proposed that new terminology be used to describe it. For example, Franklin et al. 2018 suggest that early-seral vegetation be termed “pre-forest” because trees are not the dominant life form, although they are often present as seedlings. They also suggested that the term “early-seral forest,” which has been used to define this stage, is not correct because this stage is not forested and introduces a “tree-centric” bias to discussions about conservation and management (Franklin et al. 2018). Other terms that have been used to describe this stage include grass-forb, shrub-seedling, stand initiation, and cohort establishment. Terminology to describe successional stage, structural or developmental stage, or seral stage can be confusing and not interchangeable (Powell 2012). For example, some trees such as Douglas-fir and red alder are characterized as “early-seral” species (Franklin and Hemstrom 1981, Klinka et al. 1996), which can form early-seral stands or forests. The ambiguity of the terminology around postdisturbance changes in vegetation (including later successional stages) makes it important to define how terms are used (e.g., Powell 2012), and in the case of early-seral or pre-forest vegetation to clearly identify the ecological characteristics (life forms, species, structures) and functions (habitat, nutrient cycling, productivity) that reflect the underlying meaning and use of those terms.

The timing, composition, and structure (including cover thresholds) of tree canopy cover closure (e.g., canopy cover >70 percent (Yang et al. 2005) would have differed regionally by site conditions, disturbance characteristics, and seed source availability (Freund et al. 2014, Yang et al. 2005). Canopy closure may have occurred as early as 20 to 30 years following fire in moist productive sites, or where seed sources persisted in a canopy seed bank (Larson and Franklin 2005), but could have taken almost 100 years on other sites, after very large fires and with limited seed sources. These observations are based on studies of mature forests from the western Cascades (Freund et al. 2014). Tree establishment ended as the forest floor was covered by shrub and herbaceous vegetation, and tree canopies eventually closed (Freund et al. 2014, Tepley et al. 2014).

Not all stands or patches followed the same pathway to older forest structure. Multiple successional pathways would have occurred that varied in timing of composition and structural change over the first 100 to 200 years or longer (fig. 3-14) (Spies 2009). In riparian areas and moist coastal upland forests, shrubs and hardwood trees would often become established immediately after fire, limiting the establishment of conifer trees for many decades, and creating patches of hardwoods and shrubs with scattered conifers (Spies et al. 2002). Ultimately, those shorter lived hardwoods would die, leaving lower density conifer stands (or stands with variable-canopy dominance) with large dominant trees and well-developed crowns. For example, Spies and Franklin (1991) found that some 100-year-old stands of Douglas-fir and western hemlock that developed along with shrubs and hardwoods in the Oregon Coast Range had structural diversity that approached that of 400-year-old stands. Variability in seed sources, productivity, competition with shrubs and hardwoods, and partial stand replacement disturbances would have led to low-density relatively open younger forests where conifer canopy closure never occurs. These processes and pathways may actually be a faster route to complex older forest structure in some places than pathways that go through stages characterized by a higher density of conifers and conspecific competition (Donato et al. 2011, Tappiener et al. 1997).

Where closed-canopy forests developed, succession was driven by processes of growth, competition, understory development, maturation, and small- to moderate-size canopy disturbances from wind, insects, disease, fire, hydrologic, or geomorphic processes (Franklin et al. 2002). Somewhat arbitrarily, 80 years after conifer forest establishment has been used as the onset for “mature” (e.g., OGSF 80) Douglas-fir forests, and 150 to 200 years for the onset of multilayered old-growth forests (OGSI 200), depending on environment and disturbance history (Franklin et al. 2002, Spies and Franklin 1991). Eighty years was used as the threshold for late-successional/old growth in the NWFP (USDA FS 1994) because that is about the earliest time when such stands begin to resemble maturing forests in the moist forest (does not apply to the dry forest zone). Analyses of chronosequences indicate there is considerable variation in forest structure around these age breaks (Spies and Franklin 1991) (fig. 3-15) likely driven by multiple successional pathways, legacies, and time since disturbance. The stands (i.e., sample plots) in figure 3-1 would have followed individual development pathways, some pathways may be sigmoid shaped in the case of stands developing after a nonforest condition, other pathways may have been more U-shaped in the case of stands developing with significant live or dead legacies of the predisturbance old-growth forest (Spies and Franklin 1988).

The variability in structure with stand age indicates that at a regional scale, age or time alone is only a partial predictor of forest structure. The structural features of mature and old-growth forests would have included medium- to large-size (e.g., >40 inches) shade-intolerant tree species; smaller shade-tolerant trees of similar and lesser age in the mid to lower canopy layers; large standing and down dead tree boles; and horizontal and vertical structural heterogeneity of live and dead trees. Not all stands would have grown for centuries without stand-replacement fire—sometimes reburns within a few decades of a fire would occur consuming decayed dead wood and restarting succession (Donato et al. 2016, Gray and Franklin 1997, Nonaka 2003, Tepley et al. 2013).

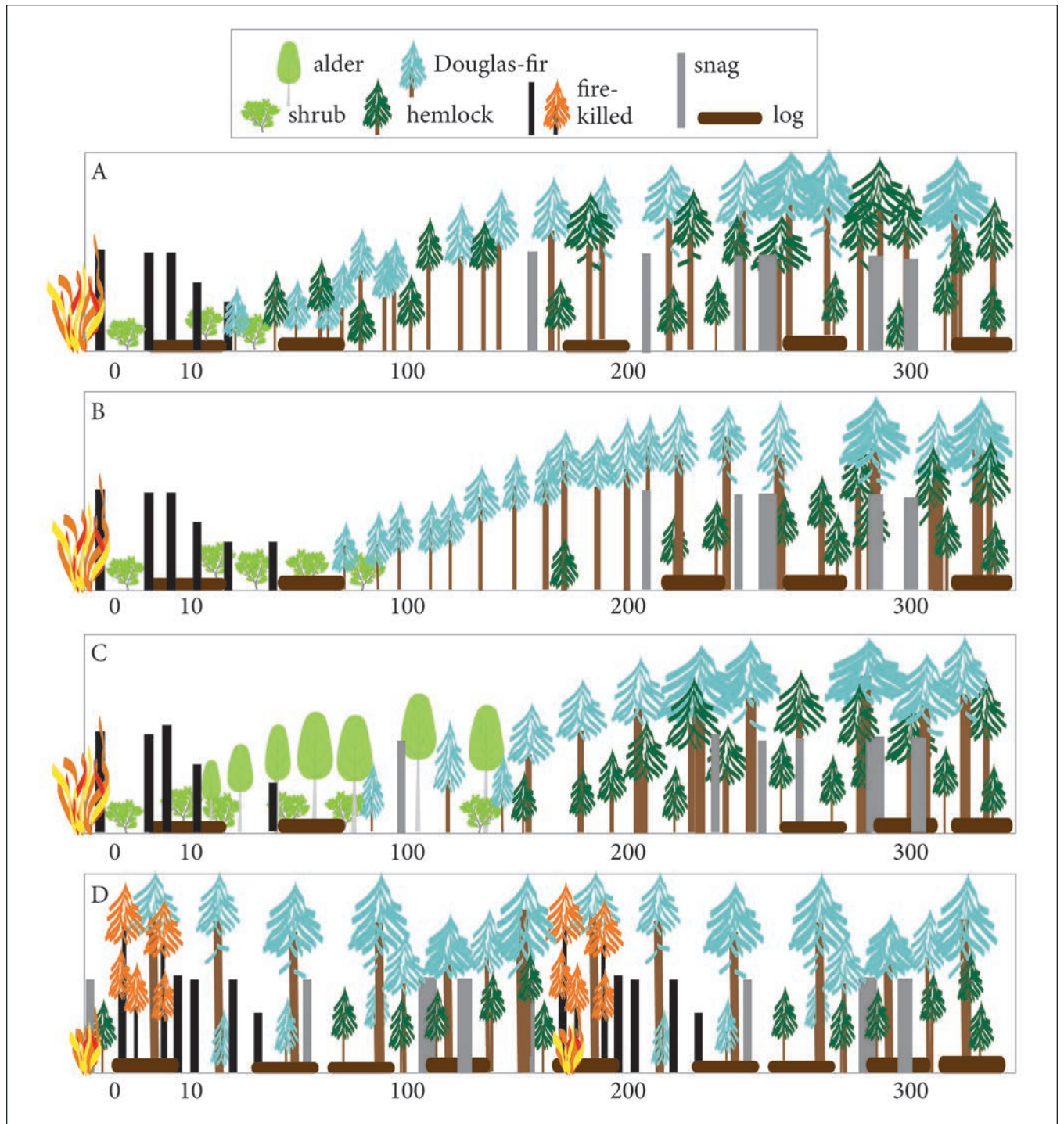


Figure 3-14—Multiple pathways of succession that could occur in the moist forests. Pathway A occurs when Douglas-fir canopy closure occurs within 50 years after a fire and western hemlock establishes early in succession. Pathway B occurs when the pre-forest shrub-dominated stage persists for many decades and hemlock is slow to establish. Pathway C occurs where shrubs and hardwood trees dominated early-successional development and reduced conifer densities so that conifer trees would not go through a self-thinning phase and large-diameter conifers and complex older forest structure would develop well before 200 years. Pathway D occurs where a partial stand-replacement fire occurs periodically in older forests and creates patches of dead trees, initiating new age cohorts of Douglas-fir or western hemlock trees beneath the surviving canopy and in openings created by the fire.

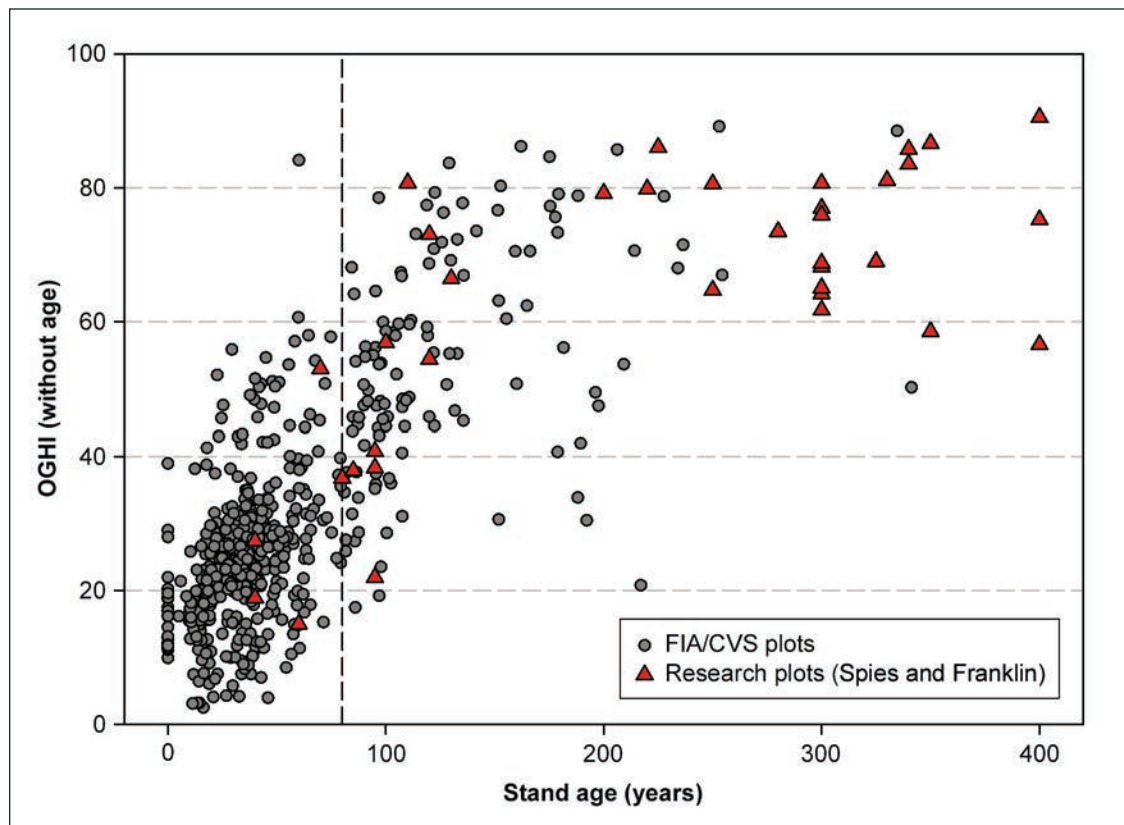


Figure 3-15—An old-growth forest habitat index (OGHI) (Franklin et al. 2005) in relation to stand age for forest inventory and research plots in the Oregon Coast Range. The index is based on number of large trees, large snags, volume of down woody debris, and tree size diversity, which is a surrogate for canopy layering. Age was not used to develop the index. The index is similar to the structure index used in Davis et al. 2015. FIA/CSV = Forest Inventory and Analysis/Continuous Vegetation Survey.

Successional and landscape dynamics in the drier, southern part of the western hemlock zone, where fire frequency was 50 to 200 years (fig. 3-4), would have included some of the same pathways as would have occurred in the infrequent fire regime, but with different frequencies of those pathways across landscapes. At the scale of large patches and small landscapes (e.g., 10^2 to 10^4 ac or ~40 to 4000 ha), these forests would have had more age, structural and compositional heterogeneity than equivalent areas for the moister parts of the region where an infrequent fire regime occurred (fig. 3-16). For example, reanalysis of data from Spies and Franklin (1991) from the old-growth forests in the southern western Cascades of Oregon indicated that stand ages (age of the oldest Douglas-firs in the stand) were younger (~270 years) and basal area, proportion of shade-tolerant trees, and density of large snags and volume of down wood were all much lower than in old-growth

stands in the northern Cascades of Oregon and the Cascades of Washington (400 to 500 years), after controlling for topography and aspect. Ares et al. (2012) found that snag densities in older forests in western Oregon also varied by aspect, with lower densities on south-facing slopes and in the foothills of the Cascades, where fire frequencies are higher than in the Coast Range. The mature and old-growth stages probably have more age classes of Douglas-fir than in the infrequent, high-severity regime forests as a result of more frequent partial stand-replacement fire (Dunn 2015, Tepley et al. 2013) (figs. 3-16 and 3-17). For example, Tepley et al. (2013) found that 85 percent of the older forest in their central western Oregon Cascades study area (primarily western hemlock potential vegetation type with some areas of Douglas-fir potential vegetation type) experienced non-stand-replacing wildfire during its centuries-long development (fig. 3-14D). These fires killed a portion of the

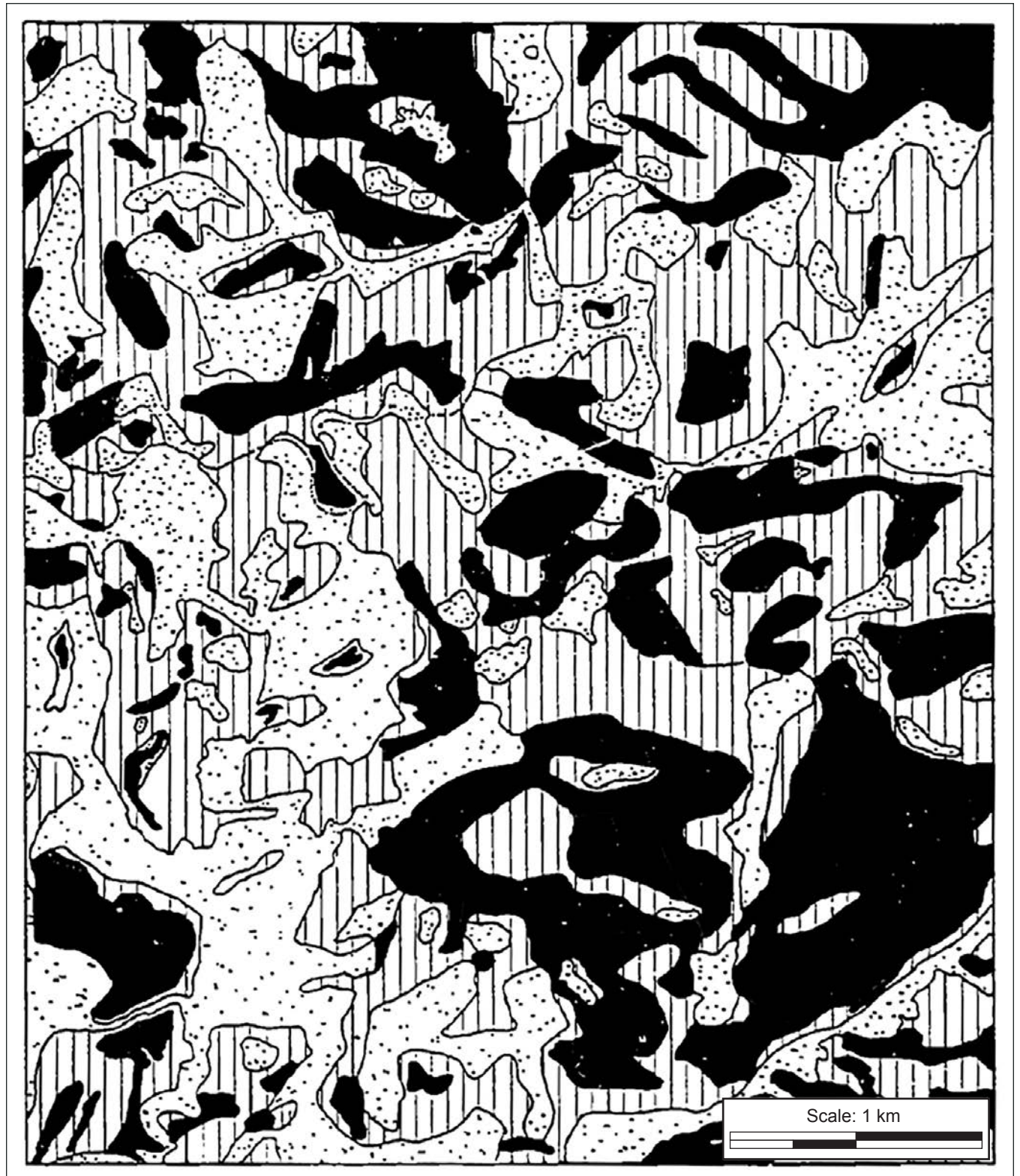


Figure 3-16—Mosaic of fire severity patches in a Douglas-fir and western hemlock landscape in the western Cascade Range of Oregon. Black = a high mortality area (>70 percent), vertical lines = moderate mortality (30 to 70 percent), and stippled = low mortality areas (<30 percent). From Morrison and Swanson 1990.

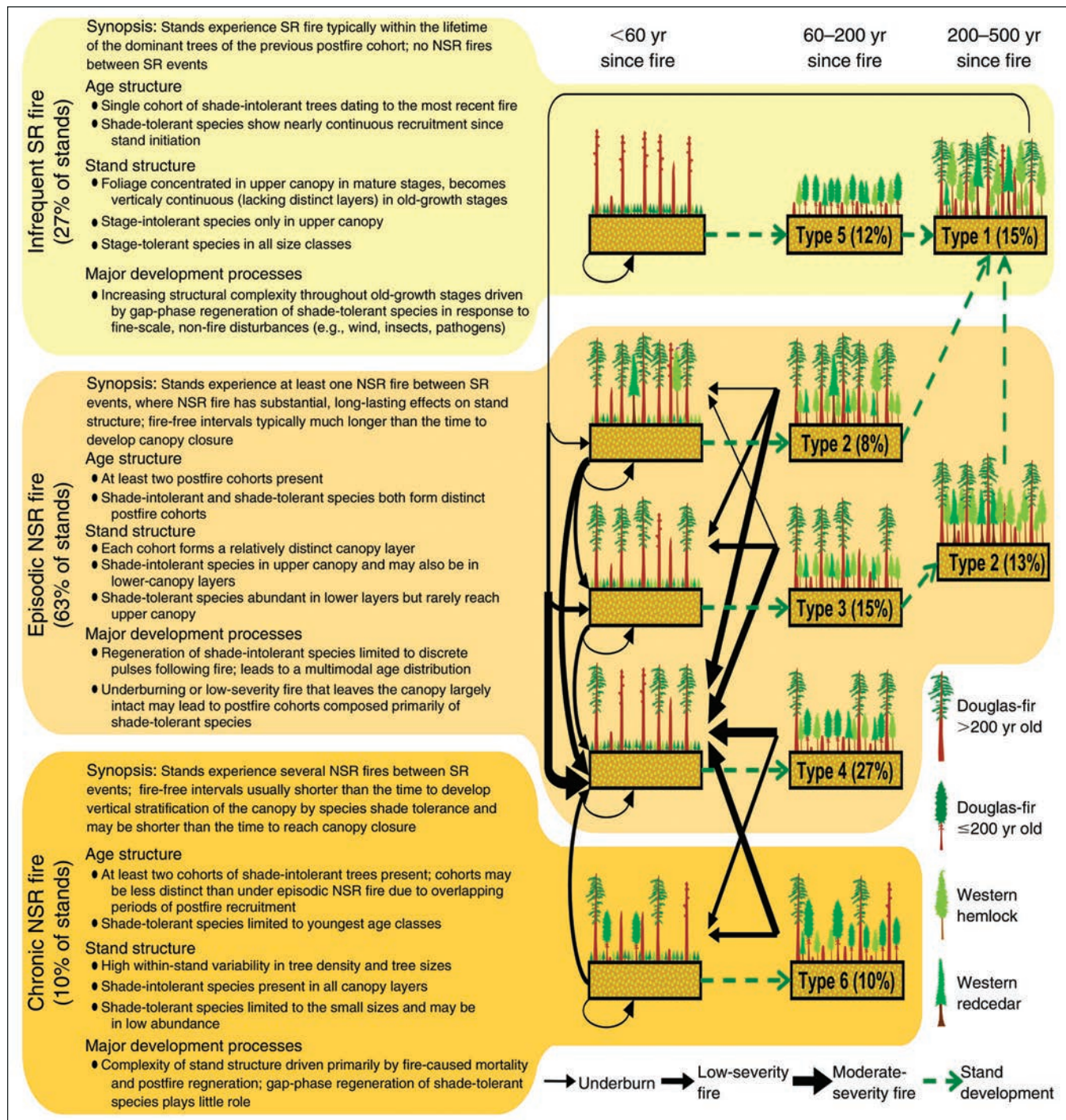


Figure 3-17—Conceptual model of stand-development pathways in Douglas-fir/western hemlock (current vegetation) forests in the moderately frequent, mixed-severity fire regime of the central western Cascade Range of Oregon. Dashed arrows represent stand development in the absence of fire, and solid arrows represent nonstand-replacing fire. Percentages indicate the percentage of the sample plots found in each structure type. SR = stand-replacing, NSR = non-stand-replacing. From Tepley et al. (2013).

overstory and established new cohorts of shade-tolerant or intolerant trees. Given the long time period that often occurred between fires, these landscapes of the infrequent and somewhat infrequent regimes would have typically been dominated by mature and old-growth forests.

Historical landscape dynamics—Many of the current old-growth stands of the wetter portions of the moist forests date to around 400 to 500 years ago (Spies 1991), a period with widespread fire (Tepley 2010, Weisberg and Swanson 2003) associated with positive phase of the Pacific Decadal Oscillation, which produced warmer conditions and drought. This warm period with many fires was followed by the Little Ice Age when cooler temperatures caused a reduction in both lightning- and human-ignited fires (Walsh et al. 2015) that may have allowed stands that established during the warm period to develop into older, multistoried forests. Empirical estimates of the amount or variation in old-growth forests or of any successional stage that occurred prior to Euro-American settlement are not available from any historical studies. Maps from the early 1900s can be used to approximate the amount of old forest present in the mid-20th century, suggesting that about 50 percent of all forest lands in this regime were covered by older forest (defined then in terms of large dominant and codominant trees), but that number varied widely across landscapes and watersheds (Davis et al. 2015). However, it is not clear how earlier mapping criteria related to current definitions of old growth, and by the 1930s, significant areas of older forest had already been lost to land clearing for settlement and agriculture, logging, and human-set wildfires.

Empirical studies of fire frequency and severity can be used with statistical models and other simplifying assumptions to estimate the age-class distributions that might have been present in a historical landscape (Agee 1993, van Wagner 1978). For example, Fahnestock and Agee (1983) used historical maps and statistical models to estimate fire cycles in western Washington. They found the proportion of large trees to be 0.6 in Douglas-fir, 0.82 in western hemlock, and 0.87 in mountain hemlock forest cover types. Spies and Turner (1999) estimated that on average, 61 percent of a given landscape would be old growth (>150 years since stand-replacing fire) if fire frequencies were 300 years. They assumed a constant climate and fire frequency,

equal flammability of successional stages, and high-severity fire—assumptions that are violated in real landscapes. For example, temperature and precipitation has varied considerably over the Holocene (past 11,700 years), including the past several thousand years when the current forest community assemblages developed (chapter 2). Susceptibility of successional stages often differ depending on fuel conditions and microclimate, and old forests can be less flammable than younger ones (Kitzberger et al. 2011).

Wimberly et al. (2000) used estimates of fire frequencies from lake cores in the Oregon Coast Range (Long et al. 1998) to estimate that fire rotation⁸ varied from about 150 to 300 years during the past 3,000 years. They then used a spatial landscape simulation model to estimate that the mean amount of old-growth (>200 years) and late-successional forests (>80 years) (including old growth) could have varied from 39 to 55 percent and 66 to 76 percent, respectively, during the 3,000 years prior to Euro-American settlement. The model indicated that the minimum and maximum amount (i.e., the historical range of variation [HRV]) of old-growth and late-successional forest in the Coast Range during this period was 24 to 73 percent and 49 to 91 percent, respectively. The range of variation was also a function of the scale of observation, with larger ranges for smaller areas, e.g., at the scale of a NWFP late-successional reserve (LSR) (~100,000 ac [~40 470 ha]) the range of late-successional forest would have been 0 to 100 percent. These analyses suggest that older forest conditions would have dominated forests of the region, but large areas of dynamic early-seral vegetation and younger forest would occur episodically as evidenced by the large blocks of old-growth forest that would have originated after fire. LANDFIRE⁹ (<https://www.landfire.gov/NationalProductDescriptions24.php>) estimated that the amount of “late

⁸ Fire rotation refers to the time required to burn an area equal to a defined landscape area (e.g., 1,000 ac [404.7 ha]). The entire area may not burn during this period; instead, some sites may burn several times and others not at all, but the summed area is equal to the defined area. Fire rotation = fire cycle.

⁹ LANDFIRE is an interagency geospatial data development program that used expert opinion to model historical amounts of vegetation stages for potential vegetation types based on published literature. The estimate of amounts of vegetation classes do not include historical ranges.

development” closed-canopy forest for the western hemlock zone was 70 percent, and the amount of open “early development” vegetation was 5 percent. Estimates of the HRV in successional stages are still needed for the NWFP area.

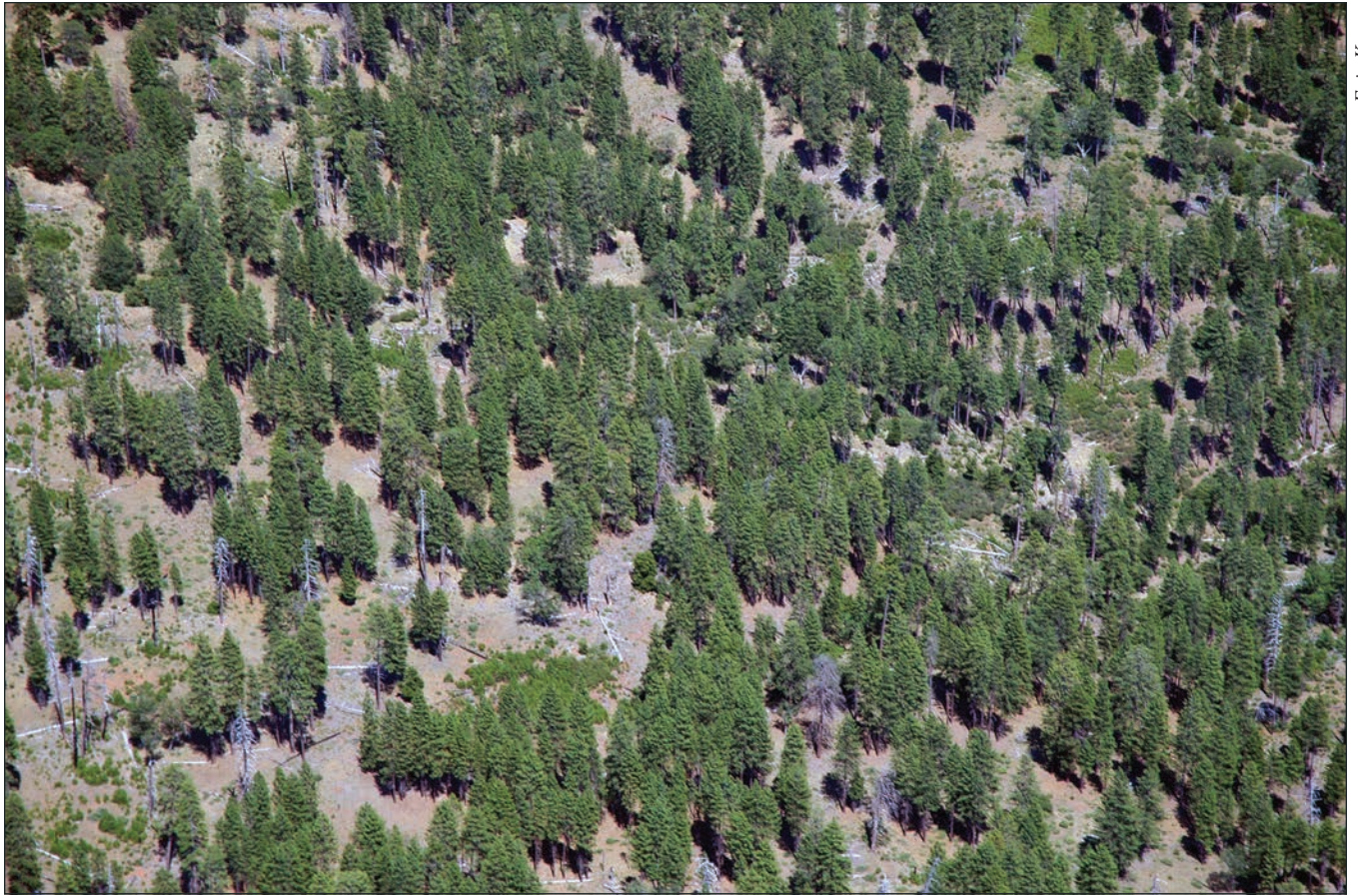
At the scale of regional landscapes or ecoregions, models suggest that early-successional patches occupied <20 percent of the area on average but may have reached as high as 30 percent over the span of several thousand years (Wimberly 2002). At the scale of LSRs, some watersheds may have been entirely composed of early-seral conditions after wildfires. Studies from Washington and southwest British Columbia (Dunwiddie 1986, Hallett et al. 2003) indicate that fire-return intervals were much longer in the northern part of this regime, so periods when early-successional conditions were abundant in these ecoregions were probably less than in the Oregon Coast Range. Moreover, the amount of fire and early-successional forest probably varied considerably over the past several thousand years in resonance with climatic variation.

The HRV in old-growth and other successional stages in the drier part of the western hemlock and other potential vegetation zones is less well known. It is also more difficult to estimate their abundance with statistical or simulation models given that many fires were non-stand replacing (Weisberg 2004) and resulted in multiaged patches and a large range of stand structures with a wide range of large live and dead tree densities, and tree species compositions (fig. 3-17). Estimates of historical amounts of old growth (i.e., areas of older trees with canopy layering) have been made from a few localities in the drier parts of the region. In the eastern part of the Oregon Coast Range, Wimberly (2002) estimated that the amount of this type of old growth over the 1,000 years prior to 1850 would have been less than 30 percent, where the fire-return interval was about 75 years, and many fires were non-stand-replacing (Impara 1997). The LANDFIRE estimates of these classes of historical amounts of “late” and early-development forest in drier parts of the western hemlock zone were 60 percent and 15 percent, respectively (<https://www.landfire.gov/NationalProductDescriptions24.php>). The amount of dense old growth without a history of non-stand-replacing wildfire, was prob-

ably less in these types, however, while the amount of other types with old trees would have been more common (Tepley et al. 2013) (fig. 3-17). The ecological functions and broader ecological significance of this diversity of old-growth forest conditions have not been studied, but Tepley et al. (2013) suggest this structural and composition diversity of older forests may have promoted resilience of large old-growth forest structures to disturbances and climate changes.

Dry forests—

As fire-return intervals decrease from over 200 years in the wetter forests to less than 25 years in the driest forests, the role of fire shifts from resetting succession and creating large patches of early-seral vegetation to regulating forest structure and dynamics altogether, and creating fine to mesoscale mosaics of different vegetation conditions, including early seral (fig. 3-18). At the shortest fire-return intervals, the simple model of succession and stand dynamics—i.e., a stand-replacement fire followed by long intervals of vegetation change without fire—no longer applies. In fact, the entire concept of succession and stand development toward multilayered old-forest structure in fire-dependent systems becomes problematic where fires are very frequent (O’Hara et al. 1996). A pathway of stand-replacement disturbance followed succession toward multilayered, closed old-growth forests still applies to some sites within the frequent, mixed-severity regime dry forests (Camp et al. 1997, Merschel et al. 2014), but not so much in the very frequent, low-severity regime where fire was more of an intrinsic ecological process than an external disturbance event. Forest structural stages (e.g., stem exclusion, old-forest multistrata, old-forest single stratum) can still be classified and identified in two dry forest fire regimes, but the structural conditions can be quite variable and complex, and pathways of change can be multidirectional owing to the interplay of fire severity, time since last disturbance, seed sources, and environmental heterogeneity (Reilly and Spies 2015). We discuss the two regimes separately below but recognize that for many landscapes and existing forest history studies, the two regimes may intermingle or have been lumped together.



Eric Knapp

Figure 3-18—Aerial photo of Beaver Creek Pinery showing spatial heterogeneity that can develop with frequent burning on a productive site in the southern Cascade Range of California.

Frequent, mixed-severity fire regimes—The potential vegetation types of the frequent, mixed-severity regime (15- to 50-year return interval) include Douglas-fir, grand fir, and white fir, and oak woodlands (fig. 3-4). The cover types of this regime include Douglas-fir, white fir, red/noble fir (*Abies procera*), and western white pine (*Pinus monticola*). Ponderosa pine can still be a component of some of these forests (Merschel et al. 2014). Forests of this type were characterized by multiaged cohorts of seral dominants and landscape mosaics created by medium to large patches of high-severity fire (fig. 3-12), but the landscapes were probably dominated by areas of moderate- to low-severity fire. In a Douglas-fir-dominated landscape of northern California, Taylor and Skinner (1998) found older stands with diverse age structure, but fire-return intervals were shorter (e.g., ~15 years), severities were lower, and

large severe fires were uncommon compared to Douglas-fir forests of the western Cascades of central Oregon. Many of “mixed-severity” areas of the drier eastern part of northern California have been mapped in our classification into the very high frequency, low-severity regime (fig. 3-6). Stands with the most diverse age structure in the Taylor and Skinner (1998) study experienced the greatest number of fires, whereas stands with fewer age cohorts had experienced fewer fires. Those with the most diverse age structure were those most closely exhibiting late-successional structure. However, in landscapes where fires were mostly low severity, the age-class/fire association was unclear (Taylor and Skinner 2003).

Mixed-severity regimes in dry forests would likely result in higher diversity of plant and animal communities and patch (area that differs from its surroundings)

heterogeneity compared to high-severity regimes or very frequent low-severity regimes (Hessburg et al. 2016, Perry et al. 2011). Areas of passive and active tree torching, mostly associated with clumps or groups of small understory trees with low limbs, would have created patches of tree mortality that would function as canopy gaps of various sizes in older forests. Subsequent fires, either by torching or girdling, would in turn thin these patches diminishing the even-aged group to a few individuals. Shade-intolerant tree regeneration would be more likely to establish in larger (e.g., >1 ac [0.04 ha]) high-severity patches. A prominent hardwood component was often associated with conditions emerging after mixed-severity fires. These hardwoods may play a pivotal role in continued mixed-severity fires (see discussion below).

The ecological importance of forests shaped by mixed-severity regimes (in both dry and moist forests) is widely recognized (DellaSala and Hanson 2015, Hessburg et al. 2016, Perry et al. 2011), but fine-scale studies that document how microclimate, wildlife, and fire respond to different expressions of vegetative heterogeneity, and different types of mixed-severity regimes have not been conducted. Our understanding of the mixed-severity regime in dry forests comes from patch- and landscape-scale reconstructions. That understanding is further complicated by lack of consistency in defining mixed-severity fire regimes across studies and lack of historical information about their spatial and temporal characteristics (app. 3). Several studies have characterized the spatial heterogeneity of patches dominated by this regime, especially for the eastern Cascades provinces (Hessburg et al. 1999a, 2000b, 2004, 2007; Perry et al. 2011).

The stand-development trajectories of high-severity patches could initially follow the pathway described by Franklin et al. (2002), but where shrubs or seed source limitations occurred, stand development might not proceed through the stem-exclusion closed-canopy stage. In addition, some elements of complex older forest structure (e.g., large-diameter trees and heterogeneous understories) might develop more rapidly than in the wetter forest types (Donato et al. 2011), which often have to develop following a relatively uniform and dense self-thinning phase.

The trajectory of development of a low-density tree patch can be altered if the area is severely burned again before trees are mature (Coppoletta et al. 2015, Lauvaux et al. 2016, Tepley et al. 2017).

Topography would have been an important driver of the mosaic pattern. Ridges and south-facing aspects with more frequent fire would tend to support more open-canopy stands of multicohort shade-intolerant and fire-tolerant trees, while valley bottoms, benches, and more northerly aspects with less frequent fire would have tended to support more complexly structured closed-canopy, multilayered stands of shade-tolerant and fire-intolerant trees (Agee 1998, Hessburg et al. 2016, Tepley et al. 2013).

For the eastern Cascades of Washington, Agee (2003) used historical fire-return intervals and simple mathematical models to estimate range of variation in forest structure classes. This region would contain both the frequent mixed-severity and very frequent low-severity regimes (fig. 3-4). The proportion of medium to large trees (>15 in [40 cm]) in dry to moist forest vegetation types (ponderosa pine, Douglas-fir, grand fir warm and cool mesic), regardless of canopy cover, ranged from 38 to 64 percent of the landscape. Agee (2003) found that late-successional forest (containing shade-tolerant tree species and multilayered canopies) was not present in ponderosa pine, warm-dry and cool dry Douglas-fir, or warm grand fir forest types, and present in about 10 to 16 percent in the “cool-mesic grand-fir” type. The amount of early-successional vegetation in these potential vegetation types in this region ranged from 6 to 15 percent (Agee 2003, Hessburg et al. 2000b). Hessburg et al. (2007) used aerial photography from the 1930s to 1940s to estimate that old, multistoried forests ranged from less than 5 percent to about 20 percent or more of dry coniferous forest watersheds, while the area of multistoried late-successional forest ranged from 17 to 68 percent in mixed-severity-regime forests. The estimates of forest conditions from this period would have been affected by logging, fire exclusion, and fires associated with Euro-American settlement around the turn of the century (e.g., the widespread fires of 1910), but Hessburg et al. (2017) used methods that reduced the impact of these anthropogenic effects.

Several historical studies have estimated pre-Euro-American settlement amounts of older forest and other successional stages for the eastern Cascades of Oregon (Andrews and Cowlin 1940 as cited in Davis et al. 2015; Baker 2015b; Hagmann et al. 2013, 2014; Kennedy and Wimberly 2009). The estimates of the percentage of forests of the eastern Oregon Cascades (across all lands in the ponderosa pine to moist mixed-conifer potential vegetation types) with large old trees are 35 percent (Kennedy and Wimberly 2009); 76 percent (Baker 2015b); 42 to 76 percent (Hagman et al. 2013); and 91 percent (Hagmann et al. 2014). LANDFIRE estimated that “late development” (both open and closed-canopy classes) covered 55 to 65 percent of the dry ponderosa pine and mixed-conifer forest environments that occur in the eastern Cascades of Oregon and Washington. Using empirical reconstructions from early 20th century aerial photos from this area, Hessburg et al. (1999a, 2000) showed that more than 40 percent of the eastern Oregon Cascades area contained patches with medium and large-size old trees in the overstory. They also noted that given logging in the ponderosa zone during the early 20th century, which they documented via photointerpretation, that amount may have been 50 percent larger, i.e., 60 percent of the area with medium- and large-size trees in the overstory. The much lower numbers from the Kennedy and Wimberly (2009) modeling study may be a result of the assumptions about the frequency and severity of fire in this region, which is not well-known given the lack of fire history studies that were available at that time (Baker 2015b). The estimates of historical older forest structure among these studies are not strictly comparable because of use of different definitions, geographies, potential vegetation types, disturbance regimes, and methods and data sources. It is especially difficult to compare different studies because of the environmental heterogeneity of the region, including strong precipitation and topographic gradients. Also, some moist mixed-conifer forests in the eastern Cascades of Washington have high fire frequencies (<25 years), which can be similar to that of drier ponderosa pine forests (Wright and Agee 2004); that relationship would mean that the moist mixed-conifer

potential vegetation type is not necessarily a good indicator of regimes with longer frequencies or higher fire severity. The frequent and very frequent fire regimes are spatially intermingled in many landscapes and are difficult to separate.

Most estimates of older forest described above are from landscape simulation studies and do not take into account canopy cover or forest density, with the exception of Hessburg et al. (2007), which is limited to the early and mid 20th century. The historical percentage of the eastern Cascades in denser older forest (e.g., areas that have not had fire for many decades, including areas that could potentially support northern spotted owls) has been estimated to be 9 percent (Kennedy and Wimberly 2009) and as much as 22 to 39 percent by Baker (2015b). Hagmann et al. (2014) estimated that areas of higher density forest (>185 trees per acre—“group 1”) and grand fir trees were historically rare in dry and moist mixed-conifer forests of the northern eastern Oregon Cascades, which would have included mixed- and low-severity fire regimes. Perry et al. (2004) also found relatively little grand-fir in the central Oregon Cascades.

The fire regimes and forest dynamics of frequent mixed-severity regime forests in California have been described by Taylor and Halpern (1991), Taylor (1993, 2000), Taylor and Solem (2001), Bekker and Taylor (2001, 2010), and Skinner (2003) and summarized by Skinner and Taylor (2006). Although no direct estimates of HRV have been made, these studies show that fire-return intervals tend to be at the low end of the range for this regime. The frequent mixed-severity fire regime is characteristic of the upper montane forests of red fir/noble fir, western white pine, mountain hemlock, and lodgepole pine. These forests are typified by precipitation being predominantly snow with snowpacks often lasting into early summer contributing to a relatively short, yet mostly dry, fire season (Skinner and Taylor 2006). Higher productivity (e.g., more fuels) and greater sensitivity of the species to fire compared to the very frequent, low-severity fire regime may help drive occurrence of moderately large patches (hundreds to thousands of acres) of high-severity fire despite the high frequency of fire.

Very frequent, low-severity fire regimes—The very frequent fire (<25-year interval), low-severity regime dry forests often occur in association with the forests of the infrequent, low-severity regime especially in the eastern Cascades and Klamath provinces in areas of topographic variability and strong climatic gradients (fig. 3-4). This fire regime would have been common in ponderosa pine, dry to moist mixed-conifer and oak woodlands vegetation types. The successional dynamics, structure, and composition of low-severity regime forests can be simplified into two pathways that lead to very different major types of old growth (Stine et al. 2014). In the first, a dominant low- or mixed-severity fire-dependent pathway maintained old-growth conditions (primarily old live and dead trees) in a shifting mosaic of open and moderately closed canopy patches (e.g., 20 to 60 percent canopy cover) (figs. 3-18 and 3-19).

A second, historically much less common pathway occurred where local climate and topoedaphic circumstances (e.g., rocky ridges) reduced wildfire frequency and led to development of patches of denser (60 to 90 percent canopy cover), multistory old-growth with shade-tolerant species (Agee 1993; Camp et al. 1997; Hessburg et al. 1999a,

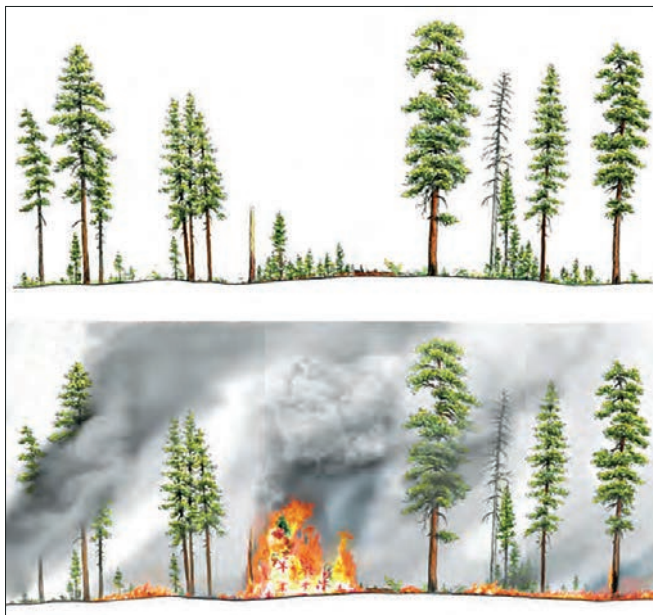


Figure 3-19—Hypothetical structural profile and typical historical fire behavior in a ponderosa pine forest of the eastern Cascade Range of Washington. From Van Pelt (2008).

1999b, 2000, 2007; Merschel et al. 2014; Sensenig et al. 2013). Levels of large standing and dead down wood would be much lower than in old-growth forest types in the other fire regimes (see Youngblood et al. 2004 for density estimates), owing to lower densities of large trees and frequent consumption of down wood (Safford and Stevens 2016, Skinner 2002). Despite the lower densities relative to denser old growth, large standing dead trees would have been present throughout though they would have been patchy and not found on every acre (Stephens and Fulé 2005). The pattern of seral stages within the forest matrix would be a fine-meso-scale mosaic of patches (<1 ac [<0.40 ha] to thousands of acres). The dominant pathway was maintained by high- to moderate-frequency, low- to mixed-severity fire (Baker 2012, Hessburg et al. 2007); scattered small- to medium-size patches with canopy tree mortality (individuals or small- to medium-size clumps) would have been present with medium and large fire-tolerant trees occurring in low to locally moderate densities (Churchill et al. 2013, Larson and Churchill 2012). For old-growth ponderosa pine in Oregon and California, canopy trees were not uniformly distributed and tended to occur in either clumps of up to 80 ft (24 m) in diameter (Youngblood et al. 2004) (figs. 3-17 and 3-18). These forests are sometimes characterized as being open, low-density forests, “park-like” stands (Agee 1993, Hessburg et al. 2015, Sensenig et al. 2013, Youngblood et al. 2004) (fig. 3-1). Bark beetles, which attack trees in small groups, may have interacted with fire in these forests to promote patchy regeneration of ponderosa pine. This would occur where beetle-killed patches of dead trees had accumulations of small branches and coarse woody debris that burned with high severity, killing rhizomatous grasses and promoting patchy regeneration of ponderosa pine regeneration in ash of the burned logs and sterilized mineral soil (Agee 1993).

The second successional pathway would lead to denser patches of pine and Douglas-fir or true fir regeneration, as mentioned above, often associated with variation in topography (steeper slopes and higher elevation), microclimate, and fire frequency that allowed trees to develop on moister microsites associated with north-facing lower slopes, concave areas, riparian areas, and wetter soils (Camp et al. 1997, Merschel et al. 2014). However, Baker (2012) did not

find that concentrations of fir were associated with aspect or topography in an analysis of General Land Office (GLO) survey data from the eastern Oregon Cascades. Following low- to moderate-severity fire on these more moist sites, white fir or grand fir could establish in the understory and occasionally reach the canopy where bole diameters and bark thickness was sufficient to withstand surface fires. On some productive sites (e.g., benches), old-growth grand-fir or white-fir patches developed even while experiencing frequent surface fires that burned in from adjacent drier ponderosa pine and grassland sites (Hessburg et al. 1999, Taylor and Skinner 2003). The relative amount of open and denser older forests may have varied over time with climate. Many studies across the area support this characterization of forest structure and dynamics for this type in some portions of the region (Bisson et al. 2003; Hann et al. 1997; Hessburg et al. 1999a, 1999b, 1999c, 2000, 2003,

2005; Keane et al. 2002, 2009; Lehmkuhl et al. 1994). With fire exclusion, the dense late-successional and old-growth pathway (either with ponderosa pine, Douglas-fir, or *Abies* spp.) has become dominant (fig. 3-20). White fir and grand fir have widely expanded out of their historical environments and fire refugia into sites that were historically dominated by ponderosa pine (or sugar pine (*Pinus lambertiana*) in California) or pine mixes with Douglas-fir (Camp et al. 1997; Hagsmann et al. 2017; Merschel et al. 2014; Taylor and Skinner 1998, 2003), or grassy woodlands often originally dominated by hardwoods (Skinner et al., in press). This expansion of shade-tolerant trees (which is discussed more below) has been widespread across a range of topographic settings and forest types, including drier mixed-conifer and ponderosa pine types (Hagsmann et al. 2014; Hessburg et al. 1999a, 1999b, 2000a, 2003, 2005, 2015, 2016; Merschel et al. 2014; Stine et al. 2014).



Thomas Spies

Figure 3-20—Old-growth ponderosa pine in the eastern Cascade Range of Oregon with understory of grand fir that established in the early 1900s after fire exclusion.

Woodlands, shrublands, and grasslands—A significant portion of some of the dry forest landscapes was occupied by patches of semistable, woodlands, shrublands, and grasslands (Hessburg et al. 2007) (figs. 3-21 and 3-22). These included oak, juniper, and pine woodlands that did not succeed to denser forest as a result of climate, soils, and frequent fire (Agee 1993, Franklin and Dyrness 1973, Hessburg and Agee 2003, Skinner et al. 2006). In many cases, a frequent grass- or shrub-driven fire cycle was responsible for maintaining low tree cover (Hessburg et al. 2016). These areas were so dominated by grasses over a geologically long timeframe that mollisols can be seen

today as the characteristic soil type. Open stands and oak dominance were maintained by American Indians in many areas using fire to promote desired resources associated with such habitats (Anderson 2005, Skinner et al. 2006) (chapter 11). Figures 3-21 and 3-22 illustrate these landscapes, and although large fires in the early 1900s would have affected these patterns, many of the large fires would have occurred in grasslands and shrublands (that were historically maintained by frequent fire) as evidenced by the lack of snags and dead trees in the large nonforest patches in these photos. Interestingly, the concept of old growth (in a general sense of a vegetation type that persisted for very



Figure 3-21—Photographs of the Mission Peak area on the Okanogan-Wenatchee National Forest in 1934 and 2010. The 1934 image illustrates the mosaic of closed forests, open forests, woodlands, and grasslands that would have characterized many landscapes with low- and mixed-severity fire regimes. Open areas typically lack snags that would be indicative of recent high-severity fire in forests. Landscapes in 1934 may have been influenced by settlement fires, logging, and fire exclusion.

long periods under natural processes) has also recently been applied to these nonforest vegetation types (Veldman et al. 2015) because they have distinct conservation values that arise as a result of being “ancient”¹⁰ ecosystems with characteristic biotic and soil properties that have been lost owing to changes in fire regimes, grazing, and other land use changes.

¹⁰ Grasslands have existed for millions of years, and some grasslands may take 100 to as much as 1,000 years to develop; and clonal grasses can live for over 500 years.

Oak woodlands dominated by California black oak (*Quercus kelloggii*) and Oregon white oak (*Q. garryana*) and other hardwoods were maintained in an open old-growth state by very frequent low-severity fire (Agee 1993, Cocking et al. 2012, Franklin and Dyrness 1973). These species can form large, old trees with high value because they produce mast or berries, as well as large cavities for wildlife. They often support a high diversity of understory plants, fungi, and associated wildlife of particular importance to tribes (see chapter 11). However, a lack of fire in many of these areas has permitted conifer



Figure 3-22—View from Eddy Gulch Lookout in the Salmon River watershed of the Klamath National Forest in 1935 (top) and 1992. The 1935 image illustrates the mosaic of closed forests, open forests, shrub fields, woodlands, and grasslands that would have characterized many landscapes with low- and mixed-severity fire regimes. Open areas typically lack snags that would be indicative of high-severity fire in forests. Landscapes in 1935 may have been influenced by settlement fires, logging, and fire exclusion.

trees such as Douglas-fir to increase shade, accumulate conifer litter, and form ladder fuels, which consequently, render mature hardwoods more vulnerable to top-kill from fires. These trends are particularly evident in riparian forests of southwestern Oregon, where the shift in fire regime has led to reductions in both hardwoods and large trees (Messier et al. 2012).

Role of shrubs and hardwoods in Klamath-Siskiyou forest dynamics—The successional dynamics of low- and mixed-severity regime forests in the Klamath-Siskiyou region of Oregon and California are distinctive for the prominent role of shrubs and hardwoods in the vegetation community and their interaction with both fire and forest succession. In the northern and western part of this region, mixed-severity fire can lead to patchy old growth with tanoak (*Notholithocarpus densiflorus*) understories (as small trees) intermixed with Douglas-fir that either survives the lower intensity fire as a large tree or regenerates in patches of high-severity fire that kill the tanoak (Agee 1993). In other areas of this region, and extending into the southern Cascades and northern Sierra Nevada, dense stands of the shrub form of tanoak (*N. densiflorus* var. *echinoides*) can be found. These stands often do not burn well under less-than-severe conditions but will strongly sprout following severe fires even though the acorns are killed by fire.

Throughout the Klamath-Siskiyou region, shrub species resprout after fire and are also stimulated to germinate from seeds stored for long periods in soil seed banks following fires (Knapp et al. 2012b) with areas of higher severity fire leading to greater density of shrubs (Crotteau et al. 2013). Hardwoods (especially oaks, tanoak, and madrone (*Arbutus menziesii*) mixed in with the often more dominant conifers are often able to resprout following high-severity fires that kill the conifers (Cocking et al. 2012, 2014; Skinner et al. 2006). This adaptation facilitates the reestablishment of trees in severely burned forest areas at an early-seral stage. For conifer forests to again occupy these areas requires sufficient time between severe burns to allow conifer trees to reestablish and mature. Where severely burned areas are reburned before such

conditions are achieved, shrubfields and hardwoods are likely to be maintained and can become a more permanent part of the landscape (Cocking et al. 2014, Coppoletta et al. 2015, Lauvaux et al. 2016). Several recent studies have documented how severely burned areas that are reburned within a few decades are likely to again burn severely (Coppoletta et al. 2015; Odion et al. 2004; Perry et al. 2011; Thompson and Spies 2010; Thompson et al. 2007, 2011). In other cases, hardwoods in mixed-wood forests may play an important role in protecting some of the coniferous forest cover from severe fire effects via their foliar moisture content (Agee 2002, Perry 1988, Perry et al. 2011, Raymond and Peterson 2005, Skinner 2006, Skinner and Chang 1996). Likewise, depending upon the forest community type, hardwood trees and shrubs may in fact facilitate conifer succession via mycorrhizal fungi shared by both hardwood and coniferous species (Horton et al. 1999).

In complex topography, such as that found in the Klamath-Siskiyou area, it is unlikely that disturbance regimes and seral stages randomly moved about the landscape. Rather, particular parts of the landscape were more prone to severe burns. Upper thirds of slopes, and especially south- and west-facing slopes, were prone to repeated severe burning that perpetuated shrub dominance (Jimerson and Jones 2003, Taylor and Skinner 1998, Weatherspoon and Skinner 1995). Shrubfields may be places where forests burned severely or places where fires have long maintained shrubfields (Baker 2012, 2014; Lauvaux et al. 2016; Nagel and Taylor 2005). In the latter case, these were not places that periodically contributed large wood and snags but reburns of shrubs, grasses, and occasional small conifers.

Alternative views of disturbance regimes of the dry forests—Some have argued that most ponderosa pine and mixed-conifer forests in the Western United States, including the area of the NWFP that we define as having had a very frequent, low-severity regime, have been mischaracterized. They contend that these forests are better characterized instead as having a more variable-severity fire regime, with significant components of mixed and

high-severity fire as well (Baker 2012, Odion et al. 2014, Williams and Baker 2012). Hessburg et al. (2007) has also been cited in support of this argument (Baker 2012); however, the results of Hessburg et al. (2007) do not fully support the claims of Baker (2012); there are some key differences. The classification of high-severity fire from aerial photos in Hessburg et al. (2007) included areas with small trees, grasslands, shrublands, and sparse woodlands. These nonforest areas would have typically burned with high-severity given the low stature of their vegetation driven by a predominantly grass-fire cycle. When Hessburg et al. (2007) restricted their analysis to forest cover types, they found that less than 20 percent of each cover type was consistently affected by high-severity fires (fig. 3-23). For example, the dominating ponderosa pine and Douglas-fir cover types exhibited 13 and 18 percent high-severity fires across the study area, respectively. Similarly, when they restricted their analyses to forest structural classes (fig.

3-24), they found that no structural class experienced more than 17 percent high-severity fire across the study area. Furthermore, Baker (2012) uses Hessburg et al. 2007 to support his claim that “substantial” areas of high-severity fire occurred in ponderosa pine and dry mixed-conifer, but he cites Hessburg et al. (2007) data from Ecological Subregion 5 (ESR5), which is not a dry forest environment, but is classified as “moist and cold forest” type, with lesser amounts of dry forests. Hessburg et al. (2007) found considerable evidence of high-severity fire in their regional analysis of dry pine and mixed-conifer forest landscapes, but much of the high-severity fire was associated with grasslands and shrublands that were common in these landscapes in the past and were intermingled with forested patches. These vegetation types would typically burn with high severity. Figure 3-23 shows the proportion of forest structural classes affected by low-, mixed-, and high-severity fire in three ecoregions.

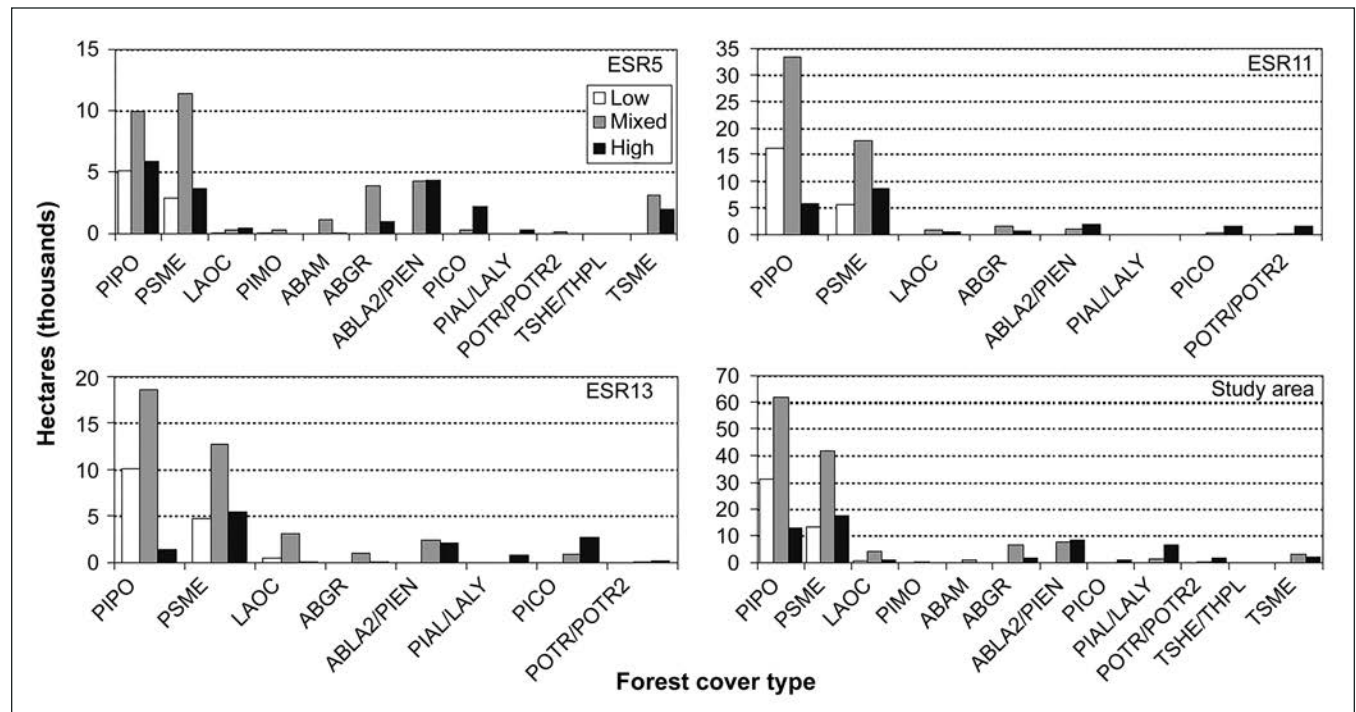


Figure 3-23—The proportions of premanagement-era total forest area (hectares) by forest cover type in low-, mixed-, and high-severity fire (corresponding with percentage of canopy mortality values of ≤ 20 percent, 20.1 to 69.9 percent, and ≥ 70 percent, respectively) of Ecological Subregions (ESRs) 5, 11, and 13. Cover type abbreviations are TSHE/THPL = western hemlock/western redcedar; PIMO = western white pine; POTR/POTR2 = *Populus* and *Salix* spp.; LAOC = western larch; TSME = mountain hemlock; PIAL/LALY = whitebark pine/subalpine larch; ABAM = Pacific silver fir; ABGR = grand fir; PICO = lodgepole pine; ABLA2/PIEN = subalpine fir/Engelmann spruce; PSME = Douglas-fir; PIPO = ponderosa pine. From Hessburg et al. (2007).

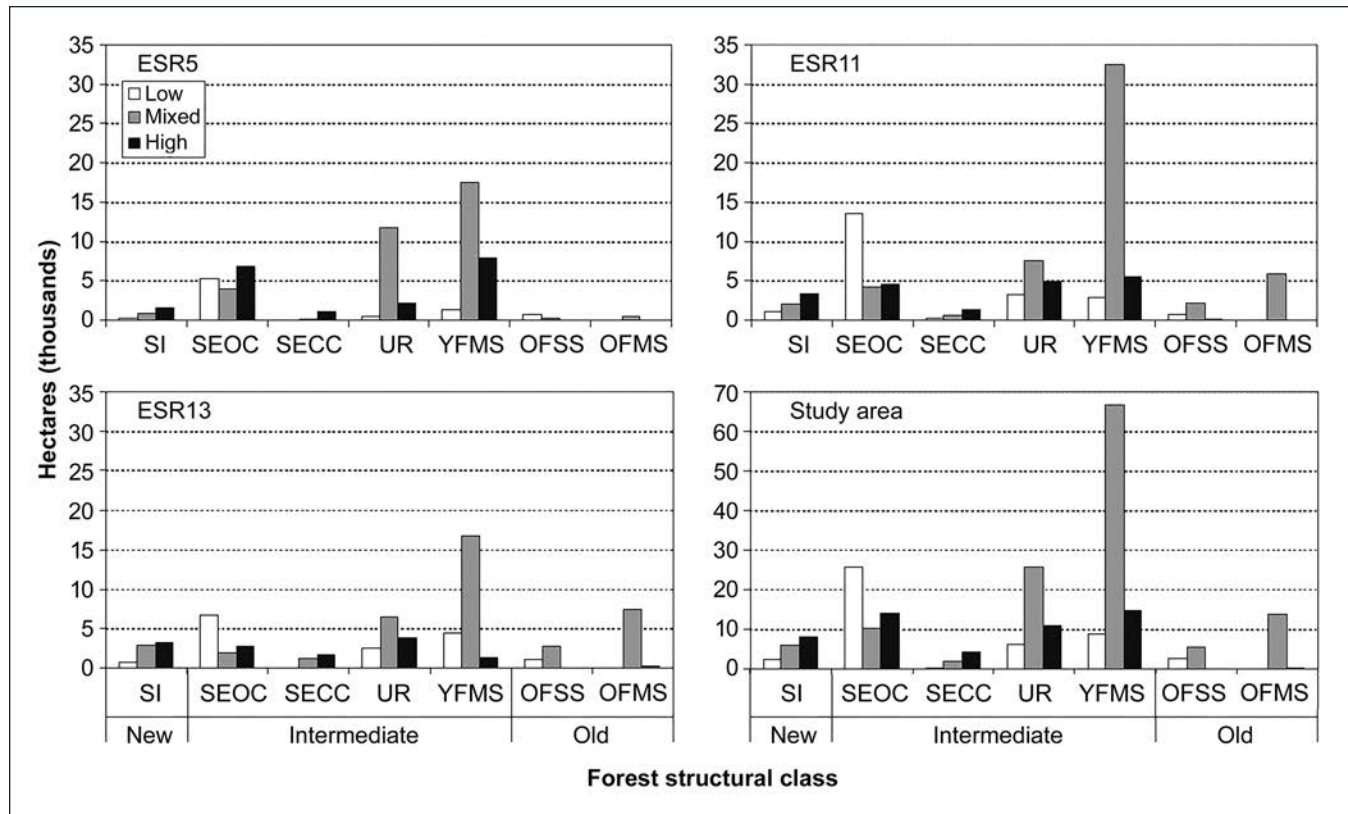


Figure 3-24—The proportions of the premanagement-era dry forest area (hectares) by forest structural class in low-, mixed-, and high-severity fire (corresponding with percentage of canopy mortality values of ≤ 20 percent, 20.1 to 69.9 percent, and ≥ 70 percent, respectively) of Ecological Subregions (ESRs) 5, 11, and 13. Structural class abbreviations are: SI = stand initiation, SEOC = open canopy stem exclusion, SECC = closed-canopy stem exclusion, UR = understory reinitiation, YFMS = young multistory forest, OFMS = old multistory forest, OFSS = old single-story forest. New, intermediate, and old designations are used to group structural classes into broad age groups. From Hessburg et al. (2007).

Williams and Baker (2012) and Baker (2012) use GLO survey data from the 1880s and 1890s on live tree sizes and species to infer historical stand densities and fire regimes from central Oregon. The evidence and methods used to support the claims that the historical role of high-severity fire in low-severity regimes has been underestimated has been the subject of several published critiques and counter arguments by both sides of the debate. In one critique, Fulé et al. (2013) point out three problems with using GLO survey data to infer disturbance history (e.g., Baker 2012): (1) the use of tree size distributions to reconstruct past fire severity and extent is not supported by empirical age-size relationships nor by local disturbance history studies; (2) the fire-severity classification based on the survey data is qualitatively and quantitatively different from most modern classification schemes, limiting the validity of comparisons

to history; (3) their finding of "surprising" heterogeneity within these stands does not actually differ substantially from other previous studies (some from ponderosa pine forests outside the NWFP area but still potentially relevant to dry forests in the NWFP area) that found areas and clumps of relatively high density in ponderosa pine and mixed-conifer forests (e.g., Brown and Cook 2006, Youngblood et al. 2004) (fig. 3-25). For example, the lower left corner (66 by 66 ft [20 by 20 m]) of the old-growth plot that Youngblood analyzed had 16 trees (equivalent to a density of upper canopy trees of about 160 trees per acre), while the upper right corner had one tree (an acre-scale density of 10 trees per acre).

Williams and Baker (2014) responded to that critique of Fulé et al. (2013) by arguing that the concerns are unfounded and based on misquoting their 2012 paper.

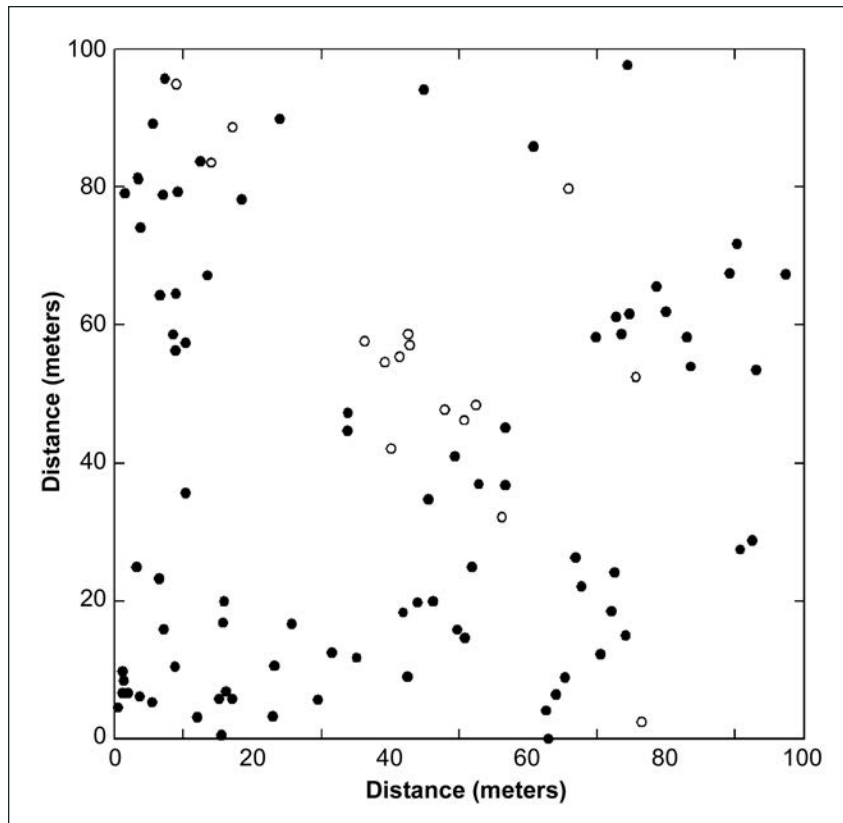


Figure 3-25—Spatial patterns of live (filled circle) and dead trees (open circle) in the upper canopy of an old-growth ponderosa pine forest in central Oregon. From Youngblood et al. (2004).

Williams and Baker (2012) used tree density and relative proportions of small and large trees to classify GLO data areas as either low- or high-severity fire. According to Baker (2012), 26 percent of pine and dry mixed-conifer forests in the eastern Oregon Cascades showed evidence of high-severity fire based in part on tree density. The findings of Baker (2012) depend on many assumptions, the most important being that the method for calculating tree density from GLO survey data (Williams and Baker 2011) produces an unbiased estimate. However, a recent paper by Levine et al. (2017) indicates that the method (Williams and Baker 2011) used by Baker (2012) overestimates tree density by a factor of 1.2 to 3.8. This finding could help explain why the estimates of historical tree densities that Baker has reported (mean of 100 trees per acre) are considerably higher than those reported from other studies, e.g., 62 trees per acre (Munger 1917) or 26 to 32 trees per acre (Hagmann et

al. 2013, 2014). Other assumptions made by Baker (2012) could explain the higher densities relative to other studies including the assumption that his survey points represent dry environments and not wetter mixed-conifer sites that often occur in the eastern Cascades where topographic and precipitation gradients are strong, and produce high variability in forest structure, composition, and dynamics (Merschel et al. 2014).

Odion et al. (2014) have also argued for the occurrence of more high-severity fire in ponderosa pine and mixed-conifer forests of western North America using inferences from analysis of current tree-age data from unmanaged areas collected through the U.S. Forest Service Inventory and Analysis (FIA) program (Odion et al. 2014). Age data were analyzed and it was assumed that if stand-age diversity was low, then fire effects represented low- or mixed-severity regimes; if stand-age diversity was high, then the

forest came from a mixed-severity regime with significant areas of high severity. However, a critique by FIA and other scientists argues that the assumptions, analysis and conclusions of this paper are invalid (Stevens et al. 2016). First, the FIA stand-age estimator underestimates the age range of trees in plots, and it routinely undersamples old trees, which would be relatively common in forests subject to low-severity fire regimes (see Merschel et al. 2014). Forests with a low-severity fire regime also continuously recruit new cohorts of regeneration, which would be poorly reflected in the stand-age estimator. Second, recruitment events are not necessarily related to high-severity fire occurrence as we have described above. Odion et al. (2016) responded to Stevens et al. (2016) and identified areas of “agreement and disagreement.” Areas of agreement include high-severity fire was a component of forests in low-severity fire regimes, that tree recruitment occurs in the absence of fire, and FIA stand data may provide evidence of past high-severity fire. Areas of continued disagreement according to Odion et al. (2016) include deciding what threshold to use for mortality from high-severity fire, plot sizes needed to detect high-severity fire, use of diameter-age relationships for reconstructing basal area, and historical data sources that document high-severity fire in patches larger than 2,500 ac (~1000 ha). We disagree that their historical sources present many examples that document the occurrence of large patches of high-severity fire in forests with low-severity regimes. Historical maps we found from the early 1900s document three patches of high-severity fire larger than 2,500 ac (~1000 ha) in Oregon and Washington that account for 1 percent of the area of this regime (fig. 3-6). In addition, the so called large patch of high-severity fire in the “eastern Cascades” of Oregon that is cited in Dellasala and Hanson (2015: 30–31) from mapping of Leiberg (1903) as evidence of a 35,000-ac (~14 200-ha) patch of high-severity fire in ponderosa pine forests actually comes from a township in the western Cascades in an area of mixed-conifer forest, containing red fir and noble fir. This township and the boundaries of this fire straddle the infrequent high-severity regime and moderately frequent to somewhat infrequent mixed-severity regimes of our regime map (fig. 3-6).

These concerns about interpretation of forest history data notwithstanding, there is essentially no disagreement that very frequent, low-severity regime forests (e.g., ponderosa pine and mixed conifer) included occasional small- to medium-size (e.g., tens to hundreds of acres) patches of high-severity fire. In addition, the broader landscapes would have contained grasslands or shrublands maintained by high-severity fire (relative to that life form) (e.g., see Hessburg et al. 2007, Perry et al. 2011). Given that many larger landscapes (including forested areas and nonforest areas) are often a mosaic of environments that support both low- and high-severity fires, it would not be surprising to find landscapes where the amount of high-severity fire to forest and nonforest vegetation exceeded 20 percent (e.g., see historical landscape data in Hessburg et al. 1999a, 2000, 2007). However, over smaller areas or areas with less topographic variability and within environments that predominantly supported forests, the amount of high-severity fire in low-severity regime forests would be expected to be lower than 20 percent. For example, Hagmann et al. (2014) found that only 9 percent of forest survey transects in 123,500 ac (~50 000 ha) of mixed-conifer landscape in eastern Oregon showed potential evidence of high-severity fire based on absence of large trees.

In summary, we believe the preponderance of evidence supports the view that large patches of high-severity fire were not a major component of dry forests with very high frequency, low-severity forest fire regimes. However, they were an important component of the frequent, mixed-severity regime. Remember that these regimes exist along a continuum of environments that differ across regions and landscapes. This means that landscapes often do not fit neatly into one regime or another. These alternative views of the role of high-severity fire in low-severity fire regimes highlights that generalizations either for or against management interventions across a wide range of forest types and environments should be made with caution. Different definitions of severity, scales of observation, and types of evidence (e.g., maps, surveys, aerial photos, tree age and size distributions, etc.) make it difficult to compare across studies because these factors influence the scope of inferences that can be made. In addition, subregional and landscape-scale variation in ecosystems and interactions among climate,

topography, soils, vegetation, and disturbance agents make it difficult to accurately extrapolate over to large areas. Efforts to infer process (e.g., disturbance history) from pattern (e.g., ages, sizes, or densities of trees, and patches of trees in maps and aerial photos), as is done in many of the fire history studies we cite, can also be fraught with some degree of uncertainty because similar patterns in biotic communities can arise from different processes (Cale et al. 1989). For example, much of the open forest reported by Baker could have been made up of aggrading meadows and shrublands that were much more common during the early 20th century (Hessburg et al. 2005, 2007). A lack of information on the presence of snags and dead wood limits any inference on fire severity in forests from studies based only on live trees (Reilly and Spies 2015, Reilly et al. 2017). Uncertainties about fire history are unlikely to be resolved given the limits of historical information (especially prior to Euro-American colonization) and the heterogeneity of ecosystems. In the end, the details of historical regimes (e.g., the level of high-severity fire in the past) may not be as important as what society wants and can have for their forests given changing climate, succession, and fire behavior (see chapter 12).

Effects of Fire Exclusion

Forest structure and composition—

Dry forests—There is less debate in the literature about the effects of fire exclusion on forest structure and composition in dry forests where fire was historically frequent. Nationally, over 95 to 98 percent of all wildfires are suppressed while small during initial attack (i.e., 2 to 5 percent escape initial attack) with suppression in the NWFP area

especially common in dry forests (fig. 3-10, table 3-3). Many of these fire starts would have resulted in larger fires that would have altered forest structure and fuel beds and created or maintained early- and mid-successional vegetation over much of the region in the ensuing century.

The recent trends in fire extent and severity in the NWFP area (chapter 2) suggest that fire has generally been less common in recent decades than would be expected under the historical fire regimes (Reilly et al. 2007) (table 3-4), especially given the occurrence of the warmest decade (~1995–2005) since the early 1900s (Abatzoglou et al. 2014) and the historical link between fire and temperature, and drought. The amount of fire (fire rotation) in the frequent and very frequent regimes (117 to 182 years for federal lands) has been considerably less than the historical range for these two dry forest regime classes (5 to 50 years) (table 3-4). For example, in the very frequent regime, most areas would have burned at least once (e.g., a fire rotation of less than 25 years), if not more, during 30 years, the length of the recent satellite record.

Forests have responded to the lack of fire in the two dry forest fire regimes through increases in density and changes in composition. It is well documented that the structure and composition of these forests have changed across the Western United States since Euro-American settlement (Hann et al. 1997; Hessburg and Agee 2003; Hessburg et al. 2005, 1999a, 1999b, 1999c, 2000; Lehmkuhl et al. 1994) as a result of fire exclusion. For example, forests are now typically several times denser in most locations than under native fire regimes (Camp 1999; Dolph et al. 1995; Hagmann et al. 2013, 2014; Merschel et al. 2014; Perry et al.

Table 3-3—Number of lightning fire starts^a between 1992 and 2013 in summer months (June–September) on federal forest lands in the Northwest Forest Plan area^b

Regime	Total fire starts	Number per 25,000 ac (10 117 ha)
Infrequent, high severity	4,271	12.2
Moderately frequent, mixed severity	2,350	13.4
Frequent, mixed severity	2,511	15.2
Very frequent, low severity	4,240	17.4

^a Most of these would have been suppressed by fire crews.

^b Sources of data: Bureau of Land Management Wildland Fire Management Information system; U.S. Fish and Wildlife Wildland Fire Information System; U.S. Forest Service fire statistics.

Table 3-4—Comparison of historical fire frequencies and rotations (in years) with recent (1985–2010) fire rotation estimates from satellite remote sensing for the Northwest Forest Plan area by fire regime class^a

Historical regime class and fire frequencies in years	Range of frequencies from historical studies, all fires (number of studies)	Range of estimates of historical rotations, all fires (number of studies)	Recent rotations (all severities) for USFS lands/all ownerships	Recent rotation (high severity) for USFS lands/all ownerships	Recent frequency (low severity) for USFS lands/all ownerships
Infrequent, high severity (200 to 1,000 years)	No data	296–834 (5)	758/1,525	1,628/3,326	3,056/6,069
Moderately frequent, mixed severity (50 to 200 years)	40–246 (19)	78–271 (6)	582/1,055	2,398/4,530	1,321/2,342
Frequent, mixed severity (15 to 50 years)	21–27 (2)	No data	110/276	333/851	305/761
Very frequent, low severity (5 to 25 years)	3–36 (18)	11–64 (4)	111/143	690/852	218/286

^a See appendix 3 for fire history data. Recent data from Reilly et al. (2017). USFS = U.S. Forest Service.

2004; Reilly and Spies 2015; Ritchie et al. 2008; Stephens et al. 2015; Youngblood et al. 2004), and composition has shifted toward shade-tolerant species. Baker (2012) did not agree with this characterization and described these forests of the late 1800s as historically “generally dense.” However, the finding that his method overestimates tree density by 20 to 380 percent (Levine et al. 2017) suggests that forests were not generally dense as he claims, and data may be coming from a period in which shifts from a more frequent fire regime had already occurred as a result of various effects of Euro-American colonization (Fry and Stephens 2006, Norman and Taylor 2005, Skinner et al. 2009), including the loss of burning¹¹ by American Indians. Even if the overestimates of the Baker (2012) method are at the low end of the range of bias found by Levine et al. (2017), they are still lower than the least dense areas found in contemporary forests (Merschel et al. 2014, Reilly and Spies

2015). Baker (2012) estimated that the interquartile range (25th to 75th) for density in mixed conifer was 69 to 142 trees per ac (170 to 352 trees per ha), whereas the interquartile range in current forests was 298 to 586 trees per acre (736 to 1,447 trees per hectare) an increase of 67 to 75 percent. Consequently, the 2012 Baker paper cannot be used as evidence that forest density has not substantially increased since the 1900s—only that the increase may not be as large as some studies indicate.

A consequence of succession in these forests is that dense understories of shade-tolerant species can shade out pine regeneration and eventually provide abundant seed sources that compete with pine regeneration in lower fire severity postfire environments. Restoring the dominance of large fire-tolerant tree species in these forests is a key component of restoration strategies (Hessburg et al. 2016). The accumulated seed source of shade-tolerant species in these landscapes and large-landscape inertia has probably altered the successional probabilities following fire disturbances toward shade-tolerant pathways as Stine et al. (2014: 140) indicates:

¹¹ Note that American Indians were marshalled onto reservations by 1850, and with this came the loss of intentional burning that occurred near seasonal encampments and customary food production and gathering places (Stewart 2002).

Landscapes exhibit varying degrees of inertia. The degree of change over the 20th century in forest structure, tree species composition, and disturbance regimes has given landscapes an inertia (which can be thought of also as ecological momentum or resistance to change) that will be difficult to alter through restoration-based management. For example, field observations suggest that after recent wildfires, instead of regenerating to ponderosa pine or western larch, some areas now quickly regenerate to Douglas-fir and white, grand, or subalpine fir, or lodgepole pine, despite intentional efforts (which often fail unless done well) to reestablish ponderosa pine or larch. The presence of abundant seed from shade-tolerant tree species (e.g., firs) provides this inertia. Likewise, high contagion of surface and canopy fuels creates large homogeneous patches that reinforce the occurrence of a higher than normal number of large and very large fires, and higher than normal fire severity.

This landscape-scale successional trend may be locally disrupted by large disturbances, but if the rate of disturbance is not high enough, or the disturbance does not kill the shade-tolerant species over large areas, the trend is likely to continue unless climatic changes alter the disturbance regime and the growth or survivorship of tree species.

Moist forests—Fire suppression also appears to be having an effect on the amount of fire in the moist, west-side forest fire regimes (Agee 1993) (figs. 3-4 and 3-10). Over 6,600 lightning-started fires were recorded in this region over a recent 21-year period, and most of these would have been actively suppressed (table 3-3). Although the vast majority of these fires probably would not have turned into large high-severity or mixed-severity fires, a few probably would have. Before the era of fire suppression, a few of these starts likely smoldered for weeks as small fires or as burning snags until a dry east wind event occurred, when those fires could spread rapidly producing large patches of high-severity fire along with patches of moderate- to low-severity fire. Recent fire rotations for high-severity fires in the two west-side fire regimes also appear to be at the high end of the historical range for U.S.

Forest Service lands (table 3-4) (Reilly et al. 2017). Historical fire occurrence in these regimes varied at centennial scales with climate and human population density (e.g., Weisberg and Swanson 2003). Thus, given the occurrence of warm, dry conditions during much of the contemporary fire period, a rotation exceeding the upper end of the range suggests we are currently experiencing much less fire than would have occurred historically under a similar climate.¹²

The effects of fire suppression in the moist, west-side forests are quite different than in the dry forests. Fire suppression in relatively productive forests with long-fire-return intervals has little effect on fuel accumulation at the stand level (Agee 1993). However, fire suppression would drastically reduce the amount of early- and mid-successional vegetation in the landscape and thereby, reduce landscape-scale heterogeneity in forest composition, structure, and patch sizes. Mixed-severity fires burning at rotations of 50 to 200 years would have created a mosaic of forest successional stages, including multicohort old-growth stands (figs. 14, 16, and 17) (Tepley et al. 2013).

Fire severity in dry forests—

Although weather is the primary controller of fire occurrence, size, and severity, in some cases, in the NWFP area (Littell et al. 2009, Reilly et al. 2017), local controls (e.g., topography and fuels) are also important (Cansler and McKenzie 2014). There is significant concern that accumulation of live and dead fuels in understories as a result of fire exclusion and suppression has increased the threat and occurrence of larger areas of high-severity fire (Hessburg et al. 2000, 2005; Miller and Urban 1999a, 1999b, 2000; Parsons 1978, Parsons and DeBenedetti 1979). This threat is thought to arise from two processes: (1) increased accumulations of surface and ladder fuels (shrubs, small trees, lower canopy base heights) that increase flame length and fireline intensity under extreme fire weather conditions, and risk of mortality, even in large fire-resistant canopy trees; and (2) higher spatial continuity of fuel beds that can lead to

¹² Note, however, that for the infrequent and moderately frequent regimes, the recent 25-year record is very short and does not necessarily indicate deviation from historical regimes where fires were relatively infrequent (e.g., 505 to 1,000 years). Note also the relatively small sample sizes of fire history studies.

more rapidly spreading and larger patches of high-severity fire (fig. 3-26). These changes in fire behavior as a result of fuel accumulation are supported by theory, simulation models of fire behavior, and empirical studies of differences in fire behavior between stands where fuels have been reduced by mechanical and prescribed fire and those that have not been treated (North et al. 2012, Ritchie et al. 2007, Safford et al. 2012b, Schmidt et al. 2008, Stephens 1998, Stephens and Moghaddas 2005, Stephens et al. 2009, Weatherspoon and Skinner 1995). Evaluation of changes in fire patch size distributions with those of pre-Euro-American settlement era fire regimes are problematic because we lack landscape-scale quantitative data on frequency-size distributions of fire-severity patches for most areas (Collins et al. 2006; Collins and Stephens 2010; cf. Perry et al. 2011; Reilly et al. 2007; Williams and Baker 2014) (app. 3).

Empirical evidence for increasing total area of fire, and increasing area of fire patch sizes in recent decades, exists from studies across the Western United States, which are relevant to the NWFP area (Cansler and McKenzie 2014, Littell et al. 2010, Miller et al. 2008, Odion et al. 2004, Reilly et al. 2017, Westerling et al. 2006). However, evidence for increased proportion of high-severity fire in recent decades is mixed. Lutz et al. (2009) found evidence for increasing proportion of high-severity fire in the Sierra and southern Cascades of California, but Miller et al. (2012) did not find evidence of increasing total proportion of high fire severity from northwest California between 1987 and 2008. Miller et al. (2012) did find the sizes of high-severity patches to be increasing along with the overall increasing size of fires. Baker (2015a) did not find evidence for increasing proportion of high-severity fire in recent years in a study of ponderosa pine and mixed-conifer forests of the Western United States. Reilly et al. (2017) found no increases in the proportion of area burned at any level of severity between 1985 and 2010 in the Pacific Northwest but did see increasingly severe fire effects (e.g., large patches of high-severity fire) related to drought and annual area burned. Cansler and McKenzie (2014) found significant positive relationships in the northern Washington Cascades between climate and fire size, and between fire size and the proportion of fire events found in high-severity fire patches.

They also found that the spatial aggregation of high-severity area within fires was greater in ecoregions with more contiguous subalpine forests and less complex topography.

It also appears that while recent fire frequencies for all severity classes are below what would have been expected for all the historical fire regimes in the region, the proportion of high-severity fire in fire-frequent regimes may be somewhat higher than it would have been historically. However, note that the recent rotations of high-severity fire in dry forests are still very low (table 3-4). Reilly et al. (2017) found that the amount of recent high-severity fire (23 to 26 percent) in the ponderosa pine, grand-fir, white fir, and Douglas-fir potential vegetation types was higher than what would be expected for these types under historical fire regimes. Mallek et al. (2013) reported that the percentage of high-severity fire in mixed-conifer forest types of the Sierra Nevada and southern Cascades of California was 5 to 8 percent during the pre-Euro-American period but was 22 to 42 percent in dozens of fires between 1984 and 2009. Miller and Safford (2012) reported that larger recent fires in pine and mixed-conifer forests in the southern cascades of California experienced 33 percent high severity, which was probably higher than the historical amount of high-severity fire. However, Odion et al. (2004) found that fires in 1987 in remote areas of the California Klamath had relatively low percentages (12 percent) of high-severity fire (defined as 100 percent scorch or consumed) and the percentage of high-severity fire in the 2002 Biscuit Fire was only 14 percent (Azuma et al. 2004). The relatively low percentage of high-severity fire in 1987 may be a result of weather conditions that were not as extreme as those of more recent fires (Taylor and Skinner 1998, Weatherspoon and Skinner 1995). Although the forests of the Klamath may have been less affected by fire suppression than more accessible forests, fire-return intervals during the suppression period are still nearly 50 percent longer (21.5 vs. 14.5 years) than during the presettlement period (Taylor and Skinner 1998). As fire sizes increase with climate warming (Odion et al. 2004), patch sizes of high-severity fire may also increase (e.g., Miller et al. 2012, Reilly et al. 2017). Very large patches of high-severity fire that kill older, dense forests would not be characteristic of the very frequent low-severity regime (Taylor and Skinner 1998), and efforts

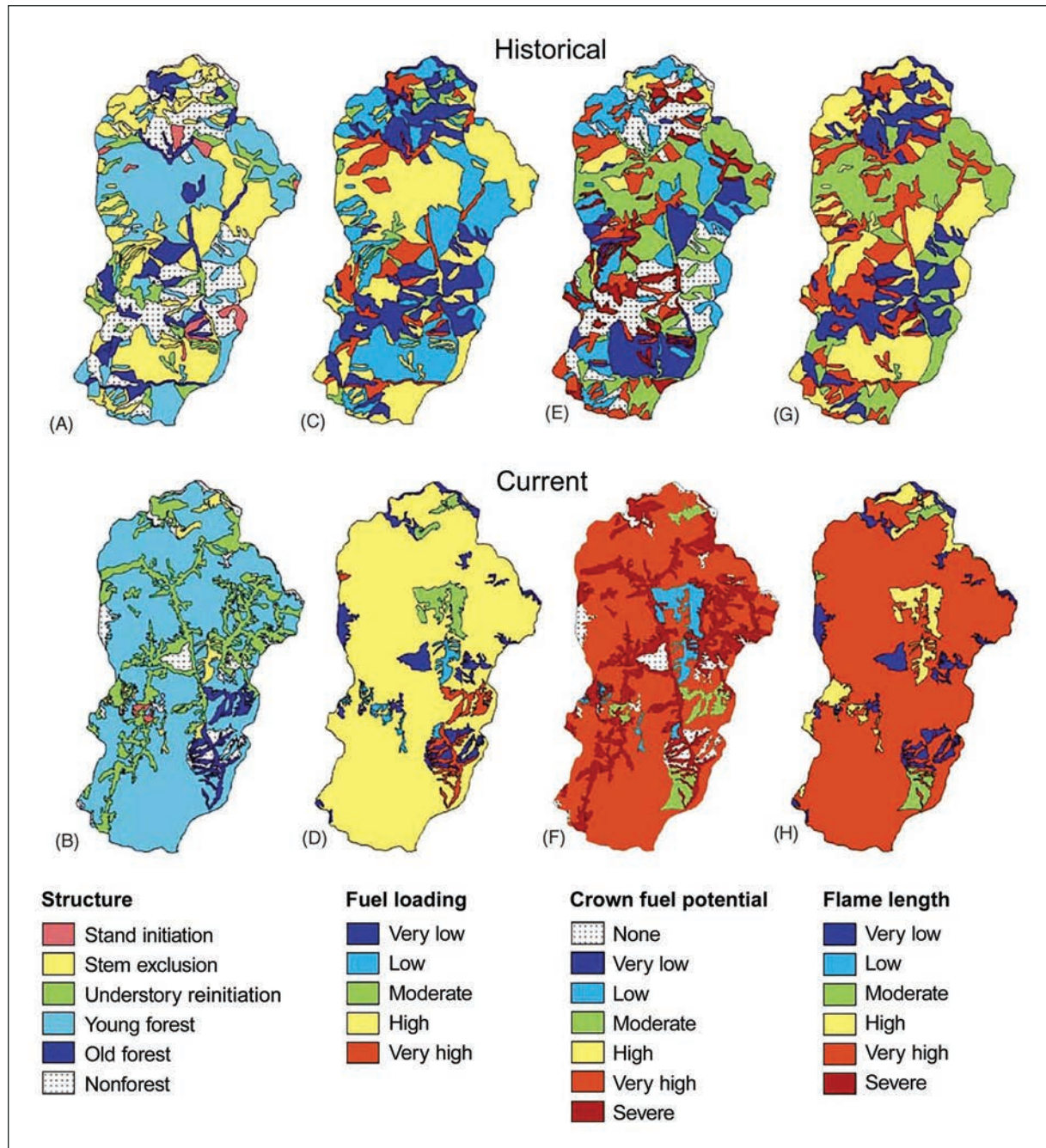


Figure 3-26—Reconstructed historical (1900s) and current (1990s) maps of dry forest subwatershed of the Lower Grand Ronde subbasin in the Blue Mountains province displaying historical and current structural classes (A and B), fuel loading (C and D), crown fire potential under average wildfire conditions (E and F), and flame length under average wildfire conditions (G and H), respectively. (From Hessburg et al. 2005). Although this is from a landscape outside of the Northwest Forest Plan (NWFP) area, similar changes have likely occurred in dry forests in many areas within the NWFP.

to restore frequent fire and reduce fuels in older and younger forests would contribute to maintaining the biodiversity (including spotted owls in the southern part of their range) that was adapted to a dynamic and heterogeneous mix of forest ages and structures.

Factors explaining variation in how fire-excluded forests burn when wildfire returns are not well understood. The observation that dry forests are experiencing less fire (excluding a direct effect of fire suppression), but more high-severity fire, or larger patches of high-severity fire than was true historically, is related to climate and fire suppression, but may also be due to shifts in vegetation-fire feedbacks. For example, it may be that with the absence of fire, coupled with succession to shade-tolerant and fire-intolerant species, is leading to forests that are less flammable under typical fire weather owing to a number of factors, including moister microclimate, denser stands that inhibit the free flow of wind, lower air and fuel temperatures owing to less direct sunlight, and more compact fuel beds (Engber et al. 2011, Estes et al. 2012, Kitzberger et al. 2011, Odion et al. 2004). For example, Weatherspoon et al. (1992) suggested that:

...success of initial attack on wildfires evidently is greater in areas of owl habitat within the Sierran mixed-conifer type. Countryman's (1955) description of fuel conditions within old-growth stands applies in large measure to fuel conditions within many mixed-conifer stands used by the California spotted owl. These stands are less flammable under most conditions, because the dense canopies maintain higher relative humidities within the stands and reduce heating and drying of surface fuels by solar radiation and wind. The reduction of wind velocity within closed stands discussed by Countryman is supported by wind reduction factors identified by Rothermel (1983) for stands with closed canopies. Windspeed at mid-flame height for fires burning in surface fuels is approximately one-tenth of the windspeed 20 ft (6.1 m) above the stand canopy.

However, they go on to say that:

As fuels accumulate, however, fires that do escape initial attack—usually those burning under severe conditions—are increasingly likely to become

large and damaging. Success in excluding fire from large areas that were once regulated by frequent, low- to moderate-severity fires has simply shifted the fire regime to one of long-interval, high-severity, stand-replacing fires...

Some areas within the 2002 Biscuit Fire (which had relatively low total area of high-severity fire) could be an example of this shift in this regime, where moist multistoried older forests on north-facing slopes burned with high severity during the most extreme weather periods (hot dry east winds) of the fire (Thompson and Spies 2009).

Note that Countryman (1955) and Weatherspoon et al. (1992) never directly tested the hypothesis of higher humidity and fuel moisture in closed stands vs. more open stands. This was simply assumed to be so. Estes et al. (2012) measured an array of different sizes of fuels in closed, unthinned stands and open, thinned stands from spring snowmelt through fire season to the onset of fall rain/snow in the southern Cascades. They found moisture differences only in the early part of fire season (May–June). Moisture differences were gone by mid-season (July), and this carried through the remainder of the fire season. Further, the more open stands responded more quickly to the few rain events (thunderstorms) than did the closed stands. It appears that the long, dry summers of the Mediterranean climate areas in the southern parts of the NWFP area negate potential differences in moisture conditions because the closed stands catch up with the dry conditions of the open stands as the fire season progresses. Thus, the ability for crews to more readily catch fires in closed stands appears to be due to differences in exposure to sunlight creating higher air and fuel temperature and greater ease of windflow in the open stands.

Thinning can alter fire potential and microclimate. Higher windspeeds in thinned stands compared to unthinned stands may have contributed to the former burning with higher fireline intensity (Raymond and Peterson 2005) than the latter in the 2002 Biscuit Fire. Although most of the differences in fire effects in that study were attributed to higher fine fuel loading and lower moisture in the stands that had been thinned but were not underburned to reduce fine fuels. Bigelow and North (2012) noted that thinning and group selection can change microclimates of forests but they did not find that such changes had a large effect on fire behavior.

The interaction between vegetation and fire severity is also determined by foliar moisture of the herbaceous, shrub, and hardwood fuels. For example, in open dry forests subject to frequent fire, well-developed herbaceous layers can reduce flammability because moisture contents can remain high into September (Agee et al. 2002). In the Klamath Mountains and western Cascades, hardwood understories can significantly reduce fire intensity (Agee et al. 2002, Perry 1988, Perry et al. 2011, Skinner 2006, Skinner and Chang 1996). Some species of evergreen shrubs can also reduce flammability of forests landscapes under most weather conditions but provide dense flammable fuels under extreme fire weather conditions (Skinner and Weatherspoon 1996, Weatherspoon and Skinner 1995).

Weatherspoon and Skinner (1995) suggested that another reason for the differences between stands of larger, old trees and those of smaller young trees and plantations experiencing different levels of fire severity in the Klamath could be simply the susceptibility of trees of different sizes to damage by fire. Large trees, especially stands dominated by old Douglas-fir and ponderosa pine, would be more likely to survive fires than younger trees, especially small trees in plantations (Agee and Skinner 2005, Skinner et al. 2006). Although these multistoried stands have similar-size trees that succumb to the fires as do the young stands or plantations, the mortality is often hidden from satellite sensors by the surviving older, main canopy trees. Thus, the older stands become classified as experiencing mostly low-severity fire effects, while the others are classified as moderate- to high-severity fire effects even though fire intensity and sizes of trees actually killed could have been very similar (Weatherspoon and Skinner 1995). This is another example of the challenge of defining fire severity using single or simple metrics across variable vegetation types, and a potential source of confusion and debate (Reilly et al. 2017).

Use of Historical Ecology in Conservation and Restoration

As illustrated above, knowledge of the ecology of the period prior to Euro-American settlement and widespread changes in land use can be very useful in understanding these forests and can serve as a starting place for developing conserva-

tion and restoration plans and management practices for them (Allen et al. 2002; DellaSala et al. 2003; Demeo et al. 2012; Hessburg et al. 1999a, 1999b, 1999c, 2000, 2005; Keane et al. 2002, 2009; Landres et al. 1999; Morgan et al. 1994; Safford et al. 2012a; Swetnam et al. 1999). Knowledge of ecological history and knowledge of the historical range of variation (HRV) are not necessarily the same thing. General knowledge of ecological history may be more useful in management than a precise understanding of the range of variation in forest conditions (Hiers et al. 2016), which cannot be fully achieved for a number of ecological and social reasons. For example, while we may lack precise models or reconstructions of HRV for many landscapes in the region, we do have a reasonable foundation of historical knowledge for most areas. Ecological history reveals that forests were dynamic and best understood in terms of a HRV or its equivalent natural range of variation. The concept of HRV recognizes that habitats and ecosystems are dynamic in space and time, with historical ranges of behavior that are strongly constrained by the dominant climate, environment, and disturbances of an ecoregion. For the NWFP area, the HRV of forest structure among the four major fire regimes would have differed based on fire frequency and severity patterns and scale as described in the previous sections (fig. 3-27). Likewise, the HRV of forest structure would have differed across the major disturbance regimes based on whether small- to medium-size severity patches or high-severity patches were the major successional influence controlling patch dynamics.

Application of historical ecology HRV concepts and potential vegetation types in the Pacific Northwest and northern California must recognize the central role of climate variability in forest dynamics (Keane et al. 2009, Wiens et al. 2012, Wimberly et al. 2000). Temporal variation in climate drove the variability of historical fire regimes (Hessburg et al. 200b, 2004; Trouet et al. 2010), which are the product of interactions between forest composition and structure, fire weather, and ignitions. Variation in climate and fire regime was the driving force of the “range” in the HRV in forest structure and composition. For example, fire occurrences in many of the moist and cool forests of the region are “climate limited” (Briles et al. 2011, Colombaroli and Gavin 2010, Littell et al. 2009) or

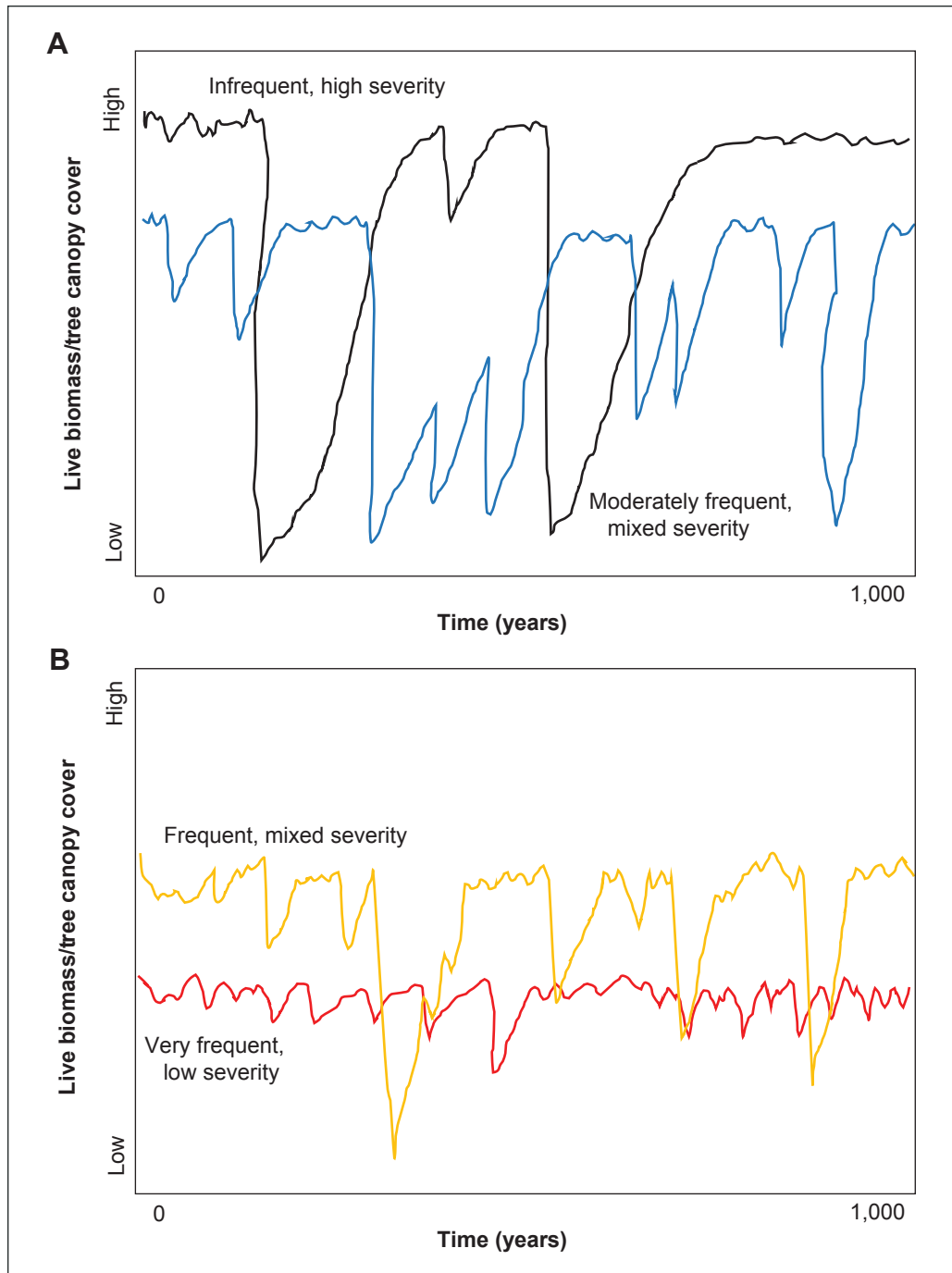


Figure 3-27—Hypothesized dynamics (historical range of variation) in live forest structure (biomass or cover) over a hypothetical 1,000-year period during the pre-Euro-American settlement period for an area of several thousand acres for (A) moist forest fire regimes and (B) dry forest fire regimes. Large declines in live biomass result from fire or wind; small declines result from fire, wind, insects, and disease.

“ignition limited” (sensu Agee 1993), but not fuel limited as the environments are typically productive enough to produce adequate fuels for burning within 10 to 15 years of a fire. If shrubs such as *Ceanothus* are present, they can act as a barrier to fire spread under less-than-extreme burning conditions (Briles et al. 2005, Mohr et al. 2000, Whitlock et al. 2004), or encourage rapid and intense fire spread under extreme fire weather conditions (Agee 1993, Moritz 2003, Schmidt et al. 2008).

Regionally, wildfire was episodic and could be synchronous in parts of the region especially in wetter climates of the high- and mixed-severity regimes (Weisberg and Swanson 2003). Although fires were frequent in the driest forest regions, variability in frequency existed, and climatically driven synchrony of widespread fire still exists even in the fire-frequent forests of the Western United States (Falk et al. 2011, Heyerdahl et al. 2008). Wildfire frequency was variable at decadal to millennial scales, i.e., it was nonstationary. According to Whitlock et al. (2008), who examined paleo fire history of forests of the Northwestern United States, “There is no stable fire regime on millennial time scales, because fire-episode frequency varies continuously as a consequence of long-term climate variations and their influence on vegetation.” They go on to say, “Without supporting long-term paleoecologic data, short-sighted inferences about natural disturbance regimes and forest sensitivity are likely to be incorrect.”¹³ In other words, there were periods with relatively less frequent fire and other periods with relatively more frequent fire, creating a larger HRV if climate context is not taken into account. However,

in drier parts of the region with more frequent fire, large-scale temporal variability and regional synchrony in fire was probably less than in regions with less frequent but larger fires (Hessburg et al. 2005; Heyerdahl et al. 2001, 2008; Kitzenberger et al. 2006; Mohr et al. 2000; Morgan et al. 2008; Skinner et al., in press; Taylor et al. 2008; Trouet et al. 2010). Nevertheless, regionally extensive fire events associated with drought did occasionally occur in the eastern Cascades of Washington (Hessl et al. 2004).

Going forward, several authors have argued that given climate change, invasive species, and widespread landscape change, using historical conditions or ranges of variation as a narrow goal or target for conservation and restoration can be unrealistic, impossible, or even incongruent with conservation goals (Millar et al. 2007, Palmer et al. 2005). This is especially true if the goals include threatened and endangered species, such as the northern spotted owl in dry forests, whose habitat can be the product of human land use activities and altered disturbance regimes. However, it is self-evident that knowledge of historical forest dynamics is essential for conservation and restoration of native (historical) vegetative communities and associated wildlife species even under climate change. The challenge for application of the concept is to be aware of limitations and apply historical knowledge with caution. Hessburg et al. (2016) offer four caveats to using historical reference conditions as management guidelines:

- Mimicking historical conditions is not an end in itself, but is a means of accomplishing objectives (e.g., resilience to fire), and therefore appropriate only when it meets those objectives.
- The true value of historical information is in understanding how interacting fire and climate, and their variability through time and space, influenced ecological patterns of forest structure and successional conditions. This information can provide valuable direction for the complex process of ecological goal setting in management planning and implementation.
- Past conditions may not fully reflect future climate-vegetation-disturbance-topography linkages as a result of pervasive climate and

¹³ Although paleoecological fire histories can give us a broader perspective on HRV, they are subject to methodological limitations. For example, fire history studies based on charcoal occurrence in sediment cores are subject to bias because charcoal production is partially determined by the nature of the fuels (e.g., herbaceous vs. woody). In the Klamath Mountains, the frequency of fire generally exceeded the resolution of the sediment cores, which was usually no finer than 30 years at best (Briles et al. 2005, Mohr et al. 2000, Whitlock et al. 2004). Further, over most of the Holocene, there was rarely a time when charcoal was not entering the lakes in the Klamath region. Rather than being an indicator of fire events, the amount of charcoal at different periods appeared to be more indicative of biological productivity. Charcoal varied by amount with the periods of light, flashy fuels characteristic of pine/oak woodlands represented by lower charcoal influx than in more productive periods characteristic of mixed-conifer forests (Mohr et al. 2000, Skinner et al. 2006, Whitlock et al. 2004).

land-use changes. Hence, one of the challenges may be deciding the degree to which past lessons are relevant to future management. Relevance will depend on goals, reasonable expectations of the future climate, and resources required to attaining those goals.

- Because regional landscapes are highly altered, restoration restricted to local landscapes is insufficient to address large-scale restoration needs.

Remember that we understand recent HRV (e.g., past 500 years) better than we understand HRV of the more distant past or what the range of variation will be in the future. Consequently, planning efforts based on ecological history or HRV will need to be flexible, adaptive, and periodically revised to keep up with new knowledge and changing ecosystems. To deal with the challenges of restoration or managing for resilience, Hobbs et al. (2014) recommended that landscape frameworks and assessments be used to identify where it is possible to retain or restore native biodiversity and where novel or “hybrid” (seminatural) ecosystems might be a management goal either because of human values (e.g., areas of dense forests for wildlife created by fire exclusion) or because of the impracticality or impossibility of returning those areas to their pre-Euro-American state or HRV (see chapter 12 for more discussion of this issue). We further discuss scientific understanding of approaches for dealing with these and other challenges of restoration or creating resilient forests in sections below.

Ecosystem Function

The preceding sections have emphasized forest structure, composition, and disturbance process, but ecosystems can also be characterized through their functions (ecological processes or activities), which also differ with successional stage and disturbance regime. Key functions include primary productivity and carbon fixation, nutrient cycling, hydrological functions, and habitat for biota (Franklin et al. 2018). We briefly review how these differ with succession here with a focus on productivity, carbon and nutrient cycling. For more information about hydrological functions and habitat, see chapters 6 and 7.

Old-growth forests are productive ecosystems, fixing a large amount of solar energy in what is termed gross primary production (Franklin and Spies 1991). Following major disturbances, ecosystem live biomass and net primary productivity (difference between carbon fixed through photosynthesis and lost to respiration) are relatively low (Bormann et al. 2015, Spies 1997), in contrast with later successional stages. As trees grow and canopies close, the rate of carbon sequestration and biomass accumulation becomes high. Biomass reaches its highest level in older forests, but net primary production declines toward zero because growth and mortality are roughly equal. While stand-level net primary productivity and carbon accumulation is low in older forests, the rate of biomass growth for individual trees continues to increase with tree size (Stephenson et al. 2014).

Carbon, which primarily resides in the wood and soils, is highest in old forests (Law and Waring 2015). Douglas-fir/western hemlock forests can continue to be a net sink for carbon for more than 500 years, thanks to the contribution of primary production of shade-tolerant understory trees (Harmon et al. 1990, 2004). Older moist forests of the NWFP area can attain higher stand (tree) carbon biomass than tropical or boreal forests (Law and Waring 2015). Young forests store less carbon but accumulate it at higher rates than old forests.

Recent large wildfires in coniferous forests of the region release carbon, but the total emitted carbon is less than previously thought, partly because most fires in the region have burned with mixed severity. For example, Campbell et al. (2007) found that only 1 to 3 percent of the carbon in trees larger than 3 inches (7.6 cm) was combusted in the 2002 Biscuit Fire (Campbell et al. 2007). Total carbon emitted from four fires in Oregon averaged 22 percent of prefire carbon for all pools (Meigs et al. 2009). As the biomass killed in fires slowly decomposes over decades to centuries, carbon is emitted to the atmosphere as carbon dioxide and other trace hydrocarbons. About half the carbon remaining after a fire stays in the soil for about 90 years; the other half persists for more than 1,000 years as charcoal (Deluca and Aplet 2008, Law and Waring 2015).

Forest management effects on carbon differ with management intensity, rotation length, and forest type. It is often thought that managing forests on a short rotation (e.g., 40 to 50 years) would provide the most effective long-term carbon sequestration, but longer rotations and selective or no harvest provides the most carbon sequestration (Harmon et al. 1990, Ryan et al. 2010). Forest management under the NWFP to promote older forests with a low level of timber harvest would provide for more carbon sequestration than more intensive management (Creutzburg et al. 2017, Kline et al. 2016).

In forests prone to frequent fires, the carbon and forest management picture is more complex, with some studies showing a positive benefit of forest fuel reduction on carbon sequestration and others showing a negative effect. Some modeling suggests that carbon stocks over the long term are best protected by fuel treatments that create relatively low-density stands dominated by large, fire-resistant trees (Hurteau and North 2009). Other studies (Ager et al. 2010, Loudermilk et al. 2016, Spies et al. 2017) found that active management reduced carbon stored in the forest landscape by 5 to 25 percent for at least several decades. The effect of management on carbon depends on how frequently management treatments encounter fire and reduce fire severity. When a fire encounters a recently treated area, less carbon is likely to be emitted than when it encounters an untreated forest of the same type. However, at a landscape scale, many treatments will not experience a fire and the management actions there will reduce carbon sequestration. The net effect at a landscape scale may be to reduce carbon sequestration unless those treatments are strategically placed and occur where fire is most likely to happen. Further, the more active the fire regime becomes under climate warming scenarios, the more important strategically placed fuels treatments (e.g., Finney et al. 2007, Schmidt et al. 2008) become in protecting carbon stores (Loudermilk et al. 2013, 2016).

Nutrient cycling varies with successional stages and forest region. Old-growth forests are highly retentive of nutrients, and sediment outputs from old-growth watersheds are typically low (Franklin and Spies 1991, Swanson et al.

1982). Many of the forests of the NWFP area are nitrogen limited, but several natural processes exist that capture nitrogen and make it available for vegetation growth. old-growth forests can support canopy lichens such as *Lobaria oregana*, *L. pulmonaria*, and others that fix nitrogen and then “leak” significant amounts of nitrogen to the ecosystem (Antoine 2004). Immediately following stand-replacement disturbance, rates of erosion and nutrient loss can be elevated until vegetation recovers (Ice et al. 2004). As plants establish and cover increases during early-successional and young forest stages, sediment losses return to pre-disturbance levels, and N_2 -fixers such as *Ceanothus* spp. and hardwoods such as red alder (*Alnus rubra*) begin to increase organic matter and nutrient availability (Borman et al. 2015, Compton et al. 2003). While red alder can add available nitrogen to forest ecosystems, the high rates of nitrification can accelerate cation leaching and soil acidification relative to conifer-dominated stands (Compton et al. 2003). Shrubs and sprouting hardwood trees can also help reduce nutrient losses after wildfire in forests of southwestern Oregon. While the longer term benefits of early-seral plant communities to conifer tree growth are still not well understood (Bormann et al. 2015), it is generally understood that early-seral herbaceous, shrub, and hardwood tree communities can all play an important role in supporting forest nutrient cycling and productivity.

Restoration efforts in dry forests can also benefit soil fertility and productivity. Fire suppression can lead to increases in nitrogen pools in ecosystems, but the majority is bound in forms that are less available to plants (Ganzlin et al. 2016). Forest restoration treatments, including prescribed burning, can produce short-term pulses of nitrogen in forms that are available to plants. Thinning alone will not produce these nutrient benefits and is not an effective surrogate for fire in terms of nitrogen. Frequent prescribed fire that emulates historical fire frequency and severity is necessary to maintain rapid rates of nutrient cycling in these dry forest ecosystems. However, while the nutrient effects of fire may be ephemeral, benefits to other soil resources and processes such as available water and photosynthetic rates may be longer term (Ganzlin et al. 2016).

Conservation and Restoration Needs

In this section, we summarize the major conservation (e.g., protection of existing vegetation) and restoration (e.g., promotion of desired conditions) needs for moist and dry forests relative to the original goals of the NWFP and of the 2012 planning rule under which the NWFP currently operates (table 3-5).

Estimates of forest change for the NWFP region suggest that the need for conservation and restoration of the ecological integrity of old-growth forests and other successional stages of the region spans a wide range of the disturbance regimes and forest types. For example, Haugo et al. (2015), found that at least 40 percent of all coniferous forests in eastern Washington and eastern and southwestern Oregon are in need of management to restore wildfire, fuel, or forest structure conditions to be more consistent with the natural range of variation. After more than 125 years of land clearing, timber harvest, 20th century high-severity wildfire associated with early logging and land use, fire suppression and succession, the sum of mature and old-growth forest (OGSI 80) across all the fire regimes is 17.8 million ac (7.2 million ha), or ~ 39 percent of all public and private forest-capable lands in the Plan area (Davis et al. 2015). When only the oldest multilayered forests with trees >200

years old (OGSI 200) are considered, the current amount is ~7.6 million ac (3.1 million ha), or 17 percent of all public and private forest-capable lands. Of that 17 percent, more than 80 percent is on federal lands. It is difficult to estimate what percentage of the historical range of older forests this represents for several reasons, including lack of quantitative studies of HRV across the region, uncertainties in estimates of HRV, and the current definitions do not fully capture the diversity of older forest conditions, especially for older ponderosa pine and mixed-conifer forests of the low- and mixed-severity regimes. If we focus on trees older than 200 years (OGSI 200) in moist forests zones west of the Cascade crest, then the total remaining may represent 17 to 23 percent of the amount that was present on average before the mid-1800s. This assumes that at least 60 percent of these forests areas were covered by forests containing trees older than 200 years (FEMAT 1993, Wimberly 2002).

Moist forests—

In the moist forests zone, losses of older forest have resulted mainly from clearcutting for timber management (Spies et al. 1994). The decline in older forest has been sharp as indicated above. For example, the vegetation structure of northern spotted owl habitat (not necessarily the same as

Table 3-5—Summary of vegetation conservation and restoration needs for moist and dry forests of the Northwest Forest Plan (NWFP) region related to the ecological goals of the NWFP and the 2012 planning rule

Forest region	Conservation needs	Restoration needs
Moist forests	Protect existing older forests stands and large patches of older forests from logging and high-severity fire. These have been greatly reduced by timber management and other land uses.	Increase vegetation diversity in plantations and accelerate development of older forest structure and composition. Reduce fragmentation and increase connectivity of older forest patches. Create or promote early-seral vegetation where needed to provide seral stage and landscape diversity. Restore disturbance processes (e.g., fire) where feasible.
Dry forests	Protect existing large fire-tolerant trees in areas of dense and open forest. Manage and protect existing dense old-growth forest stands as necessary to meet late-successional species and ecosystem integrity needs.	Restore low- and mixed- severity fire as key ecological process. Increase areas of open old forests to promote resilience to fire and climate change and meet needs of species. Develop landscape-level strategies to create desired mosaics of open and dense old forest and to increase resilience and meet simultaneous needs of wildlife species and ecological integrity. Restore diversity to plantations, including tree species mixes.

old-growth forests) has declined by 20 to 52 percent across the different provinces between 1930 and 2002 (Lint et al. 2005). Many plantations on federal lands are 30 to 60 years old and average about 20 to 25 ac (8.1 to 10.1 ha) with some as large as 60 ac (24.3 ha) (Cohen et al. 2002). They were often planted primarily with Douglas-fir (or at most a total of one or two additional species) at an even spacing. Logging and site-preparation treatments to control competing or unwanted vegetation resulted in uniform stand density with lower levels of shrub and hardwood components, and fewer snags and down wood structures (Bailey and Tappeiner 1998, Spies and Cline 1988). A large percentage of federal forest land outside of wilderness areas is covered by such plantations—as much as 40 to 55 percent of some landscapes, including many late-successional reserves (LSRs) (fig. 3-28). In summary, management efforts to ensure high density and species uniformity were often so successful that conditions in these stands do not match the heterogeneity and growth trajectories of naturally regenerated postwildfire stands (Donato et al. 2011, Freund et al. 2014, Larson and Franklin 2005, Tappeiner et al. 1997, Tepley et al. 2014, Winter et al. 2002a) (fig. 3-14).

Other vegetation restoration needs for the moist forests zone relate to early-seral and other mid-successional stages that have been reduced by fire-suppression reforestation, timber stand improvement treatments that ensured full stocking, optimal sawtimber growing conditions, and control of unwanted vegetation (Agee 1993, Cole and Newton 1987, White and Newton 1989). Fire suppression in these infrequent-fire regimes has little impact on the risk of high-severity fire but it does reduce the amount of early-seral and vegetation diversity in a landscape. Numerous small- to mid-size fires would likely have served as barriers to fire spread where they created patches of deciduous shrubs and trees. The vegetation diversity created by these fires probably regulated the frequency-size distributions, especially of the larger fires. The amount of early-seral condition may have been relatively high (<30 percent) in these regimes during the late 1800s and early 1900s when the legacy of aboriginal burning was still evident (Robbins 1999) along with Euro-American-ignited fires from land clearing and logging (fig. 3-6). The amount and diversity of early-seral vegetation created by these fires would have been reduced where snags were cut down and large-scale

planting efforts reduced the period of time before tree canopy closure. The patterns of early-seral patch size shapes, distribution, and structural heterogeneity created by logging and reforestation in the late 20th century are not representative patterns typically found under historical fire regimes (Nonaka and Spies 2005). The structure and composition of early-successional vegetation and young forests created by clearcut logging significantly differed from those of postwildfire conditions because intensive timber management removed all live and dead trees, and herbicides (in early years on federal lands), and planting of Douglas-fir seedlings reduced diversity of vegetation and shortened the nonforest period of succession. Moreover, harvest unit boundaries often followed land ownership boundaries on private lands, and older cutting units on federal lands (the most recent occurred in the early 1990s) represented small-size (25 to 40 ac [10.1 to 16.2 ha]), regularly shaped units with landscape patterns that differed from those created by fire.

Dry forests—

We have already described many of the changes that have occurred in the dry forests as a result of fire exclusion and logging. Analysis from the Interior Columbia Basin Ecosystem Management Project (Hann et al. 1997; Hessburg et al. 1999a, 2000) provides a picture of how the area of dense multilayered older forest has changed from historical to current (late 1990s) in dry forests of eastern Washington and Oregon (fig. 3-26) (table 3-6).

In another study, Lint (2005) estimated that the amount of dense older forest with grand fir and Douglas-fir that is suitable for spotted owls (we use this as an approximation of multilayered old growth, but it is not necessarily the same as dense old-growth forest structure) has actually increased by 16, 6, and 11 percent in the eastern Cascades of Washington, Oregon, and California Klamath Provinces, respectively, from 1930¹⁴ to 2002. These data suggest that the historical fire regime in these provinces did not favor large areas of either late-successional, multilayered old forest or northern spotted owl habitat.

¹⁴ Landscapes of the 1930s would have already been altered by logging, grazing, fire exclusion, and occurrence of fires associated with land use activities. Fire exclusion would have increased the amount of dense forest by 1930 (McNeil and Zobel 1980, Merschel et al. 2014).

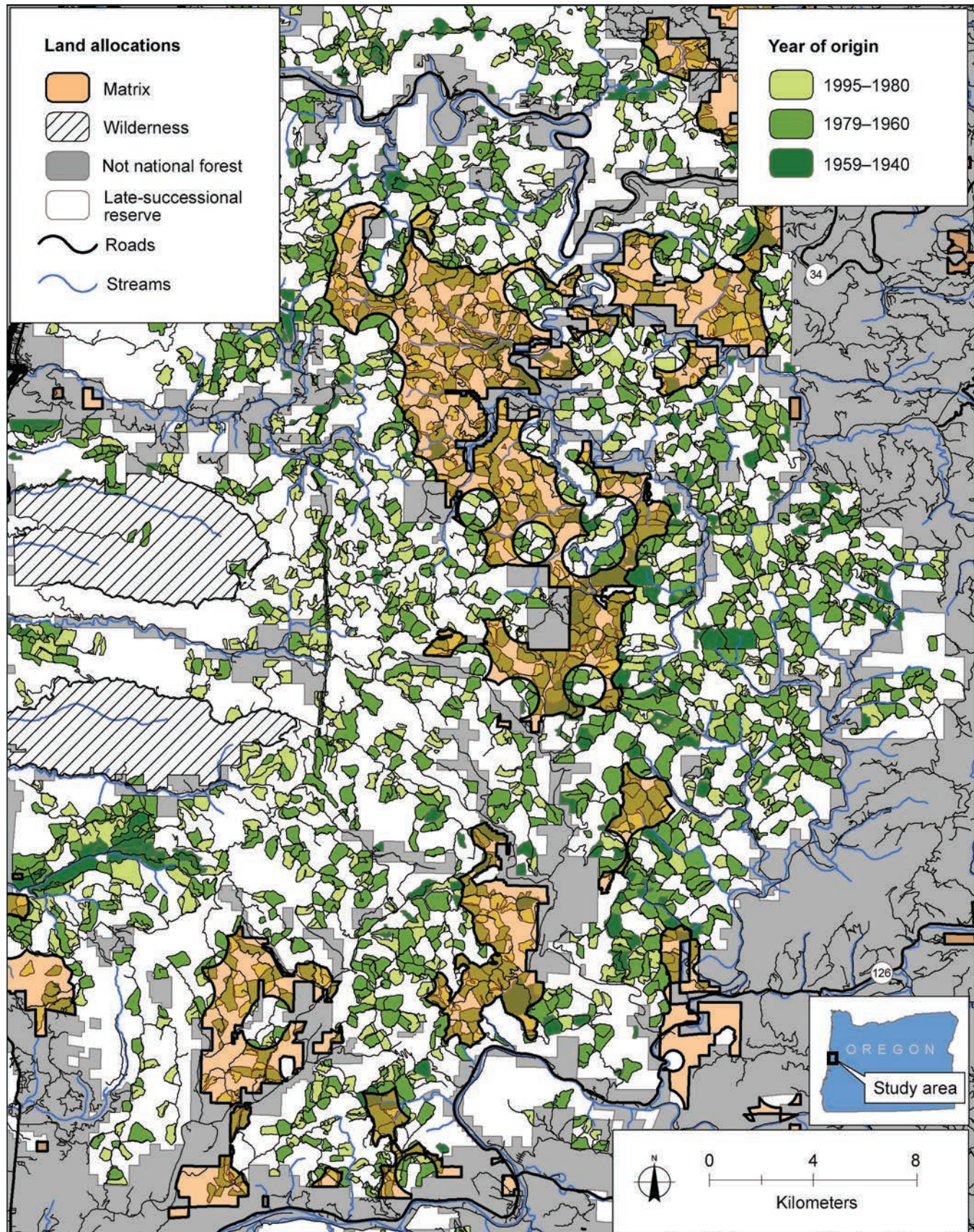


Figure 3-28—Plantations and the dates of their origin in a landscape containing late-successional reserve (in white), wilderness (striped), and matrix (orange) lands on the Siuslaw National Forest in coastal Oregon. From Stewart Johnston (retired), Siuslaw National Forest.

Table 3-6—Historical and 1990s percentages of total forest area in late-successional multistory forest in provinces of the Interior Columbia Basin Ecosystem Management Project

Time period	Province		
	Northern Cascades	Southern Cascades	Upper Klamath
Historical	7.0	0.7	4.8
Current	16.6	4.0	3.5

Sources: Hann et al. 1997; Hessburg et al. 1999a, 2000.

Changes in area of medium and large old trees have also occurred. Hessburg et al. (1999a) documented reductions in province area of forest patches with medium and large trees in the overstory (>40 percent canopy cover) in the interior Columbia River basin. In the Northern Cascades and Upper Klamath provinces, area of medium- and large-size trees in the overstory declined from 30 to 24.9 and from 28.9 to 25.3 percent, respectively. However, area of medium and large trees in the overstory significantly increased in the Southern Cascades province from 17.1 to 32.8 percent. They also show historical landscapes with significant areas of grassland, shrubland, woodland, and stand initiation forest conditions and young forests that had invaded meadows (figs. 3-21 and 3-22). These mid- to late-20th century increases in forest density are in addition to the substantial increases in stand density and shade-tolerant species that occurred between 1890 and 1930 as a result of fire exclusion (owing to grazing, logging, and eventually active fire suppression) and other factors (Merschel et al. 2014, Taylor and Skinner 2003). Currently, the percentage of relatively open, low-density (<80 trees per acre) forest with large old trees in mixed-conifer and Douglas-fir potential vegetation types is about 10 percent, while the area of dense forest (>584 trees per acre (1,442 trees per hectare)) with old trees covers about 35 to 42 percent of the potential vegetation types (Reilly and Spies 2015). These increases in shade-tolerant densities have made forests less resilient to fire as described above.

Increases in forest density are not the only conservation and restoration concerns in the dry forests. Loss of large, fire-resistant trees to logging and wildfire has also strongly affected forest ecosystem integrity, resilience, and wildlife habitat in both the very frequent low-severity and frequent mixed-severity fire regimes of the dry forest zone. For

example, the density of large fire-tolerant tree species (e.g., ponderosa pine and Douglas-fir) has decreased substantially as a result of high-grade logging (selective removal of large mostly commercially valuable trees) (e.g., Hessburg et al. 1999a, 2000, 2003, 2005; Merschel et al. 2014) and clearcutting and plantation establishment. Hagmann et al. (2014) estimated that the area of forests dominated by large old trees has been reduced from 91 to 29 percent for dry and moist mixed-conifer in one landscape in the eastern Oregon Cascades. Increases in future development of large, old fire-intolerant trees may be limited as a result of forest densification and fire suppression. We could find no disagreement in the literature on the issue of restoration needs and concerns for large old conifers (e.g., Baker 2012, Stine et al. 2014). This issue is prominent in the eastern Cascades of Washington and Oregon and in California, where topography and proximity to settlement made these large valuable trees an easy target for logging (Hessburg and Agee 2003; Hessburg et al. 2005, 2015, 2016; Merschel et al. 2014; Richie 2005). Loss of large trees is less of an issue in more remote sites in rugged and difficult-to-access areas such as the less roaded areas of the Klamath Mountains.

Timber Management and Old-Growth Conservation

The NWFP strategy was based on the assumption that historical timber management approaches (e.g., removal of large or old early-seral and fire-tolerant trees) are not compatible with the full ecological functions of old-growth forests and other successional stages. Since FEMAT (1993), no scientific evidence has emerged that intensive timber production (e.g., clearcutting and short-rotation plantation forestry) and old-growth forest conservation are compatible at stand levels for any of these forest types and disturbance regimes.

Moist forests—

In moist forests zones, partial cutting, in the form of green tree retention harvest (see section below for more discussion of this method), patch cutting (creating gaps less than a few acres), or selection harvest methods may retain the habitats of some late-successional animal and plant species (Baker et al. 2016, Gustafsson et al. 2012, Halpern et al. 2012, Hansen et al. 1995a, Rosenvald and Lohmus 2008). It also retains some of the ecological functions of old growth, but could strongly affect dead wood amounts. The accompanying road and harvest systems would add additional impacts. Very long management rotations (e.g., more than 150 or 200 years) could in theory produce some of the habitat and ecosystem service benefits of older forests (Kline et al. 2016), but it would take at least a century to quantify these effects, and no long-term studies are currently underway.

One of the only operational plans to meet both older forest conservation goals and timber production in moist forests in the literature is the “structure-based management” approach proposed by the Oregon Department of Forestry for the state forests in the northern Oregon Coast Range (Bordelon et al. 2000). In this approach, management targets were sorted into five stand types, with the two oldest, “layered” and “older forest structure” intended to meet late-successional conservation goals. There are no reserves, and older forest conditions are met through long rotations. The areas in each stand type can differ over time, e.g., between 20 and 30 percent of older forest structure, as harvesting and succession shift age and structure classes over the landscape. Spies et al. (2007) and Johnson et al. (2007) used a landscape model to approximate this strategy. Modeling results suggest that, over time, this approach created a greater diversity of habitat benefits, including increases in older forest habitats and higher levels of wood compared to federal management under the NWFP. No formal field assessment of the ecological or economic implications of this approach has been attempted. At this stage, the Oregon Department of Forestry is under pressure from the counties to increase revenues and is in the process of modifying or abandoning the approach

(<http://www.nwtimberblog.blogspot.com/2013/11/board-of-forestry-seeks-better.html>; <http://www.northcoastcitizen.com/2016/12/officials-say-county-will-not-opt-out-of-class-action-lawsuit-over-timber-harvest/>).

Other examples of management agency efforts to meet biodiversity and timber management goals exist for moist forests but have not been published or reviewed in the peer reviewed literature. The most prominent and well-developed approach for integrating timber management with old-growth forest conservation in moist forest zones may be the Washington Department of Natural Resources Habitat Conservation Plan for state trust lands (<http://www.dnr.wa.gov/programs-and-services/forest-resources/habitat-conservation-state-trust-lands>), which has been implemented across more than a million acres of state and private land with the goal of maintaining old-growth forest species and providing sustainable levels of timber production. It is based on maintaining a mosaic and network of patches of old-growth and mature forest structure for terrestrial and aquatic species.

Until more research is done, including field-based tests and monitoring, there is little debate that the best way to conserve and maximize old-growth values in moist forests is to exclude intensive timber management activities (e.g., clearcutting and plantation establishment) in old growth. This was the direction of the NWFP when it placed 80 percent of the remaining old-growth forest patches on federal lands into LSRs. The remaining 20 percent was placed into matrix lands—open to timber management, using innovative silviculture (e.g., ecological forestry) according to approved plans (USDA FS 1994) (fig. 3-29). The suggested management approach of the NWFP in the matrix lands, along with experiments in adaptive management areas, had they been implemented, would have enabled scientists and managers to learn about tradeoffs associated with managing for timber and ecosystem values at patch levels. As it stands now, we know relatively little about these tradeoffs because of a lack of implemented studies—the exceptions being the simulation studies of Cissel et al. (1999) and Spies et al. (2007) for moist forests.

Gary Rost

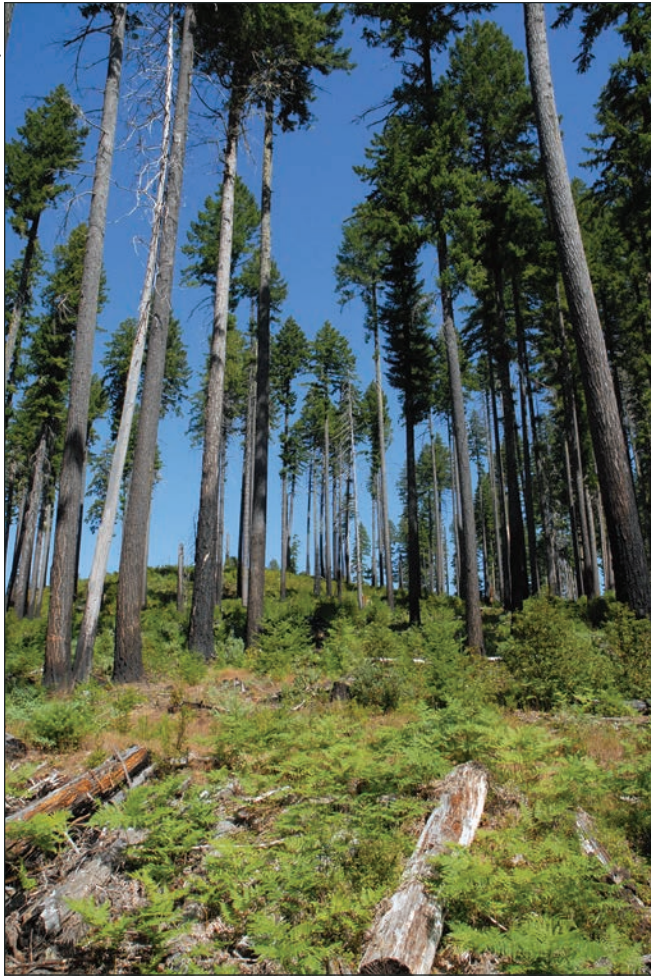


Figure 3-29—Example of a green tree retention unit created on Central Cascades Adaptive Management Area on the Willamette National Forest. The goal was to emulate stand structure created by a partial stand-replacement fire and produce timber.

Dry forests—

Clearcutting and plantation management are also not compatible with management for ecological integrity and resilience in dry forests (Franklin et al. 2013). However, other forms of management (table 3-5) may be needed to promote ecological integrity and resilience to climate change as characterized by the 2012 planning rule. Restoration thinning and prescribed fire in forests containing trees over 80 years would promote resistance and resilience to fire and climate change both within and outside LSRs. Some of these restoration activities could provide economically valuable wood products. Areas of dense old, multilayered

forests and owl habitat can still be provided at landscape scales, but they would be more dynamic, shaped by fire and other natural disturbance agents. A holistic landscape-restoration strategy has been proposed for the 4-million-ac (~1.6-million-ha) Okanogan-Wenatchee National Forests. The plan seeks to use a variety of vegetation and fuels management techniques to reduce wildfire vulnerability across the landscape, including in areas adjacent to owl habitats in “critical habitat” (USDI 2012), and to restore fire regimes, to increase resilience to climate change. More research is needed in these dry dynamic landscapes to develop and evaluate approaches for achieving both ecosystem and focal species goals (see chapter 12).

Reserves in Dynamic Ecosystems

Concepts—

Protected areas or reserves are a well-established strategy for conserving biodiversity by limiting human activities (e.g., intensive timber management and development) that are incompatible with certain ecological objectives (Lindenmayer and Franklin 2002). However, the efficacy of reserves as the **sole** basis for conserving biodiversity has been challenged by a number of authors (e.g., Fischer et al. 2006, Lindenmayer and Franklin 2002). These challenges relate to several concerns: (1) biodiversity reserves cover only a small part of the Earth’s land surface (e.g., <6 percent) (Fischer et al. 2006); (2) globally, the majority of reserves tend to be small in area (tens to <25,000 ac [~10 000 ha]) (Bengtsson et al. 2003), making them susceptible to impacts from large rare events (e.g., fire and wind) and influences (e.g., invasive species and human activities) from outside the reserves; and (3) most reserves are static and climate change may shift environments and species distributions to unreserved areas (Carroll et al. 2010).

A fundamental design recommendation for reserves is that they should be considerably larger than the largest disturbance patch size if they are to maintain habitat and populations of the most extinction-prone species (Pickett and Thompson 1978). This concept, which is known as “minimum dynamic area” requires knowledge of patch size distributions of infrequent disturbances that would

be considered incompatible with conservation goals. Such knowledge is lacking for most disturbance regimes, especially under climate change, but it can be estimated using historical information and power laws (e.g., see Moritz et al. 2005).

The reserve design of the NWFP was a late-successional forest coarse-filter strategy that was based heavily on the needs of the northern spotted owl and leveraging existing reserves (e.g., wilderness) where appropriate. The reserve strategy attempted to mitigate the shortcomings of other reserve-based conservation approaches by increasing the proportion of reserves on federal lands to 80 percent (including congressional reserves, LSRs, riparian reserves, and administratively withdrawn areas). The congressional reserves and LSRs represented 28.1 percent (15.8 million ac or 6.4 million ha) of all public and private forest lands in the NWFP area, which made it one of largest reserve systems for any temperate forested ecoregion in the world. The individual LSRs under the NWFP are also relatively large. For example, 47 percent of the individual LSRs are larger than 25,000 ac (~10 000 ha), and three are larger than 250,000 ac (~100 000 ha) (fig. 3-30). Compared to the size of recent patches of high-severity fire (fig. 3-30), the sizes of the reserves are typically larger, although many (>120) LSRs are relatively small (e.g., <25,000 ac) and could be completely burned in a single fire event with large patches of high-severity fire (e.g., 25,000 ac).

The NWFP hypothesis was that a large network of reserves well-distributed across the region would be resilient to expected losses from wildfire over a period of 100 years. While losses were expected, there was no estimate of how much loss would be too much for the goals of the Plan. The reserve patch size and fire-size analysis indicated that, for the most part, the reserves have been large enough and numerous enough to absorb many recent large fires with limited loss of OGSi 80 or OGSi 200 forests in many but not all provinces. However, it must be remembered that recent historical fire history trends will not necessarily continue in the future. Given current trends, it is likely that one to several of the LSRs, especially the small ones, will experience significant losses of OGSi to large patches of

high-severity fire over the next few decades. The infrequent fire regimes of the area have the potential to burn with very large fires, and it remains to be seen if the sizes and numbers of LSRs are sufficient to meet the goals of the Plan under climate change or other threats (e.g., invasive species).

The effectiveness of the NWFP regional reserve-matrix strategy in meeting ecological goals under current and future climate has received relatively little attention in scientific literature. The limited studies suggest that the existing network and standards and management guidelines of reserves, which spans a wide range of elevations and 10 degrees of latitude, will provide a good (but not necessarily optimal) foundation for meeting conservation goals in moist forest zones under a changing climate (Carroll et al. 2010, Spies et al. 2010). However, other than Carroll et al. (2010) and Carrol (2010), no quantitative studies of the NWFP reserve network or the regional plan as a whole have been conducted outside of efforts focused on conservation planning for the northern spotted owl (USDI 2012, USFWS 2008). In general, the science of regional conservation planning and assessment, including evaluation of reserve networks, has advanced considerably since the NWFP was implemented. For example, Margules and Pressey (2000) presented a systematic approach for evaluating reserve network plans and implementation and Virkkala et al. (2013) demonstrated a methodology to evaluate the viability of reserve networks for protecting biodiversity in the face of climate change in Finland. According to Carroll (2010), “Rigorous assessment of the implications of climate change for focal species requires development of dynamic vegetation models that incorporate effects of competitor species and altered disturbance regimes.” In his assessment of the resiliency of the NWFP reserve network for multispecies conservation under climate change, Carroll (2010) did not address how wildfire might affect the conservation goals of the Plan, which is a significant concern. The development of regional-scale vegetation and species occurrence data and vegetation dynamics models, including spatial fire landscape models (e.g., Scheller et al. 2011, Spies et al. 2017), in recent years suggests that a more rigorous and comprehensive evaluation of the NWFP regional strategy would now be possible.

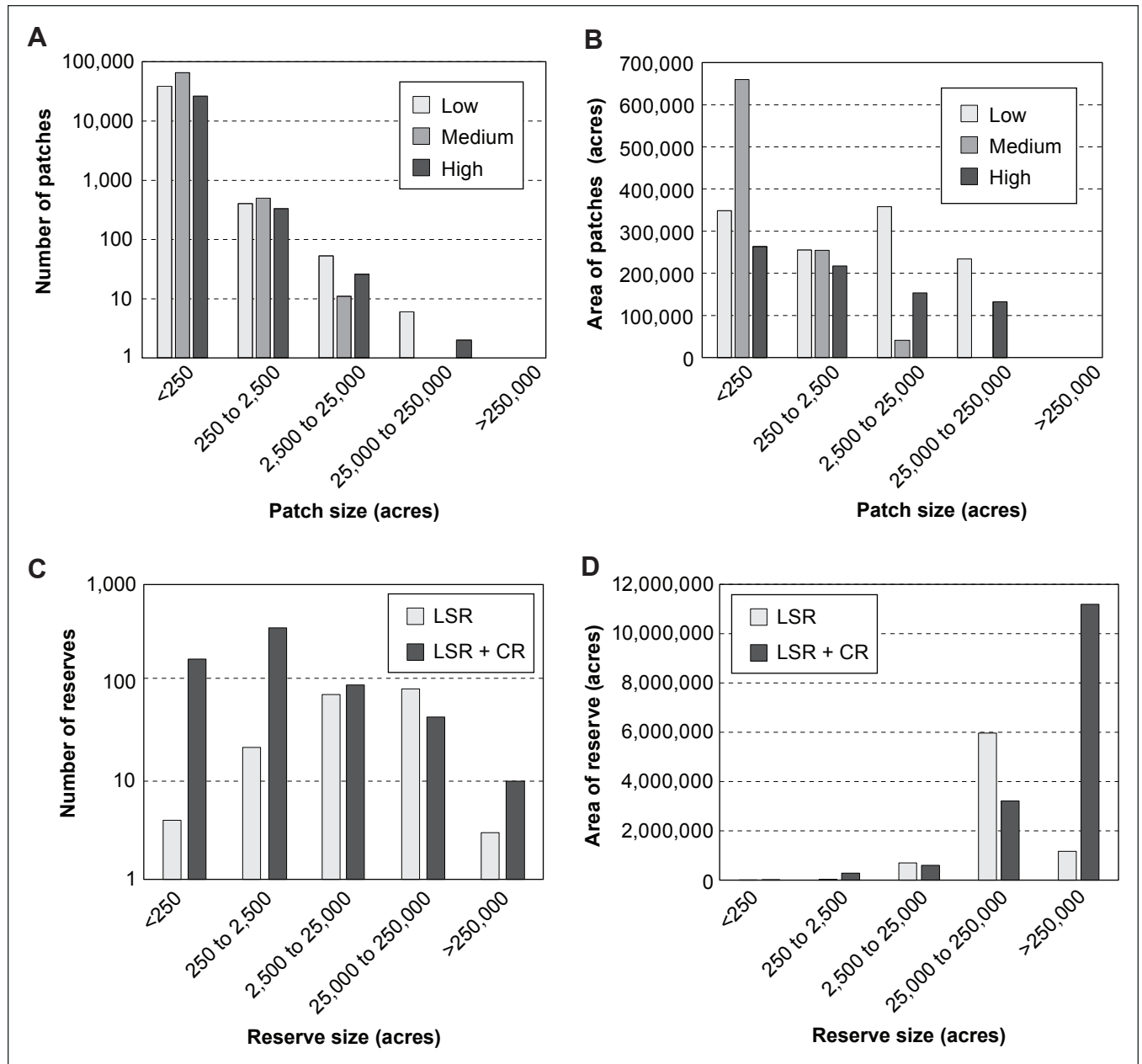


Figure 3-30—Patch distributions of recent fire (2000–2012) and sizes of Northwest Forest Plan (NWFP) reserves: (A) frequency distribution of number patches by fire severity and area class, (B) fire-severity patch sizes by area class, (C) frequency distribution of number of NWFP reserves (late-successional reserves [LSRs] alone and LSRs plus congressionally designated reserves [CRs]) by area, and (D) area of reserves by size class.

Reserves or protected areas are not necessarily areas where all human activities are excluded or are inconsistent with ecological conservation goals (Soule 1985). There are many types of protected areas with different degrees of human activity permitted (Spies 2006), including recreation areas, management allocations for degree and type of veg-

etation manipulation, invasive species removal areas, and fire management (prescribed fire or fire suppression) areas (Pressey et al. 2007). In most cases, including the NWFP standards and guidelines, biodiversity reserves permit and encourage restoration activities that further the species and ecosystem goals of the reserved area. For example, the

NWFP indicated that restoration activities within reserves were needed for both moist and dry forests (USDA FS 1994) in plantations in wetter and drier forests, and in older forests in fire-frequent regimes where forest structure and composition has been altered by fire exclusion and logging of older trees.

Wildfire and fire exclusion both pose serious challenges and dilemmas to managers seeking to conserve biodiversity using reserves or any other conservation approach (Driscoll et al. 2010, Fischer et al. 2006, Spies et al. 2012). This observation may seem contradictory or ironic, but it is the reality when conserving fire-prone forests in the Western United States. The multifaceted nature of wildfire makes it difficult to find a conservation and management “sweet spot.” For example, fire is a vital and dynamic ecological process that maintains some communities, renews other communities, and increases plant growth and productivity (Ahlgren and Ahlgren 1960), but it also kills trees and destroys valued habitats, forest resources, and human infrastructure and lives (DellaSala and Hanson 2015). The assumption that reserves could conserve habitat for the northern spotted owl and other old-growth-associated species in dynamic ecosystems subject to fire, succession, and climate change was a major hypothesis of the NWFP. We examine this hypothesis below using data from the monitoring program (Davis et al. 2015) and new scientific knowledge.

Is the reserve system meeting the original goals of the Northwest Forest Plan?—

The reserve-matrix system was intended to protect and recover older forests in response to threats from logging and natural disturbances that destroy older dense forests. The general goal was to increase the amount of late-successional/old-growth forest in the reserves to recover toward levels that were present before extensive logging began on federal lands in the early 1950s. No specific targets for the future proportion of late successional/old growth in reserves were made in terms of HRV at the LSR scale, but the expectation was the amount of late successional/old growth in general on federal land would approach 60 percent over 100 years (Davis et al. 2015), including expected losses owing to wildfire. Dry zone forests were included in this

rough estimate though the likelihood of achieving this goal was considered to be lower in dry forest zones than in moist forest zones (FEMAT 1993: fig. IV-3). It was expected that millions of acres of younger forests and plantations would eventually grow into an old-growth condition making up for any losses to wildfire or other disturbance agents. Between 1993 and 2012, disturbances, including wildfire and planned timber harvest, have reduced older forest (OGSI 80) area by 6.0 percent and OGSI 200 by 7.6 percent (Davis et al. 2015). Wildfire has accounted for the greatest reduction in older forest: annualized losses to wildfire were 0.22 percent and 0.28 percent for OGSI 80 and OGSI 200, respectively. In comparison, FEMAT (1993: IV-55) assumed that the annualized percentage of high-severity fire in reserves across all provinces would be about 0.25 percent over the first 50 years. At the scale of the entire NWFP, the losses from wildfire approximated expectations (Davis et al. 2015, FEMAT 1993) across the entire plan area (no projected losses were made by province), but losses from timber harvest were much less than planned.

The rates of change in OGSI 80 were not uniform across the physiographic provinces. Provinces with net declines that were higher than the regional averages are in order: Oregon Klamath (-9.9 percent), Oregon Western Cascades (-4.9 percent), and California Klamath (-4.1 percent).¹⁵ Net change in OGSI 80 in eastern Oregon and eastern Washington Cascades, where wildfires have been relatively common (Davis et al. 2015), (table 3-6) were at or less than the regional average (e.g., -2.8 and -2.2 percent). While losses to fire and other disturbances get much attention, monitoring reveals that forest dynamics are also about succession, which will always at least partially offset losses: 757,900 ac (306 842 ha) of loss to disturbance appears to have been partially offset by 396,100 ac (160 364 ha) of gain from succession (Davis et al. 2015) (table 3-6). If losses from timber harvest are excluded (to highlight the role of natural disturbance agents), those losses (609,800 ac

¹⁵ For OGSI 200, more physiographic provinces exceeded the regional average of -2.8 percent net change: Washington western lowlands = 7.0 percent; Oregon western Cascades = - 6.0 percent; Oregon Klamath = -10 percent; California Coast Range = -3.0 percent; California Klamath = -7.9 percent. table 3-8. From Davis et al. (2015).

[246 882 ha]) from all disturbance agents drop to 4.7 percent from 6.0 percent as gains from succession replaced about 65 percent of those over 20 years. Some provinces (e.g., Washington western Cascades, Oregon Coast Range, California Coast Range, and California Cascades) actually showed a net increase in OGSi 80 on federal lands (Davis et al. 2015) (table 3-6).

At the scale of individual LSRs, the range in net changes in OGSi 200 forests ranged widely (from -52 to >100 percent) (fig. 3-31) as would be expected for relatively small land areas. Most of the LSRs with the largest net changes are relatively small in area, with the exception of those in the Klamath regions of Oregon and California, where large patches of high-severity fire have occurred in the past 20 years. Three reserves in the eastern Cascades of Washington show relatively high rates of net loss, but all of these are relatively small reserves and the total net change in this province is about the regional average. The majority of the LSRs show little or no change. In general, large reserves have been more stable than smaller ones (fig. 3-32), which was why some of the largest reserves were drawn in fire-prone areas during FEMAT.

If rates of loss of dense old-growth were much higher, LSR function would be threatened because they were designed to be dominated by dense, complex older forests and serve as stepping stones for connectivity of old-forest species across the NWFP area. The loss of large areas of older forest in one or more of these reserves could challenge the connectivity design functions; however, no research has investigated the degree of change in the reserve network that might affect its overall function. At the recent rate of net change (-0.15 percent per year) (Davis et al. 2015) (table 3-6), the original matrix and reserve system appears sufficient to maintain areas of OGSi 80 at a regional scale, with greater declines (-0.23 percent per year) in the dry forests. This is especially so if it is assumed that the rate of ingrowth into denser older forest types will increase dramatically in coming decades as large areas of younger plantations and early 20th century wildfire-initiated stands begin to reach the age and structure where old-forests characteristics appear (Davis et al. 2015). However, the current trends may not hold given that fire activity is projected to

increase across the NWFP area. With increasing drought fire sizes, including patches of high-severity, fire may increase (Reilly et al. 2017). Projections of the amount of increase in area or size of fires differ considerably across the NWFP area and among studies. For example, Stavros et al. (2014) found that the probability of very large fires will increase for Oregon and Washington, but increases would be minor in northern California. Littell et al. (2010) found that area burned is likely to increase by two to three times for Washington. Ager et al. (2017) modeled increases in fire and their effect on northern spotted owl habitat and fire regimes in the eastern Cascades of Oregon. They found that increases of two to three times in rates of wildfire would reduce spotted owl habitat by 25 to 40 percent within 30 years. They also found, however, that as fire increased, negative feedbacks on fire area and intensity occurred, suggesting that as fire increases, fuel limitations would affect future fire behavior. Most climate projection studies focus on area burned and not on severity and do not include fire feedbacks. Studies are needed to evaluate how climate change and fire might affect the LSR network conservation goals for different network configurations and management guidelines (e.g., levels and types of restoration).

While understanding annual rates of change in LSRs during the past 23 years is important to assessing Plan outcomes, it is also important to acknowledge that annual rates of disturbance or loss over short periods of time (e.g., 23 years) have limited value in the infrequent, high-severity regimes and across all regimes given climate change. Large fire or wind disturbances may be rare or episodic in infrequent regimes but can strongly control landscape dynamics and leave legacies that persist for centuries or longer (Foster et al. 1998, Spies and Franklin 1989). The real test of the reserve network can only be done over very long periods of time, and ultimately managers will have to be prepared for surprises and inevitable large events. Knowledge of trends and annual rates of change are useful but are of limited value for predicting the future in ecosystems, where fire, wind, volcanic eruptions, earthquakes, or invasive species can change forests rapidly over large areas.

The “losses” of late-successional/old-growth structure in reserves to fire may be a loss from the perspective of

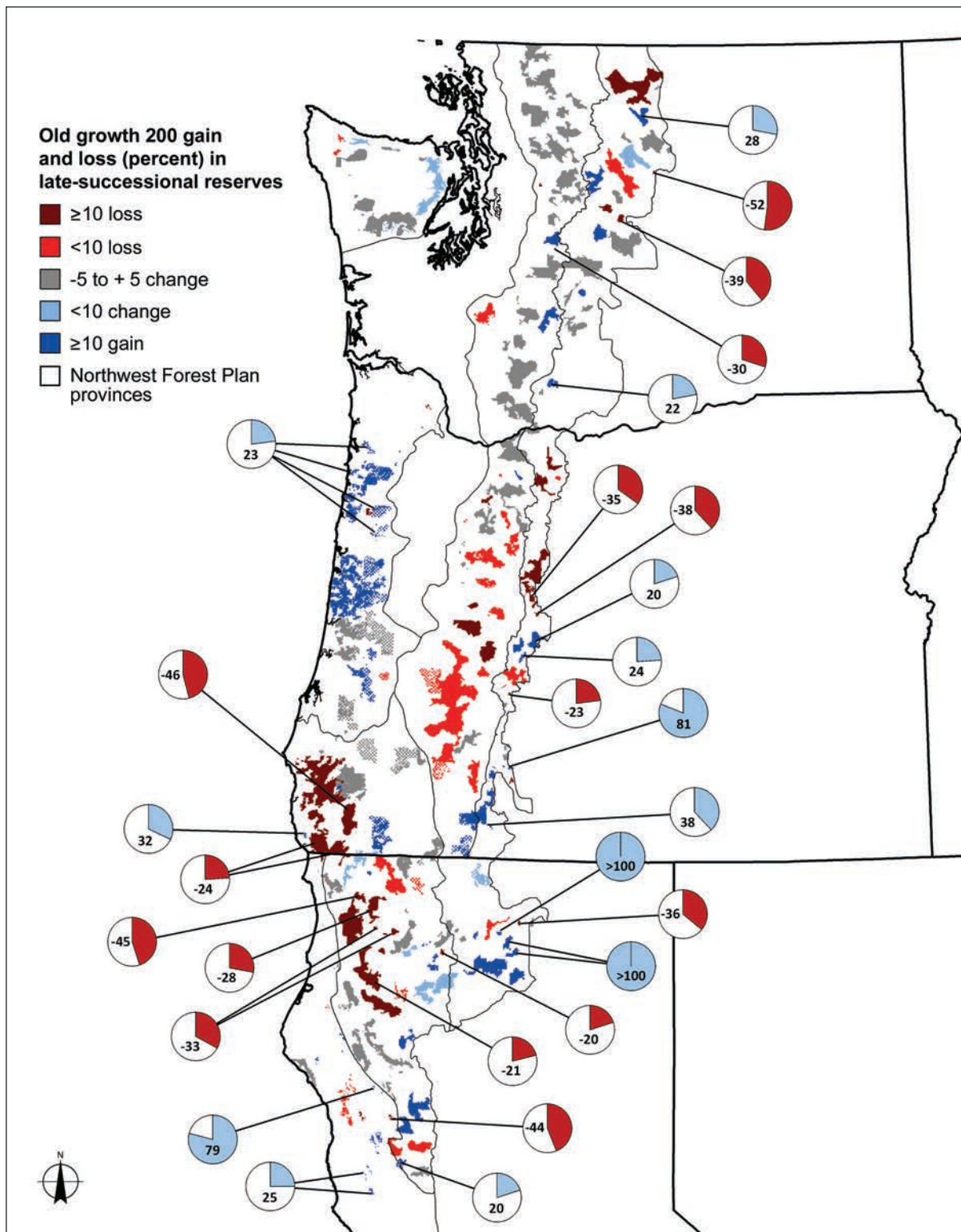


Figure 3-31—Map of 192 late-successional reserves (LSRs) in the Northwest Forest Plan area showing percentage of net change (gain or loss) in old-growth structure index (OGSI) 200 from 1993 to 2012. The LSRs are color coded by degree of gain (blue) or loss (red). The LSRs with little net change are shown in gray. Pie charts only show LSRs with greater than 20 percent net change (e.g., annualized rate of 1 percent), either gains or losses. Colored sections and numbers in pie charts indicate percentage of OGSI 200 in LSRs that was gained or lost. Percentages can exceed 100 percent where gains occur. Data based on Davis et al. 2015.

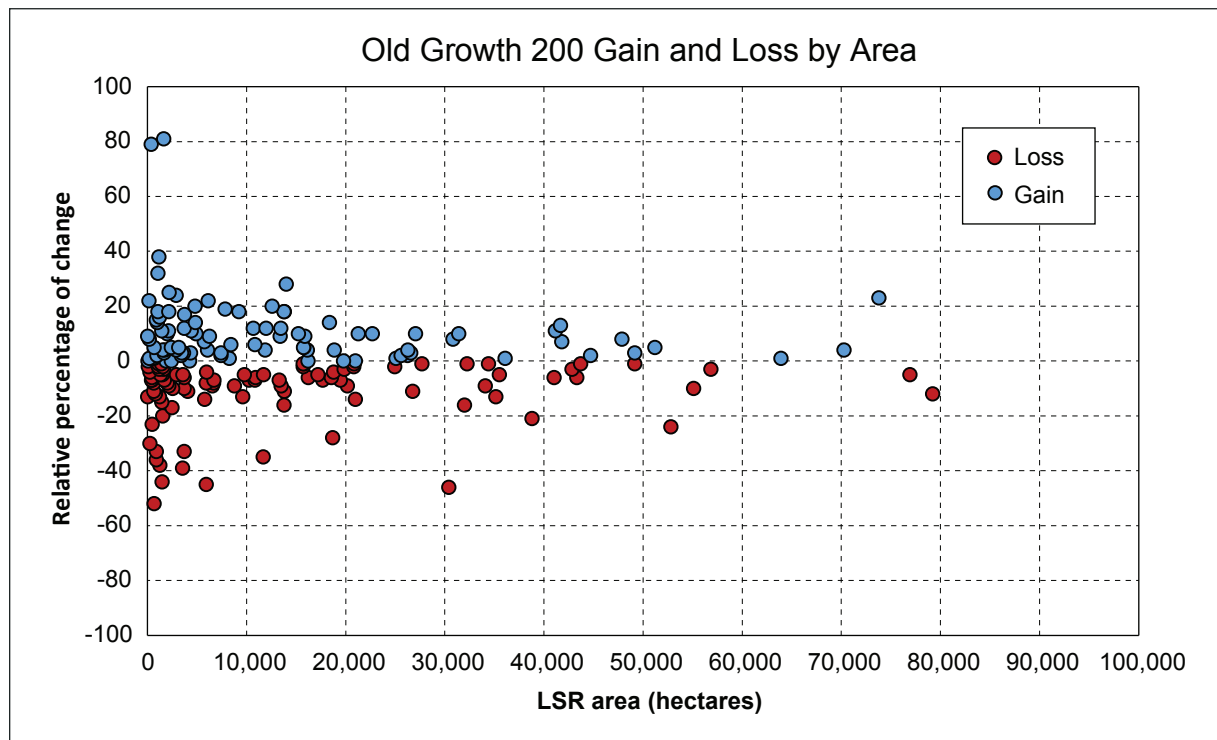


Figure 3-32—Relative change in old-growth structure index (OGSI) 200 in reserves between 1993 and 2013 in relation to late-successional reserve (LSR) size. Reserves smaller than 50,000 ac (20 224 ha) tended to show more change than larger reserves.

conservation of dense older forests, but they do not necessarily represent a loss from a broader biodiversity perspective (e.g., ecosystem integrity), especially where those fires burn at lower severities and thin out understories, leaving lower densities of fire-tolerant species. This is especially the case in dry forest landscapes, where open old growth and mosaics of old and early successional were characteristic. However, as mentioned above, the OGSI thresholds in frequent and very frequent fire regimes were based on plots from existing older forests that have been subject to fire exclusion and succession that would have increased stand density, layering, and amounts of shade-tolerant and fire-intolerant species. Hence, the reference conditions for older forests do not typically represent the older forest structure and composition types that developed under more frequent fire regimes. Large fires such as the 2002 Biscuit Fire often have less than 20 percent of their total area in high-severity patches and have large areas of historically moderate to low severity (Reilly et al. 2017, Thompson and Spies 2009). Lower and moderate-severity wildfire shifts

stands from dense old forests to more open old forests (i.e., thins out understories but leaves many of the older fire-tolerant trees) that were characteristic of forest structure and composition under frequent fire regimes (Kane et al. 2013). However, monitoring and inventory definitions for these more open older forest types do not exist (Spies et al. 2006b, Taylor and Skinner 1998) and were not applied in the monitoring program.¹⁶ Reilly and Spies (2015) classify forest structure in the NWFP area using existing inventory plots and identify conditions that may approximate the historical structure of more open old-growth forests. The lack of focus on open types of old growth was probably the result of the original emphasis of the NWFP on dense late-successional old-growth forest habitats of the western Cascades of Oregon and Washington which are associated with northern spotted owl and other species.

¹⁶ The OGSI for pine types was based solely on density of large live trees, which may approximate historical amounts, but they do not include canopy cover and layering.

Concerns—

Although general trends revealed by monitoring at the regional scale appear consistent with NWFP goals and expectations, there are other less obvious trends that may be cause for concern in dry forests. First, in the Klamath Mountains and other regions, where chaparral and other shrub species are an important component of the vegetation, an increase in size and frequency of high-severity fire patches can lead to more extensive areas of early-seral or chaparral vegetation that can become a semipermanent landscape feature (Lauvaux et al. 2016, Tepley et al. 2017). It is not clear how much of this type of change would be desirable to meet ecological or social goals, and management may be needed to promote succession toward trees that are resistant to fire and climate change. On the other hand, Donato et al. (2011) suggested that low-density conifer regeneration in the presence of hardwoods and shrubs is an alternative successional pathway to promote early development of old, complex old-forest structure.

Very large patches of high-severity fire also occur in other low- and mixed-severity forest types in the NWFP area (Hessburg et al. 2016) with the possibility that recovery to forest is slowed or precluded as a result of lack of conifer seed rain (Dodson and Root 2013). This is especially in large reburn patches and may require planting to mitigate these effects (see restoration section below). The degree to which large patches of high-severity fire are slowing forest succession after recent large fires in the NWFP area is not known. On the other hand, relatively large patches of high-severity fire can result in areas of nonforest vegetation (e.g., grasslands and shrub lands) that were more common in the past than today in many dry forest landscapes (figs. 21, 22, and 26).

A second concern in dry forests is that older forests and landscapes in reserves and outside of reserves are slowly transitioning to conditions characterized by denser forests, more shade-tolerant species, buffered microclimate (less wind and shaded and cooler forest), and less flammable fuel beds. Thus, they become less likely to burn under low to moderate weather conditions and more likely to burn

under high-severity conditions. Assuming continued fire suppression (Calkin et al. 2015, Stephens and Ruth 2005) and increased warming, the forests of the reserves in mixed- and low-severity regimes will continue to change in ways that do not support the historical dynamics of these forest types.

On balance, the science reveals that fire-dependent forests in LSRs are continuing to be squeezed into altered states and dynamics by two forces: (1) succession toward historically unprecedented structure, composition that affects biodiversity, landscape structure (e.g., larger more connected dense forest patches), and ecosystem function in absence of fire; and (2) a shift toward much less frequent but higher severity fire regimes as a result of fire exclusion, climate change, and changes in vegetation, including increased fuel loading and contagion. Losses of old growth and owl habitat to high-severity fire are the focus of the current monitoring reports and strategies, and succession toward dense forests with shade-tolerant species (e.g., owl habitat) is typically considered a positive outcome relative to the goals of the NWFP. However, within the dry forest zone and some drier parts of the moist forest zone, these types of forests are not a desirable outcome if the goal is ecological integrity based on frequent fire, open fire-resilient old growth, diverse successional conditions, and disturbance processes and landscape dynamics that maintain resilience and a full complement of native biodiversity. Landscape-scale research and strategies are needed to find options that provide for late-successional species while improving the overall resilience and functions of dry forests (Hessburg et al. 2016; Sollmann et al. 2016; Spies et al. 2006, 2017). Frameworks based on knowledge of ecological history or on NRVs or the HRV and departure from those references (Haugo et al. 2015) could be used to guide development and implementation of alternative approaches for dry forests to meet the goals of the NWFP and the 2012 planning rule. For more discussion of reserves and possible alternatives to static reserves, see chapter 12.

Connectivity and Fragmentation

Connectivity and fragmentation of mature and old-growth forests were important considerations in developing the NWFP (FEMAT 1993). The spatial pattern, size, and isolation of habitat patches of older forests can affect species richness, population dynamics, as well as the spread of fire and other disturbances. Davis et al. (2015) found that older forests on federal lands have become slightly more fragmented by disturbance over the period of the Plan. However, this analysis only takes into account late-successional and old-growth conditions and does not factor in changing connectivity relations over the remainder of the landscape, which may be the larger story. Consequently, it is not clear what the cumulative ecological effects (e.g., species richness, microclimate) of spatial pattern changes have been as a result of disturbance and succession over the past 20 years. It is now recognized that the ecological effects of spatial pattern of vegetation types and successional stages (e.g., edge effects, patch size effects, connectivity) differ with species and processes and are difficult to generalize about using a coarse-filter approach (Betts et al. 2014). Cushman et al. (2008) found that maps of existing forest cover types and successional stages in the Oregon Coast Range were not effective in estimating abundances of breeding birds and cautioned that maps based only on coarse vegetation classes may not provide a good metric of species abundance. If maps of vegetation types have limitations for conservation, then the analysis of spatial pattern is also likely to have limited value for predicting community or species outcomes. Fahrig (2013) has recently hypothesized that habitat amount is a better predictor of species richness than patch size and isolation for community-scale (i.e., coarse-filter) approaches to conservation. However, this does not mean that patch size, isolation, and connectivity are not important components of habitat at the scale of individual species (e.g., fine filter) or for key processes. The implication for the NWFP is that patch size and connectivity concerns are best dealt with at the individual-species scale (e.g., northern spotted owl, carnivores) or processes (e.g., fire spread through landscapes). The question of connectivity for late

successional/old growth as a coarse-filter metric and even use of maps of late successional/old growth to represent “habitat” in general (e.g., concern of Cushman et al. 2008) is an area of uncertainty and needs research. See chapter 12 for more discussion of regional-scale issues.

Restoration Approaches

Here we address our scientific understanding of management actions that could be used to achieve goals for ecosystem restoration, especially those related to successional diversity and natural disturbance regime processes. We use a loose definition of restoration given that climate, landscape, and species changes make it from difficult to impossible or perhaps undesirable to really restore the structure, composition, and function of past ecosystems (Spies et al., chapter 12). Ecological restoration has been defined as “*the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed*” (<http://www.ser.org/resources/resources-detail-view/ser-international-primer-on-ecological-restoration>). Despite the limitations of restoration, management can promote resilience of ecosystems to fire or climate change or increase vegetation diversity that has been lost as a result of management actions such as timber management or fire suppression. Restoration may be able to promote some of the features of the pre-Euro-American period (e.g., dead wood, large fire-resistant trees, or multistoried old-growth habitats), but ecosystems may not have the same overall structure and function (or even fall within their historical ranges) as those of the pre-Euro-American period. We address these management actions by forest zone and disturbance regime, acknowledging that these ecological management approaches may be similar across regimes. Numerous authors have addressed restoration needs specified in the NWFP (Baker 2012; Franklin and Johnson 2012; Franklin et al. 2008, 2013; Haugo et al. 2015; Hessburg et al. 2016; North et al. 2009, 2012; Stephens et al. 2009; Stine et al. 2014). In general, these restoration needs are to restore disturbance processes (e.g., fire) and longer times for natural succession to operate without disturbance (Haugo et al. 2015) as young forests develop following logging (table 3-5).

Moist forests—

Stand scales—Forest plantations are the primary focal point of restoration in these forests. Approaches to restoring old-growth forest conditions in plantations include:

- Passive management—increasing the amount of older forests by electing to simply allow younger postlogging forests to naturally progress, through growth and mortality to older life stages (Haugo et al. 2015).
- Active management—using variable-density thinning (restoration thinning) (Carey 2003, Churchill et al. 2013, Haugo et al. 2015, Muir et al. 2002) to increase structural and compositional diversity in unnaturally uniform plantations that reduced typical shrub and herb layers and accelerate development of future mature and old-forest structures (figs. 3-33 and 3-34).

Currently, the most common approaches are to allow younger stands to age and mature on their own and to use variable-density thinnings (i.e., restoration thinning) to increase habitat diversity within uniform plantations (especially 30- to 80-year-old stands, where thinning is typically profitable) and thus accelerate the development of older forest structure and composition (Carey 2003) (figs. 3-33 through 3-35). They can also be used to promote elk habitat, huckleberries, and other species associated with forest openings (chapter 11). While restoration thinning is a relatively new practice for ecological goals, the effects of standard thinning (Tappeiner et al. 2007) on tree growth and mortality in regular-spaced plantations are relatively well known. For example, growth-growing stock relationships for Douglas-fir suggest minor differences in stand volume growth over a range of residual densities (Marshall and Curtis 2002), which provides some flexibility in terms of thinning prescriptions (Dodson et al. 2012). However, extremely low residual densities and gap creation obviously lead to lower stand-level tree growth. However, where stand-level foliage biomass is concerned (which is important for tree growth and litter production), thinning can stimulate growth of foliage biomass on a branch and tree scale, which may not be a desirable outcome from a restoration perspective where reducing canopy fuels is a goal (Ritchie et al. 2013a). Decreases in stand growth owing to low tree numbers are partially offset by better growth of residual

trees (Dodson et al. 2012), and by establishment and growth of regenerating trees.

Given the recency of restoration thinning practices and studies, our understanding of how this practice affects older forest development is based on only short-term results (typically less than 20 years) (Poage and Anderson 2007). To understand possible ecological effects, we extrapolate from the many studies of standard thinning operations, which suggest that such approaches would not produce many of the outcomes associated with old-growth forests (e.g., spatial heterogeneity, large dead trees, compositional diversity) in the short term (up to 50 years), other than larger diameter trees (Anderson and Ronnenberg 2013).

In contrast to standard thinning operations, restoration thinning includes preferentially retaining minority species and creating a wider range of density conditions from open gaps to unthinned patches of various sizes (Carey 2003, Davis et al. 2007, Neill and Puettmann 2013). This appears to be key to increasing the heterogeneity in thinned stands and accelerating development of late-successional elements (Anderson and Ronnenberg 2013, Cissel et al. 2006, Poage and Anderson 2007). Also, the initial responses to variable-density thinning treatments suggest that not all structural components and processes react in synchrony (Puettmann et al. 2016). For example, one study found that after a brief delay, likely due to increases in crown size (Ruzicka et al. 2014), restoration thinning led to an increase in average-tree-diameter growth. However, larger trees, which would likely become the dominant trees that are the major features of an old-growth stand, barely responded unless they were growing in extremely low densities, e.g., adjacent to gaps (Davis et al. 2007, Dodson et al. 2012). Also, diameter growth responded rather quickly within the first 5 years, while changes in other vegetation components were slower or delayed, such as in crown structures (Davis et al. 2007, Seidel et al. 2016) or bark furrows (Sheridan et al. 2013). That study also found that other vegetation components followed a counterproductive trend relative to late-successional/old-growth biodiversity goals. For example, the shrub layer was knocked down during harvesting operations and did not recover to

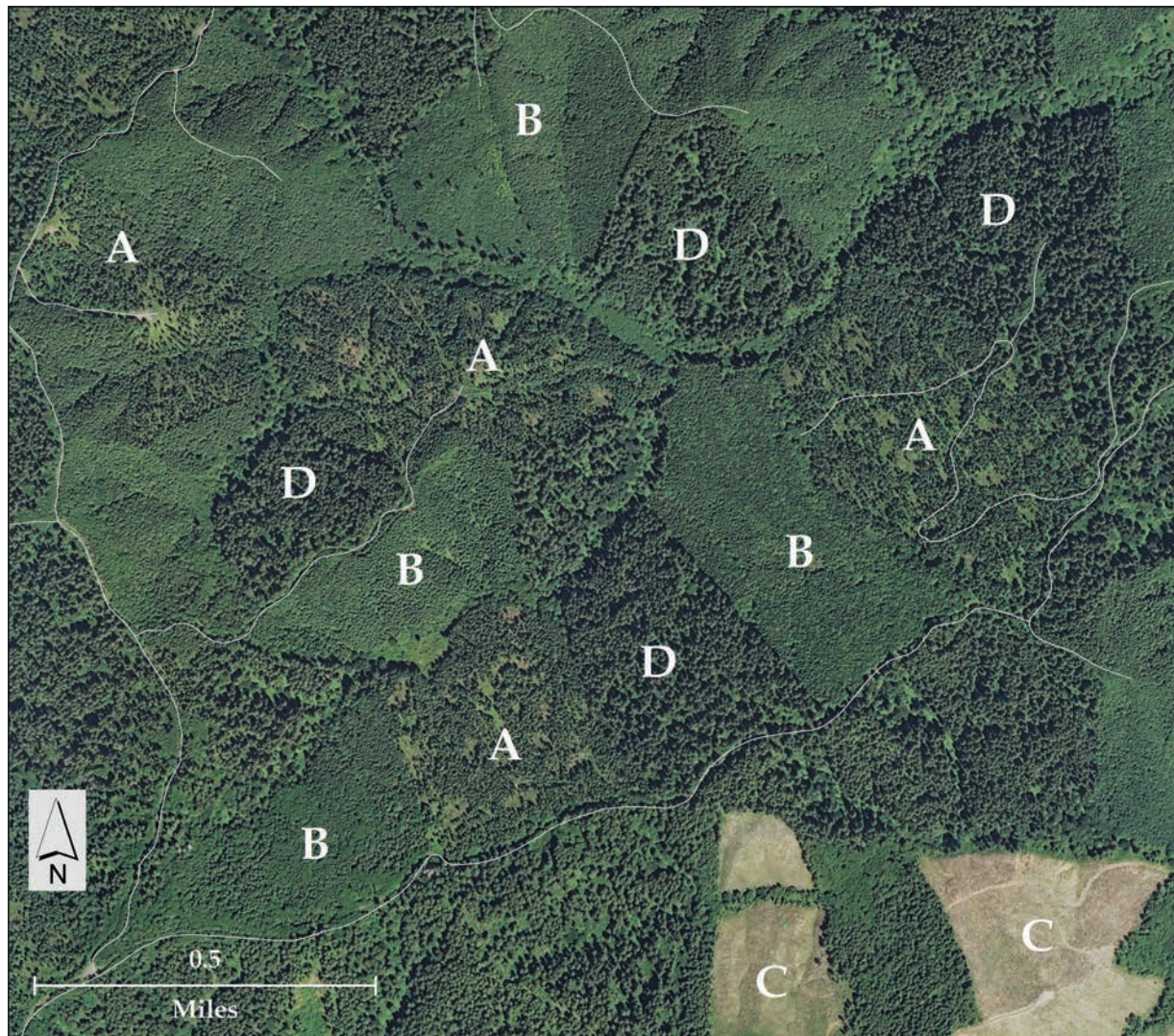


Figure 3-33—Aerial image from 2011 of management units and unmanaged stands in an area of late-successional and riparian reserves and matrix allocation on the Siuslaw National Forest, and private lands in the Oregon Coast Range: (A) plantations treated with variable density thinning, (B) uniform plantations that have not been thinned (these plantations are younger than those that have been treated), (C) recent clearcuts on private land, and (D) older naturally regenerated forests that have not been managed. Note areas of hardwood and shrub gaps in the older conifer forests that occur in root rot (*Phellinus sulphurascens* pockets). Roads are indicated by white lines. From Oregon Explorer Natural Resources Digital Library.

preharvest levels within the first decade (Puettmann et al. 2013). Also, the understory vegetation composition shifted toward a higher component of early-successional species. This trend started to reverse within a decade (Ares et al. 2009, 2010) but was still detected 20 years after a precommercial thinning (Lindh and Muir 2004). Exotic species remained a minor component after restoration thinning and showed a similar trend of decline after a decade. With little postharvest mortality after thinning, snag recruitment

was reduced 11 years after thinning (the time of the last measurement) (Dodson et al. 2012) and likely in the longer term as well (Garman et al. 2003, Pollock and Beechie 2014). This trend can be counteracted by creating snags (Lewis 1998); however, if this is done during restoration thinning, these snags would be smaller and shorter than in older stands. Alternatively, leaving untreated patches of high tree density ensured that competition-related mortality continued, although this led to snags at the

Sam Chan

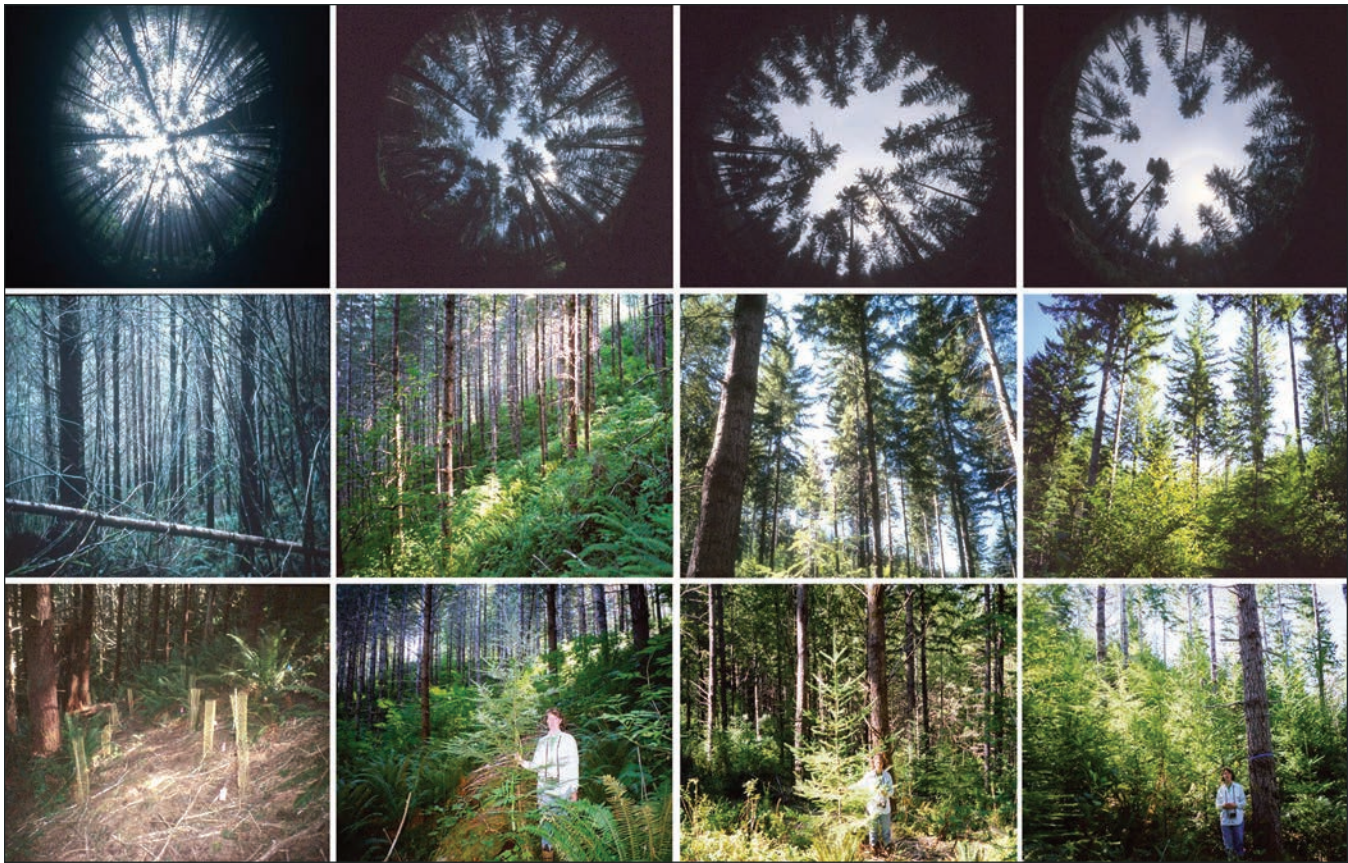


Figure 3-34—Canopy (fisheye) and understory photographs of unthinned and thinned 30 to 40 year old plantations of Douglas-fir on the Siuslaw National Forest. Densities of four stands from left to right: unthinned; 100 trees/acre; 60 trees/acre; and 30 trees/acre.

smaller end of the size distribution (Dodson et al. 2012). Tree regeneration typically increased right after restoration treatments (Dodson et al. 2014, Kuehne and Puettmann 2008, Urgenson et al. 2013), showing three general trends. First, while stand-level differences were obvious, studies showed very high spatial variability at small spatial scales. Second, seedling establishment increases after thinnings, but densities appeared to be similar, regardless of thinning intensities. Third, seedling and sapling growth differed by species and responded to higher degrees of overstory removal (e.g., Shatford et al. 2009).

The benefits of restoration thinning relate as much or more to increasing spatial heterogeneity as to reducing density per se, as high-density patches are not uncommon in natural stands. For example, Spies and Franklin (1991) reported that stand densities (trees >2 inches [5.1 cm] diameter at breast height) in young stands (40 to 79 years old) that

regenerated naturally after wildfire in western Washington and Oregon averaged about 400 stems per acre (1,000 stems per hectare.). Some plantations 40 to 60 years old that regenerated naturally after logging (Curtis and Marshall 1986) or following clearcutting and planting can have similar densities, though plantations with much higher densities (e.g., 800 stems per acre [~2,000 stems per hectare) occur.¹⁷ In some places, natural regeneration (e.g., western hemlock) will establish itself in Douglas-fir plantations (Puettmann, personal observation) leading to extremely high tree densities. While average tree density can be high in plantations, density differences do not explain all potential differences between natural young stands and plantations. The differences are also

¹⁷ Pabst R. Personal communication. Senior faculty research assistant, College of Forestry, Oregon State University, Corvallis, OR 97331.



Google Earth™



Figure 3-35—Example of variable-density thinning from 2013, including skips and gaps (1 to 2 ac [0.40 to 0.80 ha]), in a 56-year-old plantation on the Willamette National Forest: (A) the pattern across the entire treatment area and the surrounding unthinned plantation, (B) a view from inside the thinned area, and (C) the view looking across the gap. The goal was “volume production, promotion of high-quality elk forage in the short term, while encouraging development of elk-optimal cover.”

expressed in spatial variation in density and variability of tree age and size (Tappeiner et al. 1997). The age ranges and spatial heterogeneity of trees in naturally regenerated stands may lead to greater variability in canopy differentiation than would occur in plantations where trees are the same species, the same age, and are planted with uniform spacing (Oliver and Larson 1990). A combination of tall shrubs, hardwoods, or other vegetation would have occupied much of the open growing spaces (i.e., spaces not occupied by conifer regeneration in naturally regenerated stands). The short-term effects of variable-density thinning aimed at improving longer term structural and compositional diversity may be to fragment canopies and root systems and temporally reduce habitat quality for animal, plant, or fungal species keying in on canopy and root structure (Davis and Puettman 2009, Pilz et al. 2006). This is an important issue requiring more research. Alternative ways of implementing thinning prescriptions (e.g., leaving larger unthinned areas or thinning very young stands) may actually improve conditions for lichens (Root et al. 2010) and may help to mitigate some of the short-term negative effects of discontinuous forest canopies on canopy species (Wilson and Forsman 2013).

Empirical studies are critical, but evaluating long-term and landscape-level effects of variable-density thinnings requires landscape simulation models. Traditional growth and yield models provide fairly reliable information about tree growth for more or less evenly spaced, even-aged Douglas-fir plantations (Fairweather 2004). Most models assume the absence of disturbances, but ongoing efforts include a better representation of disturbance (e.g., insects and pathogens) on tree and stand growth (Crookston and Dixon 2005). Predictions for open or irregular-spaced conditions (Lord 2005) and growth of other species are less reliable or missing (Gould et al. 2011, Kuehne et al. 2015, Weiskittel et al. 2007). Similarly, there is a broad understanding and agreement about general trends, e.g., in understory vegetation, but specific dynamics cannot be modeled with high precision because they are based on interactions of initial conditions, species traits, local environmental conditions, and stochastic events (Ares et al. 2010, Burton et al. 2014), which may vary over time (Thomas et al. 1999) and space (Burton et al. 2014, Chen et al. 1992).

In the few modeling studies (Garman et al. 2003, Pollock and Beechie 2014), thinning promoted the development of large boles, vertical diversity, and tree-species diversity over 100+ years, compared to controls. At the same time, less dead wood was produced over many decades compared to no thinning, highlighting that at least some of the early trends found in the experimental studies (e.g., Dodson et al. 2012) may last longer. As mentioned above, the negative effects of thinning on deadwood production can be countered by creating snags (Lewis 1998) or leaving cut trees on the sites where they can immediately contribute to terrestrial and ecological functions (Huff and Bailey 2009, Walter et al. 2005).

Thinning has variable effects on wildlife and plant communities. In the short term, it can increase species diversity and abundance of some species, especially those associated with more open forest conditions (Ares et al. 2009, Berger et al. 2012). This can lead to increased flowering and seed productions, i.e., provision of food resources for selected insects, mammals, or songbirds (Neill and Puettmann 2013, Wender et al. 2004). The response of songbird populations showed similar trends (Hagar et al. 2004), but responses appear to vary by species and over time (Yegorova et al. 2013). Thinning may also attract avian predators that prey on marbled murrelet (*Brachyramphus marmoratus*) nests (chapter 5).

Although general stand-level trends from restoration thinning are mostly understood, uncertainties remain. For example, vegetation development for specific locations appears partially unpredictable for several reasons, including microclimatic conditions, initial variability in plantations, and stochastic events such as seed crops, disease, and windthrow (Dodson et al. 2012, Lutz and Halpern 2006). In addition, there are important effects of thinning on residual trees, such as harvesting damage to residual trees. Damage is typically higher the more wood is harvested and often concentrated near skid trails (Han and Kellogg 2000). Through careful layout and logging (e.g., Picchio et al. 2012) and avoidance of early summer harvests, damage can be reduced to levels that are not likely to affect future health of Douglas-fir stands (Bettinger and Kellogg 1993, Kizer et al. 2011). However, other species such as western hemlock may be more affected (Hunt and Krueger 1962). With proper logging layout, techniques, and timing (e.g.,

avoidance of wet soil conditions), the impact of thinning operations on soils should be limited to removal of humus and upper soil layers (Froehlich et al. 1981). However, these impacts that are concentrated near or in skid trails are only temporary as patches of exposed soils are reinvaded quickly.¹⁸ In this context, harvesting operations that removed limbs and crowns before skidding (and in some cases limited maximum log length that could be skidded) not only scattered down wood throughout the stand, but led to lower soil damage, as well as lower damage to residual trees (K.J. Puettmann, personal observation).

In summary, ecosystem dynamics after restoration thinning are generally predictable, but specific responses can be highly variable owing to small-scale variability in environmental conditions and initial vegetation composition. In addition, other factors, such as weather patterns; seed availability; impacts of insects, diseases, and herbivores on seed or seedlings; as well as harvesting impacts as described above, suggest that restoration treatments are not likely to hit any specific target perfectly in terms of vegetation conditions and dynamics. Instead, restoration efforts may be better off acknowledging these inherent uncertainties by setting structural goals that allow for a range of conditions; e.g., between 10 and 30 percent of the restored area should have regeneration at a density from 50 to 500 trees per acre. Similarly, rather than locking in a spatial layout of prescriptions, any treatment prescription that can accommodate already existing variability within the homogenous stands that are to be restored will likely be more efficient at increasing heterogeneity in that stand (Puettmann et al. 2016). For example, a goal to provide more broadleaf shrubs and trees may be achieved more easily with prescriptions that protect existing patches of broadleaves during harvesting than by creating open conditions that facilitate their development (Davis et al. 2007). Similarly, the provision of snags may be more efficient if it accounts for the harvesting damage to residual trees. Finally, flexibility in restoration prescriptions and adequate monitoring is key to efficient and successful operations.

Landscape scale—Landscape-level effects of restoration thinning are not well-studied, and experimental studies are very difficult at this scale. In a simulation study, thinning in plantations on federal ownerships increased habitat for olive-sided flycatchers (*Contopus cooperi*) but had only a slight or no effect on total habitat for northern spotted owls and other associated late-successional species (Spies et al. 2007a). The lack of effects on habitat of owls and other late-successional species was probably due to several factors, including a relatively short simulation period (100 years) compared to the several hundred years needed for old growth to fully develop. Also, the thinning prescriptions were conservative, the number of thinned trees retained for dead wood recruitment was fairly low, and the proportion of landscape thinned in the first 10 years was limited to less than 8 percent of the entire federal landscape (Spies et al. 2007a). The scope of landscape-scale restoration benefits is also limited by the state and rate of succession in the population of plantations. While young plantations cover up to 30 percent of federal forest ownerships, not all of them have the structure (high density of small and relatively young conifers) that would benefit from restoration thinning. Also, even with increased resources, it likely will take decades to treat an area that is sufficiently large enough to have a major landscape-level impact, especially as some of the ecological benefits do not show up instantly but develop slowly over time. Lack of information about the structural and compositional conditions of plantations (and location amount of restoration treatments) as well as limited understanding of the importance of fragmentation and connectedness across the region limit our ability to assess restoration needs and potential at landscape scales.

A byproduct of any large-scale restoration program is the need to maintain or even increase infrastructure. Road systems and associated travel, which are needed for various management objectives, have also been shown to negatively affect terrestrial and aquatic biological diversity and ecosystem processes (Forman and Alexander 1998, Trombulak and Frissell 2000) by serving as travel corridors for invasive species (Parendes and Jones 2000), for example. Consequently, scientific reviews note that reducing roads through decommissioning is important for meeting many biodiversity goals (chapter 7) (Franklin and Johnson 2012, Trombulak and Frissell 2000).

¹⁸ Unpublished data. On file with: K.J. Puettmann, Oregon State University, Forest Ecosystems and Society, 301L Richardson Hall, Corvallis, OR 97331.

The 80-year rule—Under the NWFP, harvesting for any goal, including thinning for old-growth restoration, is generally restricted in moist forests in LSRs to stands less than 80 years old (USDA and USDI 1994: c-13) (though some exceptions may occur). The NWFP allowed management in stands >80 years old in the matrix lands. This 80-year rule for LSRs is a one-size-fits-all approach that does not take into account that stand age is only a rough proxy for stand structure and development potential, both of which can differ greatly based on site conditions and history (Pabst et al. 2008, Reilly and Spies 2015) (fig. 3-15). That said, in general, treatments of stands >80 years old are not expected to result in substantial short- or medium-term shifts in developmental trajectories, as characterized by size and shape of trees and crowns, because trends established early in a tree's life are not easily reversed (Wilson and Oliver 2000). Understory vegetation would be more responsive. In that context, restoration thinning to promote development of complex older forest structure (e.g., large live and dead trees in stands >80 years old) of moist west-side forests is less likely to have large benefits for development of old-growth forests in the long term compared to younger forests, as many stands around age 80 begin to have some characteristics of older forests (Spies 1991, Spies and Franklin 1991) (fig. 3-15).

Our scientific understanding of the ecological effects of restoration thinning in older forests has not changed much since the early 1990s, as few empirical studies and modeling of management in older forests have been conducted (see Cissel et al. 1999 for a landscape-level modeling study). Removing larger trees could have negative impacts on the number of large live and dead trees, as trees over this age are often beginning to function as habitat for late-successional species in middle-aged stands; e.g., they develop bark characteristics that may act as microhabitat for a variety of species (Sheridan et al. 2013). However, the age, or better, the set of structural conditions (e.g., density, spatial pattern, size distribution) at which such negative impacts become important will differ with tree, stand, site, and landscape conditions, and such relationships have not been quantitatively tested. Research and adaptive management studies are needed to

test and evaluate the alternative approaches and assess the relative benefits and tradeoffs of restoration thinning in forests >80 years old.

Fire and early-successional vegetation—Possible activities relative to restoring or emulating the beneficial effects of wildfire in moist forests include creating early-seral forest and creating some of the effects of partial stand-replacement fire that were common in mixed-severity regimes of the drier part of this region. There is relatively little research and management experience with either of these activities. Managing wildfire to promote desirable fire effects may be increasingly feasible in the dry forests and remote areas of the wetter forests. However, relatively little is known about public perceptions of risk in moist forests and their willingness to tolerate wildfire in remote areas, but they do understand that any fire in moist forest is likely to be “catastrophic” (Hall and Slothower 2009). This leaves mechanical treatments and prescribed fire as the primary way to schedule and produce fire effects. The first problem in creating early-seral vegetation is determining where to create these habitats on a landscape that has already experienced a significant decline in old forests from clearcutting. Creating early-seral habitat from older forests is possible (Cissel et al. 1999, Hansen et al. 1993) and would most closely mimic natural processes that have been disrupted; however, such treatments could also reduce habitat for at-risk, older forest species and have encountered public resistance (Franklin and Johnson 2012). Consequently, Franklin and Johnson (2012) suggested that forest plantations (<80 years old) be the primary focus of any efforts to create early-seral habitat. Heavy partial harvest (i.e., retention harvest), leaving dead trees and islands of live trees, and prescribed fire would constitute an approach to creating early-seral vegetation in plantations and create variable within- and between-stand patterns for late-seral development. Such efforts would be a compromise between how wildfires would have created such communities—they would lack large live and dead trees, might not have some of the same ecological effects of fire on soil surfaces and vegetation, and would not occur in very large patches—but they would restore some components and values of this ecosystem. Combining plantations into large groups would help address the patch

size issue. A larger problem is how to determine how much of this vegetation should be created and how to schedule and distribute it in landscapes where wildfires could appear in any year and create thousands of acres of this vegetation type in a few days.

Moderately frequent mixed-severity fire regimes—

Similarly, little published research exists on restoration in moderately frequent to somewhat infrequent, mixed-severity fire regimes, which occur in the drier parts of the moist forest zone (Tepley et al. 2013) (fig. 3-6). Managers have had some experience implementing treatments that attempt to emulate partial stand-replacement fire in older forests (fig. 3-29). Cissel et al. (1999) modeled stand and landscape management based on the mixed-severity fire regimes of the western Cascades of Oregon. They found that it produced more old-forest habitat and larger patches of older forests than would have occurred if the NWFP reserve-matrix strategy had been implemented as originally designed. However, it probably would have produced less older forest structure than if no timber harvests had occurred in the matrix and wildfire was suppressed. The broader ecological effects of mixed-severity fire in forests more than 80 years old have not been studied. One hypothesis is that some late-successional conditions (e.g., spatial heterogeneity, species cohort composition, diameter diversity and development of large-diameter trees) in the drier parts of the western hemlock and Pacific silver fir zones are no longer developing at the same rate because lower severity fire would have thinned the older stands, creating gaps, initiating new shade-tolerant cohorts, and accelerating growth of surviving canopy trees (Brown et al. 2013, Tepley et al. 2013, Weisberg 2004). In general, landscapes with more fire-severity diversity (“pyrodiversity”) (e.g., mixed-severity landscapes) are known to support more biodiversity (Kelly and Brotons 2017, Perry et al. 2011, Tingley et al. 2016). Landscapes with more vegetative diversity would likely affect the rate of wildfire spread and wildfires would create more heterogeneous vegetation. Research is needed to evaluate alternative approaches to restore successional diversity in this moist forest regime through mechanical treatments, prescribed fire, and wildfire.

Ecological forestry—The “ecological forestry” approach (Franklin and Johnson 2012, Seymour and Hunter 1999), which seeks to use knowledge of disturbance ecology and retention-based management to achieve ecological and commodity goals simultaneously, has been promoted as a restoration approach for meeting goals of the NWFP. It can be applied to both moist and dry forests and is, to some degree, a branding of a collection of management actions (including those already identified for moist and dry forests [table 3-5]) that can be applied to meet ecological and social goals. Ecological forestry encompasses restoration thinning in plantations, prescribed fire, and retention silviculture (focusing on what to retain rather than on what to remove) to create early-successional patches in plantations or older forests (e.g., >80 years old) where appropriate (figs. 3-35 and 3-36). The theory behind ecological forestry is supported by scientific understanding and rooted in established concepts in silviculture and ecology (Batavia and Nelson 2016; D’Amato et al. 2017; Franklin et al. 2007b, 2018; Seymour and Hunter 1999).

No published empirical research studies exist that evaluate long-term ecological and socioeconomic effects of ecological forestry in the NWFP area. However, several of its components, including retention silviculture and disturbance-based forest management, have been evaluated in the Pacific Northwest and other places with shorter term studies. For example, global studies (Baker et al. 2016, Gustafson et al. 2012) and work in the Pacific Northwest (Halpern et al. 2012, Hansen et al. 1995a, Urgenson et al. 2013) show that retention silviculture can provide habitat and “life boats” (i.e., refugia) for older forest species (Rosenwald and Lohmus 2008) within patches of early-successional vegetation. Cissel et al. (2002) simulated a landscape-scale design for a watershed in the western Cascades that contained many elements of Franklin and Johnson’s ecological forestry approach. They found that their approach produced better ecological outcomes than implementation of the current NWFP standards and guides; however, relatively little empirical research has been published on this issue in the NWFP area.

Batavia and Nelson (2016) recently criticized ecological forestry for its lack of a clear normative or ethical goal

(e.g., conserve all species, or maximize timber production). They suggested that this deficiency will limit its practical application and subject it to the same social pitfalls as earlier and current management concepts or frameworks for finding solutions to balancing ecological and social objectives, such as “new forestry” (Franklin 1989), ecosystem management (Christensen et al. 1996, Grumbine 1994, Franklin 1997), or sustainable forestry (Lindenmayer and Franklin 1997). Different world views and values appear to present a major challenge to the implementation and acceptance of any of these approaches that attempt to achieve multiple goals from the same stands or locations. For example, DellaSala et al. (2013) criticized ecological forestry on federal lands

as placing too much emphasis on timber production and not enough on protecting habitat for the northern spotted owl, especially given the threat posed by the barred owl (*Strix varia*). At the same time, Oregon county commissioners are seeking higher levels of timber production, especially from Bureau of Land Management lands, and complain that ecological forestry does not produce enough timber for local lumber mills (Hubbard 2015). Clearly, the social aspects of active management to restore or create desired ecological patterns and processes (in any of the disturbance regimes) and producing socioeconomic values are as important to consider as the biophysical aspects (see chapter 12 for more discussion of the tradeoffs and value issues).

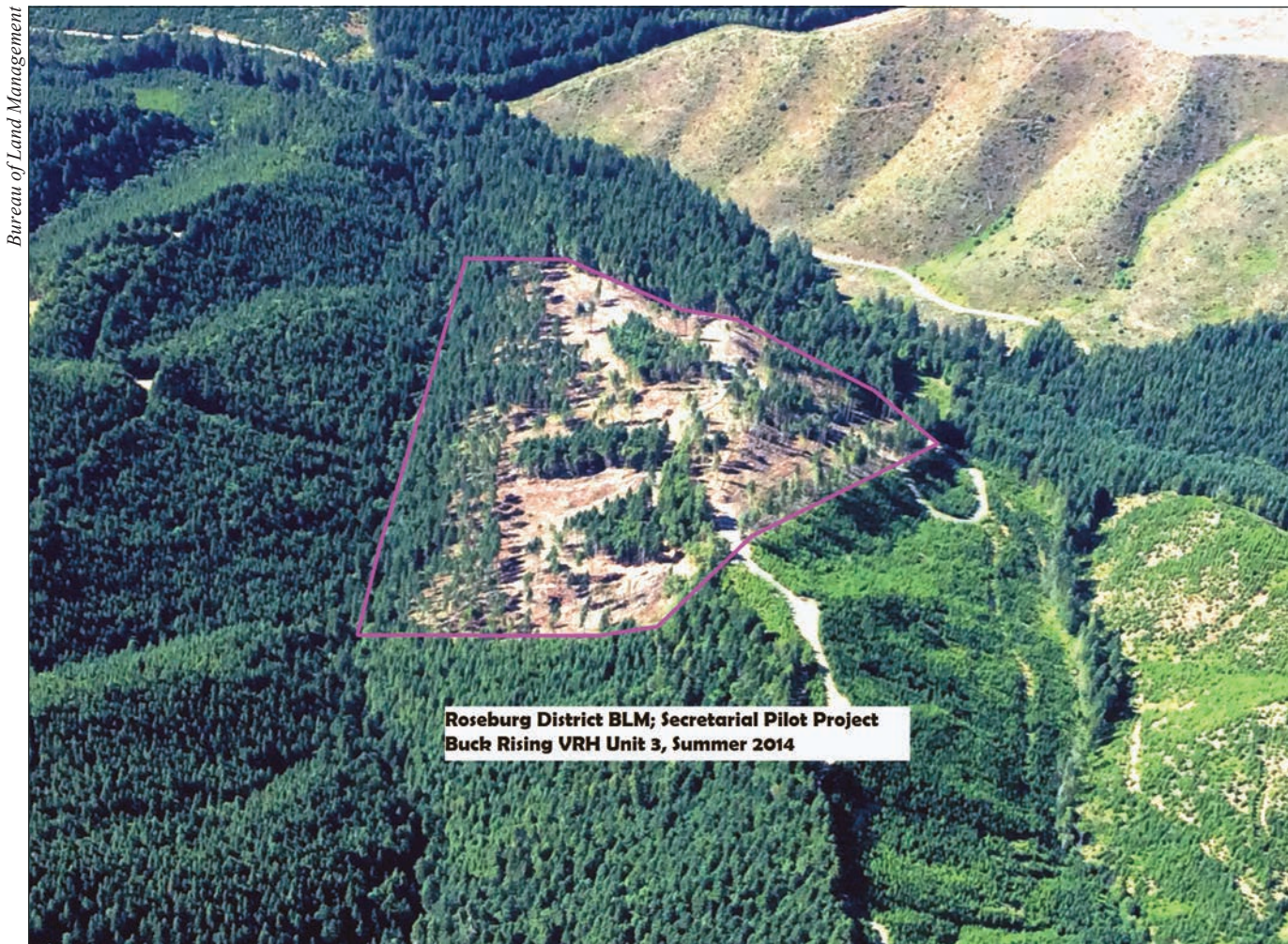


Figure 3-36—Management unit designed to create a mosaic of early habitat and leave trees, and produce wood from a young Douglas-fir forest on Bureau of Land Management (BLM) land in western Oregon. VRH = variable-retention generation harvest.

Dry forests with frequent, mixed-severity fire regimes—

Restoration approaches in both fire regimes of the dry forests include mechanical treatments and use of fire in plantations and older forests to restore or create seral stages, surface fuel beds, forest density conditions, and spatial patterns of trees that are more resistant and resilient to fire and better adapted to warming climate. Restoration strategies for the frequent mixed-severity regime in the area of the NWFP have recently been summarized in Hessburg et al. (2016) who provide an indepth review. Restoration challenges are large in this regime because of the complexity of successional pathways and variable disturbance patterns. The management strategies outlined include:

- Restoring pyrodiversity at landscape levels through prescribed fire and managed wildfire.
- Creating and maintaining successional heterogeneity based on local disturbance regimes and the needs of late-successional forest species.
- Using topography to tailor restoration treatments across landscapes.
- Protecting and restoring large and old, early-seral fire-resistant trees.
- Restoring diversity to plantations.
- Creating and maintaining early-seral vegetation, including grasslands and shrublands.
- Mitigating threats from climate change, forest insects, and pathogens.

Prescribed fire and wildfire—The literature on restoring forest fire regimes indicates that prescribed fires and wildfires managed under moderate conditions are vital components of ecological restoration. Thinning and other mechanical manipulations can achieve many structural and composition restoration goals. However, they cannot replace many important ecological processes and effects of fires, whether prescribed or wild (McIver et al. 2013). Fire, in particular, reduces surface fuels and coarse woody debris and can both increase and decrease snags and large-diameter logs depending on severity. Fire also affects soils (Certini 2005), insects (e.g., carabid beetle) (Niwa and Peck 2002), and other arthropod communities (Apigian et al. 2006). On the other hand, fires can also lead to increases of exotic

plant species (Keeley 2000) and weaken high-value trees as well as attract bark beetles (Gibson and Negrón 2009). This may be viewed negatively in a narrow sense, but in a larger ecosystem context, such indirect impacts can feed a whole suite of ecosystems processes. For example, larger bark beetle populations can attract more woodpeckers that in turn spread more wood decaying fungi, thus providing more cavities, dead and down wood and associated habitat for a whole suite of species.

Prescribed fire is often implemented at least initially following variable-density thinning to reduce stand density. Here, thinning and prescribed fire can be implemented in denser stands with or without large fire-resistant trees. Such treatments can increase the range of microclimate and resource conditions (e.g., soil moisture, light) (Ma et al. 2010). For example, Dodson et al. (2008) found a neutral to positive treatment effect from thinning and prescribed fire on understory vegetation, while other studies showed a short-term decline followed by an increase (Abella and Springer 2015). The high variability of responses appear to reflect (among others) the variability in initial conditions and the scale of observation (Dodson and Peterson 2010), with areas of low understory richness benefiting most (Dodson et al. 2008). At the same time, such treatments would reduce the likelihood of very large patches of high-severity fires that are incompatible with ecosystem and habitat needs for many species (Harrod, et al. 2009, Hessburg et al. 2016, Knapp et al. 2012a).

Landscape-scale perspectives are needed to understand the potential effectiveness of fuel treatments in modifying fire behavior. Fuel treatments affecting a small area of landscape have a low probability of intersecting a fire, given the relatively low frequencies of fire in these dry forests under full fire suppression strategy (Rhodes and Baker (2008). To be effective, treatments need to be widespread enough to influence the current level of landscape inertia (see Stine et al. 2014), and then be allowed to interact more commonly with wildfire ignitions not influenced by suppression. Spies et al. (2017), using a landscape dynamics model, found that a doubling of rates of restoration in central Oregon, which is still a relatively small area compared to historical fire frequencies, led to only a small reduction in the mean occurrence of high-severity fire over a projected 50-year

period. That study found that treatments were more effective in reducing high-severity fire years with more fire and that resilience of the entire landscape and the potential for high-severity fire was significantly lowered by higher rates of fuel treatment. Similar findings about the effectiveness of fuel treatments in altering fire outcomes have been reported by Loudermilk et al. (2013, 2014) for the relatively dry forests of the Lake Tahoe basin. Treatments to reduce density and surface fuels will need to be repeated at intervals that depend on the treatment intensity and productivity of the site (Collins et al. 2010). Given the widespread effect of fire exclusion, large areas will need to be treated (Hessburg 2016), which may be difficult for administrative and social reasons. Strategic spatial optimization of treatments can improve effectiveness per unit area treated (Finney et al. 2007), where prior commitments of land area to reserves or unique management allocations are minimal. Where major parts of the landscape are already committed to any management allocation that prevents optimal treatment allocation, spatial optimization efforts are essentially equivalent to random treatments (Finney et al. 2007).

Use of naturally ignited wildfires to achieve resource objectives is very important because, in most areas, current amounts of prescribed fire are too little to affect a sufficient area (North et al. 2012, 2015). Managing wildfire to promote ecological benefits is especially well suited for remote areas, with steep, complex topography, although it can become a more viable option in other landscapes when used in conjunction with prescribed fires, fuel reduction treatments, and footprints from past fires to create a patchwork that helps to contain the spread of natural ignitions to achieve desirable outcomes. Such fires will promote a high diversity of fire effects under moderate weather, including patches of low-, mixed-, and high-severity fires (Miller et al. 2012; Skinner et al., in press). Fire suppression and exclusion would also still be an important management tool, especially where dense older forest habitat conditions are desired, where landscapes may not yet be adapted for wildfire (e.g., contain many younger unthinned forests), or where human values are at risk from fire or smoke. Effectively managing wildfire depends on having moderate weather conditions that reduce the risk of high-severity fire effects (e.g., Estes

et al. 2017). There are few published studies about restoring fire processes and structural diversity in older forests within the mixed-severity fire regimes in the NWFP area. However, examples exist from forests of the Sierra Nevada that are quite relevant to the dry forests of the NWFP area (Collins et al. 2006, 2008, 2010; North et al. 2009; North and Sherlock 2012; van Wagendonk et al. 2012; Webster and Halpern 2010) and the Rocky Mountains (Holden et al. 2010; Larson et al. 2013; Parks et al. 2013, 2016). Among other things, these studies point out the importance of patch heterogeneity and topography as a driver in dry forest restoration.

Landscapes and resilience to climate change—

Successional heterogeneity is a product of pyrodiversity and is fundamental to biodiversity and resilience of forests to climate change (Hessburg et al. 2016). This heterogeneity occurs across a range of spatial scales from tree clumps, patches and patch neighborhoods, to landscapes (Hessburg et al. 2015). Using variable-density thinning or varying prescribed fire treatments can promote heterogeneity at these fine scales (Churchill et al. 2013, Fry et al. 2014, Lyderson and North 2012). Developing landscape-scale prescriptions for use of thinning, prescribed fire, and managing wildfire can help promote landscape-scale heterogeneity. Landscape strategies are also important to maintaining and providing habitat for species that used dense, late-successional forests (Hessburg et al. 2015, 2016) or a mosaic of late- and early-successional forests (e.g., Franklin et al. 2000). Landscape-scale models and scenario analysis are needed to better understand tradeoffs associated with managing mixed-severity landscapes for a diversity of seral stages and biodiversity objectives (Lehmkuhl, et al. 2007, Roloff et al. 2005, Spies et al. 2017). Topography can provide a valuable template for implementing landscape strategies in mixed-severity regimes (Hessburg et al. 2016). Topography, whose patterns and effects differ regionally can be used to help set goals for seral stages and prioritize treatment locations (Lyderson and North 2012, Taylor and Skinner 2003).

Increasing resilience of forests to insects, pathogens, and drought can be accomplished through efforts described above related to managing for pyrodiversity, and successional diversity in a landscape context. Altering species composition can address a number of insect and disease

concerns, including spruce beetle (*Dendroctonus rufipennis*), laminated root rot (*Phellinus sulphureus*), and western spruce budworm (*Choristoneura freemaii*) (Hessburg et al. 2016). Thinning forests can lower the likelihood of mortality associated with mountain pine beetle (*D. ponderosae*) and western pine beetle (*D. brevomis*) (Fettig et al. 2007). Thinning can reduce dwarf mistletoe infestations. Strategies to increase resilience to climate include reducing surface and ladder fuels, reducing and maintaining lower tree densities, and restoring horizontal spatial heterogeneity in forest structure, including openings where early-seral species can establish (Churchill et al. 2013). Baker and Williams (2015) argued that efforts to remove most small trees may compromise resilience, because the presence of small trees can increase resilience to insect outbreaks, which can disproportionately affect large trees. They further argued that reducing stand density is not consistent with restoration of forests, because most dry forests were historically dense (based on their GLO survey, which overestimates tree densities as we discussed above). Allen et al. (2010) in a global review of drought-induced mortality found situations where mortality in forests increases with tree density as a result of increased competition, and situations where mortality was not related to density. Bradford and Bell (2017) examined thousands of forest inventory plots from the Southwestern United States and found that mortality during warm and dry conditions was related to basal area. Similarly, Guarin and Taylor (2005) found mortality associated with basal area and tree density in mixed-conifer forests of Yosemite. Both Allen et al. (2010) and Bradford and Bell (2017) suggested that thinning is one option for increasing resilience of forests to drought. Baker and Williams (2015) argued that forest resilience is a function of diverse sizes of trees and species, which is consistent with the literature that supports the idea that efforts to increase resilience should focus less on stand or landscape averages but focus on increasing heterogeneity and forest structure and composition at multiple scales (Hessburg 2016).

Large, old, fire-resistant trees—The number of large, old, early-seral, and fire-resistant trees have been reduced in many areas as mentioned above. These keystone forest structures promote forest resilience to fire and climate change

(Agee and Skinner 2005, Hessburg et al. 2016). Management actions for maintaining and promoting these trees include (1) identifying environments that support them; (2) protecting them from logging, crown fires, and drought stress; and (3) developing future cohorts through stand management practices (e.g., reducing stand densities and prescribed fire) that promote their regeneration, growth, and crown development.

Plantations—Although plantations are a strong focus of restoration in the wetter forests, many thousands of acres of plantations also exist in dry forests landscapes that are in need of attention to promote resilience to fire and other threats. For example, precommercial thinning and prescribed burning can be used to reduce the near-term risk of loss of young, dense plantations to high-severity fire, while variable-density thinning can promote development of early-seral fire-resistant species where they are lacking in commercial-aged plantations (Stephens and Moghaddas 2005, Weatherspoon and Skinner 1995). Where desired species are lacking, planting may be needed (Hessburg et al. 2016). Where thinning is done, it will be important to treat surface fuels because logging slash will typically increase severe fire behavior in the residual stand (Huff et al. 1995, Raymond and Peterson 2005, Weatherspoon and Skinner 1995) unless trees are whole-tree yarded and slash piles are burned.

Early-successional vegetation—To cover the full suite of landscape conditions found under natural conditions, restoration efforts in the mixed-severity regimes may also consider providing early-successional habitats (Haugo et al. 2015), as mentioned above (Hessburg et al. 2016). Collins et al. (2010) suggested that silviculture could be used to mimic stand-replacing fire patches in a portion of the mixed-severity fire regime landscape. Other restoration treatments in older forests would not be stand replacing but may be targeted to remove at least part of the vegetation that established after fire exclusion, thus improving growing conditions and vigor for dominant residual trees (Latham et al. 2002). We lack research that provides guidance on how to implement restoration for early-seral conditions at landscape scales given that wildfires will continue to create this vegetation type, but early-seral conditions may highly differ from those of historical conditions depending on the successional stage of

the predisturbance forest. Collins et al. (2010) cautioned that silvicultural prescriptions may never achieve the complexity that freely burning fire can. Instead, allowing for more freely burning wildland fires would increase patch heterogeneity across landscapes and decrease potential for spread of very large high-intensity fires. However, cautions apply. Fires freely burning through dense layered stands produce very different fire effects in comparison to those where stands are open canopied and surface fuels are more limited (Miller and Urban 2000b).

Dry forests with very frequent, low-severity regimes—

Management approaches—The restoration needs and approaches (e.g., variable-density thinning, prescribed fire, and promotion of large fire-tolerant trees) in the very frequent, low-severity regime have many similarities to the frequent mixed-severity regime, but targets in terms of density, tree sizes and species, spatial patterns, and disturbance processes (e.g., frequent fire) are quite different. We emphasize some of the approaches that are unique to this fire regime. The overall needs for restoration in the very frequent, low-severity fire regime forests are larger given that fire suppression and widespread logging of large trees in many ecoregions has had a greater overall effect on forest structure and composition than in other dry zone forests; e.g., the larger number of fire cycles that have been missed owing to fire suppression.

Guidance for restoration of forests of this disturbance regime can be found in Franklin et al. (2008), North et al. (2009, 2012), Stephens et al. (2009), Franklin and Johnson (2012), Franklin et al. (2013), Stine et al. (2014), Haugo et al. (2015), and Hessburg et al. (2015, 2016). Strategies to restore old hardwood components of forests and woodlands are described for California black oak in Long et al. (2016), for Oregon white oak in Devine and Harrington (2006), and for riparian areas in southwestern Oregon in Messier et al. (2012). We summarize some of the recommendations from these publications below. A combination of harvesting and fire management is important to foster regeneration and development of large shade- and fire-tolerant canopy trees, associate understory and midstory vegetation, and to increase structural heterogeneity (e.g., areas of relatively open patches with large canopy trees). In forests that have

become denser as a result of fire exclusion, the old-tree component is often diminished or absent. This is especially prominent in drier forest areas, likely owing competition from the higher number of younger, competing trees (Dolph et al. 1995, Ritchie et al. 2008). Restoration thinning that is aimed at improving growing conditions for the larger trees appears to reverse this process (Latham et al. 2002). Thinning stands for resilience to drought and fire will require very low densities, especially of small trees and shifting composition to fire- and drought-tolerant species (Churchill et al. 2013). Studies by Hagmann et al. (2013, 2014, 2017) provide snapshots of the structure of low-density pine forests in central Oregon. Where large trees are lacking, sufficient numbers of intermediate-size trees will be needed to produce future large trees (Ritchie 2005). Flexible tree size criteria for thinning are needed to remove relatively large shade- and fire-intolerant trees that have developed in the past century of fire exclusion. It will be important to treat fuels created by mechanical treatments to reduce the risk of high-severity fire. Thinning and fuel treatments and prescribed fire should seek to reintroduce spatial heterogeneity into stands and landscapes (Haugo et al. 2015, 2016). Prescribed fire should aim for low levels of canopy mortality (e.g., 5 to 10 percent) to promote snag recruitment and spatial heterogeneity. In some cases, it may be necessary to plant drought- and fire-tolerant tree species. Topographic and soil patterns can provide a template for distributing treatments across landscapes (Hessburg et al. 2016, North et al. 2009). It will be important to consider understory plant communities in restoration plans (Franklin et al. 2013) as they have been severely degraded by grazing and are important for wildlife habitat, productivity, and providing fine fuels to promote the movement of low-severity surface fire through the landscape. For example, introducing prescribed fire after a long period of fire exclusion and accumulation of litter can lead to locally intense fires that still kill trees and rhizomatous grasses that are important for browse and form surface fuels that are needed to sustain relatively frequent surface fires. Other important considerations in restoration planning include developing efficient and effective marking guides that promote spatial heterogeneity (e.g., the individuals, clumps, and openings method) (Churchill et al. 2013, Franklin et al. 2013).

Landscapes—Landscape-scale considerations are important for altering successional patterns, general resilience to drought and wildfire, and for providing habitat for wildlife species that depend on different types of habitat, including dense conditions that may not be resilient to fire. Where restoration actions such as thinning and prescribed fire are done, it will be important to treat large patches to reduce the likelihood that treated areas will be rapidly recolonized by shade-tolerant tree species and certain shade-intolerant trees (e.g., lodgepole pine) that seed-in from nearby untreated areas. The landscape inertia (e.g., mass effects) (Stine et al. 2014) created by large areas dominated by shade-tolerant tree species will be a major influence on the rate and potential for restoring successional dynamics in these landscapes. Patch types and sizes differ in their susceptibility to high-severity fires and considering their patterns and relative abundances in landscapes is critical for restoration planning in low-severity forests and in other fire regimes. The following patch types are listed from highest to lowest susceptibility to high-severity fire (Odion et al. 2004, Thompson and Spies 2009). Note that order is not necessarily the same as management priorities, which take multiple factors into account. Landscape context (e.g., edge effects, also can play a large role in determining fire severity (Weatherspoon and Skinner 1995):

- Young homogenous plantation vegetation without slash treatment greater than 10 years after logging or fire.
- Young naturally regenerated and shrubby vegetation greater than 10 years after fire.
- Dense uniform stands of young conifers with low crown base heights.
- Dense young to mature forests without large trees.
- Dense forests containing large fire-tolerant trees and fuel ladders.
- Relatively open forests with large fire-resistant trees and low fuel ladders.

This list does not account for deciduous and evergreen hardwoods that can make patches less flammable, under less than extreme burn conditions. The appropriate mix of these types and management actions can only be determined using multiscale (patch, landscape, ecoregion) approaches

that integrate fire protection, fire restoration, and wildlife habitat goals (Hessburg et al. 2016, North et al. 2009). An overarching aim of restoration efforts could be to introduce more heterogeneity in fuel conditions at landscape levels with the goal to reduce the likelihood of rapidly spreading large fires that include large patches of high-severity fire. Such landscapes would have lower threats to large overstory fire-resistant trees that were once common and widely distributed across a large percentage of these forest landscapes (Baker 2015; Hagmann et al. 2013, 2014; Sensenig et al. 2013). A special concern with large fires that may burn as large high-severity patches is that they can remove habitat for the northern spotted owl and other late-successional species (Camp 1999, Camp et al. 1997). However, the effect on spotted owl habitat at landscape scales is a subject of uncertainty and active research (chapter 4).

Williams and Baker (2012) argued that restoration programs for ponderosa pine and dry mixed-conifer forests are “misdirected in that they are seeking to reduce all high-severity fire.” Eliminating all high-severity fire patches from forests with predominantly low-severity or mixed-severity regimes would not be supported by our understanding of fire history and ecology in these systems. Instead, efforts to reduce the size of high-severity patches or the homogeneity of current fuel loads that lead to large high-intensity fires can be justified where knowledge of local landscape conditions and fire regimes indicates that such patches would not be characteristic of the landscape or would pose a threat to important social and ecological values.

Consideration should also be given in these regimes for promoting open woodlands (e.g., oaks), open shrublands, and meadows and grasslands that have been lost as a result of overgrazing, fire exclusion, succession to forest, and other land use changes (Hessburg and Agee 2003, Hessburg et al. 2005). However, because reintroduction of fire to these systems may increase exotic species or have other unintended effects, restoration actions need to be done thoughtfully (Perchemlides et al. 2008).

Invasive Plant Species and Pathogens

Nonnative invasive plants, insects, and disease can have major economic and ecological effects on forests (Lovett et

al. 2016, Moser et al. 2009). While the problem of invasive plants and pathogens is most severe in the forests of the Northeastern United States, there are several species of plants and pathogens that are having or could have significant impacts on forests within the NWFP area (Brooks et al. 2016, Gray 2005, Lovett et al. 2016, Moser et al. 2009).

Invasive plant species often have early-successional life histories and are well adapted to colonizing disturbed areas. Examples of this type of invasive plant in this region include Scotch broom (*Cytisus scoparius*) and Himalayan blackberry (*Rubus armeniacus*), which can invade disturbed areas and oak savannas, altering soil nutrient conditions, limiting tree regeneration, and promoting growth of other nonnative species (Gray 2005, Shaben and Myers 2009). Management of these species requires an understanding of their ecology and does not lend itself to a one-size-fits all solution (D'Antonio and Meyerson 2002). Once tree canopy closure is attained, these species typically drop out of the ecosystem.

Although many invasive species invade disturbed, early-successional and open-canopy forests, closed-canopy forests, including old-growth forests, are not immune to invasive species (Martin et al. 2009). Invasion of forests by shade-tolerant species may just be slower but not necessarily less impactful in the long run than invasion of disturbed nonforest vegetation. Shade-tolerant invasive species of concern in this region include the perennial false brome (*Brachypodium sylvaticum*) and English holly (*Ilex aquifolium*). These species can outcompete native species, alter fire regimes, and possibly alter soil conditions where they occur within forests (Berger and Fischer 2016, Stokes et al. 2014, Taylor and Cruzan 2015). Management strategies for reducing spread of false brome, which is most likely to be found in lower elevation forests, include limiting disturbance within stands, cleaning clothes and equipment to reduce seed dispersal, and possibly promoting hardwoods, whose litter is less suitable for germination (Taylor and Cruzan 2015). False brome may increase flammability of forests, and short-interval fire may promote it; as climate warms, invasion of forests by false brome is expected to increase (Brooks et al. 2016).

Invasive pathogens with significant effects on forests of the NWFP area include white pine blister rust (*Cronartium ribicola*), Port Orford cedar root disease (*Phytophthora lat-*

eralis), and sudden oak death (SOD) (*P. ramorum*) (see also chapter 11). Whitebark pine (*Pinus albicaulis*), a high-elevation species of the Cascades, is in decline throughout its range as a result of the combined effects of white pine blister rust and native bark beetles (Ellison et al. 2005). The loss of this species is having cascading effects on hydrology and other species.

Sudden oak death is of particular concern because it has caused extensive mortality of tanoak (*Notholithocarpus densiflorus*), coastal live oak (*Quercus agrifolia* var. *oxyadenia*), California black oak (*Q. kelloggii*), and several other oaks in coastal forests of northern California and southern Oregon. The pathogen also infects a number of other tree and shrub species, many of which have special cultural significance to tribes (see chapter 11). Management strategies for SOD have focused on preventing or reducing transmission through quarantines that limit commercial movement of wood and host plants, and stand-level treatments, including killing and removal of infected trees and host plants, especially California bay laurel (*Umbellularia californica*), through cutting, burning, or herbicide application (Rizzo et al. 2005, Swiecki and Bernhardt 2013). Moritz and Odion (2005) reported that infections in stands that had experienced fire since 1950 were extremely rare; they suggested that a lack of fire could contribute to infestation by increasing shading, stand density, and abundance of hosts. Meentemeyer et al. (2008) concluded that reductions in fire frequency have likely facilitated SOD by increasing woodland cover and continuity at the expense of grasslands and chaparral, and by increasing bay laurel and creating more shaded, cooler microclimates.

The loss of mature tanoaks and various oaks has significant impacts on forest ecosystems in the infested areas. In heavily infested areas in conducive environments, stands formerly dominated by tanoak have been converted to shrubfields (Cobb et al. 2017, Klein et al. 2013). Additionally, infested stands could form stands with multiaged structures, a higher proportion of redwood and a lack of tanoak, and large canopy gaps (Waring and O'Hara 2008). While such changes could enhance stand structural heterogeneity, they could also jeopardize valuable ecological services such as nut production and abundance of large tree

cavities in hardwoods, which are important for fisher, owls, and other animals (Long et al. 2016). Other likely effects of the dieback include increased fuel loads, risk of high-severity burns, hazardous conditions for firefighters, increased soil erosion, and spread of invasive plants (Forrestel et al. 2015, Swiecki and Bernhardt 2013). Research in one burned landscape indicated that stands with recent SOD establishment may experience higher vegetation burn severity, while stands where dead trees have fallen may experience increased soil burn severity (Metz et al. 2011). Although high-severity fire in particular can reduce pathogen load, infected bay laurel plants that survive within such burns may infect the resprouting vegetation (Beh et al. 2012). The combination of severe fires and SOD infection may increase the likelihood of extirpating tanoak in redwood-dominated areas, because redwood generally outcompetes tanoak after fire (Ramage et al. 2010). Consequently, it is important for managers to consider landscape-scale strategies that could promote resilience to both the disease and other disturbance agents such as severe wildfire and drought. Evaluating restoration strategies through an adaptive management framework seems particularly important given the complex dynamics among vegetation, SOD and other diseases, and fire (Odion et al. 2010, Rizzo et al. 2005). Use of managed wildland fire, especially in stands that are not already heavily infested with SOD, may be particularly important as a means of promoting forest resilience. Meanwhile, infected stands may be a priority for silvicultural treatments to reduce the potential for severe crown fires (Kuljian and Varner 2010).

Postfire Salvage and Management

Ecological effects—

Postfire salvage logging is typically proposed as a means of recovering some of the lost economic value in dead or damaged trees. The ecological consequences of salvage logging are often considered negative from the perspective of soils, hydrology, postfire seedling establishment, and wildlife habitat resources, although species responses differ. Early scientific understanding of salvage logging after wildfire was hindered by a lack of studies with sufficient replication and controls (McIver and Starr 2001), but recent

research offers a more complete understanding of some ecological effects of salvage logging (Long et al. 2014). Table 3-7 summarizes key findings from several reviews to help inform management decisions surrounding postfire salvage; research on this topic is developing as more large and severe fires occur in fire-excluded landscapes. We focus on effects of salvage logging (i.e., the removal of dead trees and those that are likely to die following wildfire) rather than a broad range of other postfire management activities. However, it is important to recognize that managers often avoid replanting in areas that have not been salvage logged for crew safety and for silvicultural reasons.

Immediate stand-level effects of fire are primarily related to intensity, duration, and corresponding severity, most commonly interpreted through some measure of tree mortality and combustion of surface fuels, including dead and down wood and organic matter stored in duff, litter, and soils. Fire can reduce live tree density and canopy cover and increases the density of standing dead trees (snags) and the future abundance of dead and down wood. Although enormous amounts of carbon stored in live and dead biomass may be lost to the atmospheric carbon pool in a large fire (Campbell et al. 2007), most is retained in biological legacies, including snags, dead and down wood, charcoal, and live remnant trees (Acker et al. 2013, Baird et al. 1999, Donato et al. 2013). This carbon pool is then slowly lost from the forest as the retained deadwood decomposes or is consumed in subsequent fires (Campbell et al. 2016b, Donato et al. 2016). These biological legacies play important ecological roles that differ from the enrichment of recovering vegetation to providing microhabitats, stabilizing soils, and moderating harsh environmental conditions on burned sites (Lindenmayer and Noss 2006, Lindenmayer 2004).

Salvage logging alters postfire vegetation structure by reducing the basal area and density of live and dead trees (McIver and Otmar 2007) and decreasing the persistence of remaining snags (Russell et al. 2006) and altering the microclimate of a site (Marañón-Jiménez et al. 2013). What's more, once a tree dies, it functions as a snag, down log(s), mulch, and charcoal in soils for a period that can far exceed the period spent as a live tree (DeLuca and

Aplet 2008), although those dynamics should vary widely based upon moisture and fire regimes. Cumulatively, these reductions result in decreases in live and dead biomass (Donato et al. 2013) and reduced soil carbon. However, the down dead wood would not likely have been able to decompose in frequent fire regimes before the onset of fire suppression (Skinner 2002). Studies have shown that as wood becomes more decayed, it is more likely to be consumed in subsequent fires (Knapp et al. 2005, Uzoh and Skinner 2009). Numerous studies document initial short-term decreases in natural regeneration following salvage (McIver and Starr 2001) for various reasons, including direct mortality from mechanical damage (Donato et al. 2006) as well as indirect effects of altered competitive interactions with shrubs and harsher microclimate (Marañón-Jiménez et al. 2013, Ritchie and Knapp 2014, Stuart et al. 1993). However, one study 10 years after salvage showed no difference in natural regeneration

following a severe fire with different levels of salvage ranging from leaving everything to taking everything (Ritchie and Knapp 2014). Planting following salvage may be needed to mitigate any effects on regeneration or to establish tree species and genotypes that are better suited to climate warming or diseases. The effects of salvage logging versus no intervention on loading of fine fuels and coarse fuels and the effects of reburn are expected to differ considerably over time. If not followed by fuel treatment or accomplished through whole tree harvesting (Ritchie et al. 2013b), salvage logging can increase fine fuels to levels that support high-severity fire, which kills regeneration (Donato et al. 2006). There are few studies of the effects of salvage on subsequent wildfire, but Thompson et al. (2007) found higher reburn severity in stands that were salvaged and planted than in unmanaged stands. The Thompson et al. (2007) study hypothesized that salvage logging without sufficient treatment of the slash after logging and uniform

Table 3-7—Suggestions for ecologically based postfire management in terrestrial ecosystems from three major reviews

Recommendations	Karr et al. 2004	Beschta et al. 2004	Lindenmayer and Noss 2006
Promote natural recovery	✓	✓	
Retention of old, large trees and snags	✓	✓	✓
Protect soils against compaction and erosion	✓	✓	✓
Protect ecologically sensitive areas (e.g., reserves, roadless areas, steep slopes, fragile soils)	✓	✓	✓
Rehabilitation of roads and fire lines, avoid creation of new roads	✓	✓	
Limit reseeding and replanting	✓	✓	
Protect and restore watershed before fire	✓	✓	
Continue research, monitoring, and assessment of the effects of salvage treatments	✓		
Educate public on the natural role of wildfires, allow natural regimes	✓	✓	
Ban introduction of exotic species		✓	
Curtail livestock grazing		✓	
Low-intensity or no harvesting in unburned or partially burned patches		✓	✓
Limit removal of biological legacies from particular areas (e.g., burned old-growth stands)			✓
Ensure maintenance and creation of essential habitat elements for species of concern			✓

conifer plantations likely contributed to higher surface fuel loads after salvage and consequently to the higher reburn severity. More work is needed to evaluate the effects of salvage logging and adequate slash disposal on risk of high-severity fire. One study found that fine fuel loading following salvage returned to untreated levels after about 25 years (McIver and Ottmar 2007).

Salvage logging also reduces large fuel loads over time through removal of snags that would otherwise begin to fall and increase large dead wood on the ground as early as the first 10 years following fire, but typically over much longer periods (Dunn and Bailey 2015, McIver and Ottmar 2007, Peterson et al. 2015). One study showed that regardless of intensity of salvage logging, more than 80 percent of tree biomass left standing had transitioned to become surface fuel after 8 years (Ritchie et al. 2013b) with pines falling more rapidly than either white fir or incense cedar (*Calocedrus decurrens*) (Ritchie and Knapp 2014). Greater log biomass in unsalvaged stands resulted in higher surface temperatures during prescribed fire 20 to 30 years following wildfire (Monsanto and Agee 2008). Large areas of the Western United States have been burned by high-severity fire or killed by bark beetle outbreaks. The resulting dead fuels will become future surface fuels. Long-term research is needed to better understand the tradeoffs among postfire salvage logging and future surface fuels, and the ecological benefits of dead and down wood and future fire severity and community succession.

Salvage logging can affect ecosystem processes by altering microclimate and hydrology, increasing sediment production, and reducing soil nutrients and carbon sequestration in the forest. Removal of snags can affect microclimate by reducing shade (sometimes referred to as dead shade) and consequently reducing temperatures at night and increasing temperatures during the warming part of the day (Fontaine et al. 2010). Risk of accelerated erosion comes with ground disturbance during salvage logging (Wondzell 2001); however, there is a noticeable lack of studies from the Northwest on this issue. In one Western United States study, Wagenbrenner et al. (2015) found that salvage logging increased soil compaction, decreased soil

water repellency, and slowed recovery of vegetation, but the degree of impact depended on the method of logging, local climate, and soils. Where a winter snowpack is typical, the potential for hydrological impacts is greatest where harvest operations occur outside of the winter months. Logging over snow and frozen ground could reduce the effects on soil and sediment (Poff 1989). Indeed, Peterson and Dodson (2016) found that postfire commercial logging on dry or frozen soils in northeastern Oregon displaced or compacted an average of 15 percent of the soil surface in commercial logging units and 19 percent of the soil surface in the fuel reduction logging units, yet they found no persistent impacts on understory vegetation 15 years following treatment. In a study from central Oregon, compaction following salvage logging decreased soil respiration and available nitrogen, while later subsoiling to alleviate compaction decreased available phosphorus (Jennings et al. 2011). In several studies of boreal forests, postfire removal of snags reduced soil carbon for several years (Bradford et al. 2012, Kishchuk et al. 2015, Poirier et al. 2014). In two studies from relatively dry Sierra Nevada forests, Johnson et al. (2005) and Powers et al. (2013) found that postfire salvage resulted in a substantial reduction in onsite carbon compared to fire alone, although the authors of both studies noted that their studies, as with many other studies, did not account for sequestration in the resulting wood products. Moreover, it is important to consider long-term carbon dynamics given future fires (Carlson et al. 2012), because planting treatments can potentially accelerate carbon storage in trees, and fuel reduction treatments can potentially reduce future tree mortality.

The impacts of salvage logging on biota are mostly associated with the removal of snags and deadwood, which are important habitat components for a variety of terrestrial and aquatic organisms. Salvaging has been reported to have negative effects for several species of cavity-nesting birds, such as black-backed woodpeckers (*Picoides borealis*), three-toed woodpeckers (*P. tridactylus*), and mountain bluebirds (*Siala currucoides*), (Hutto 2006, Hutto and Gallo 2006, Saab et al. 2007), but neutral or positive effects have been documented on a few species

(Peterson et al. 2009). In a recent study from the Sierra Nevada, White et al. (2015) suggested that it was important to retain some relatively dense stands of dead or dying trees (40 to 60 per acre) at the landscape scale, to promote snag-associated species such as black-backed woodpecker, mountain bluebird, and olive-sided flycatcher, rather than evenly thinning all stands and retaining smaller numbers of snags; they suggested further research would be needed to guide the extent and configuration of such treatments. Soil bacteria and fungi appear resilient to salvage (Jennings et al. 2011). Removal of snags and large coarse woody debris could adversely affect habitat for carnivores such as fisher (*Pekania pennanti*) and Pacific marten (*Martes caurina*), if the large dead wood would have otherwise persisted into closed-forest stages where the animals use large structures for den and rest sites (Bull et al. 2001).

Fire may have positive effects by contributing wood and coarse sediment for aquatic habitats (Benda et al. 2003, Reeves et al. 1995) that may be partially negated by removal of wood during salvage logging, especially when the large wood is removed from key source areas to streams. Many aquatic and riparian organisms are adapted to fire (Flitcroft et al. 2016, Reeves et al. 2006) so postfire management is typically not needed to support aquatic ecosystems. Hillslope processes and subsequent erosion after periodic fires are critical to aquatic habitat succession, and native fish populations can often rebound within a decade after a wildfire, especially when they can recolonize altered reaches from connected refugia (Bisson et al. 2003, Dunham et al. 2003, Rieman and Clayton 1997, Rieman et al. 1997).

Management of postfire environments—

The ecological effects of postfire salvage logging can differ depending on treatment, fire severity, and biophysical setting (Peterson et al. 2009). In general, research supports the conclusion that salvage logging does not benefit native species and terrestrial or aquatic ecosystems (Beschta et al. 2004, Karr et al. 2004); an exception might include, e.g., fire-suppressed forests with high densities of trees. Further long-term research on contemporary salvage practices would greatly enhance understanding of the circumstances under which salvage might be beneficial.

Peterson et al. (2015) and Hessburg et al. (2016) identified situations, including elevated long-term woody fuel loads, lack of seed sources, and potential for reburns that maintain undesirable shrubfields, in which postfire management might be used to meet ecological goals. These include (1) fuel reduction treatments that reduce long-term levels of large woody fuels (which may be elevated as shade-tolerant species increased under fire suppression and that may pose a risk to soil fertility were the area to reburn), (2) fuel treatments or planting trees to reduce potential for high-severity reburns and forest succession where potential for large semistable patches of shrubs is high and regeneration is lacking (Dodson and Root 2013), and (3) removing surface fuels that may impede establishment of trees. The effects of particular strategies may differ considerably with ecological conditions across the NWFP area. In some cases, shrub removal may be important for promoting native plant species richness (Bohlman et al. 2016) in subsequent decades. However, shrubs may also have important roles in increasing soil carbon and nutrients, especially nitrogen. For example, in a dry ponderosa pine site in central Oregon, Busse et al. (1996) found that shrub removal aided tree growth in the first two decades, but the effect then leveled off and shrub removal was associated with decreases in soil carbon and nitrogen 35 years later.

Tree replanting, which as mentioned above is often practically tied to postfire snag removal, may be an important strategy to consider in areas where natural regeneration is too low to meet objectives for a landscape in the time desired. One example of such low regeneration was reported for several fires in the northern Sierra Nevada (Collins and Roller 2013) bordering the NWFP area but that has similar species to the Klamath region. The authors of that study noted that several studies from mixed-conifer forests in the mixed-severity regime of the Klamath-Siskiyou Mountains (Donato et al. 2009, Shatford et al. 2007) had found generally abundant conifer regeneration in stand-replacing patches. Where sites reburn and high-severity patches are large, regeneration can be low (Tepley et al. 2017). Lower and less consistent moisture may also

contribute to incidents of sparse conifer regeneration in regions predisposed to a frequent fire regime. Because promoting vegetation heterogeneity may reduce fire spread and burn severity (Thompson et al. 2007) and promote biodiversity, managers have experimented with more variable planting patterns (e.g., spacing and clustering) than have traditionally been used, but more research is needed to evaluate outcomes from such strategies.

Accumulation of large dead fuels can lead to severely burned soils if forests reburn. A study from the eastern Cascades of Oregon found that severely burned soils can have lower fertility and depleted microbial communities (Hebel et al. 2009). However, this study also found that several native plants appeared highly competitive in severely burned, low-resource soils; based upon a laboratory study component, they suggested that those native plants might be more competitive in those burned soils than invasive nonnative species. Relationships between plant diversity and fire severity are complex because they reflect variation in environment (especially precipitation and fire regime) and species composition (such as presence of invasive species). For example, DeSiervo et al. (2015) hypothesized that diversity would be promoted in fires that matched the reference fire regime, and they indeed found that native species richness was greater in areas of low to moderate vegetation burn severity of northern California (in a region of frequent fire), while areas burned at higher severity experienced more incursion by cheatgrass and other nonnative species. Similarly, Stevens et al. (2015) found that high burn severity shifted composition toward nonnative species and native species with southern-xeric affinity and away from native species with northern-temperate affinity.

Application of salvage logging in these contexts would need to consider overall effects of a wildfire on the larger affected landscape, and tradeoffs with other ecological and economic objectives. More research is needed to better understand the ecological effects of low to moderate levels of salvaging that may be done to recover economic value (Campbell et al. 2016a) from fire-killed trees.

Research Needs, Uncertainties, Information Gaps, and Limitations

While much has been learned about the ecology, conservation, and restoration of these forests, many knowledge gaps and uncertainties remain. We mention them throughout the document and summarize the major ones here:

1. While the range- and regional-scale patterns of disturbance regimes are known, much less is known about them at subregional and landscape scales. Our knowledge of the region is based on extrapolation from relatively few fire and forest history studies. Research is needed to help fill in the gaps in our knowledge especially as they relate to fire sizes, frequencies, and function in mixed-severity regimes of both the moist and dry forests.
2. We know much about the structure of old-growth forests from studies of contemporary older forests across all forest types but lack stand-structure definitions for use in monitoring and inventory related to old-growth forests that developed in the mixed- and low-severity fire regimes of moist and dry forests. Our current monitoring efforts (e.g., definitions and indices) use reference conditions for old growth that are based on forests that have been altered by fire exclusion and do not take into account structures associated with historical disturbance regimes. Research is needed to develop old-forest definitions and landscape-scale targets based on HRV, desired levels of resilience given fire, and future climate change or other considerations such as species habitat needs.
3. We lack information about the biodiversity and ecosystem functions of early-seral vegetation as well as frameworks for developing landscape-scale goals for these conditions given fire suppression. Mechanical treatments and prescribed fire can be used to approximate some of the ecological functions of diverse early-successional habitats. We also lack knowledge of what restoration actions (e.g., planting in post-wildfire environments) might be beneficial for longer term successional goals (e.g., recovery of conifer forest canopies).

4. The effects of fire suppression on forest biodiversity and ecosystem function in older forests are not well studied in much of the NWFP area. This is apart from knowledge of how succession has altered fire regimes and fire risk. Lack of fire in high-fire-frequency forests is altering plant community diversity, but more research is needed on the long-term ecosystem effects of increased stand density and shade-tolerant species in forests that were burned frequently to moderately frequently by low- to moderate-severity fire.
5. We lack a solid understanding of how drought, beetles, and disease are likely to affect forests given climate change and interactions with fire.
6. The ecological tradeoffs associated with variable-density thinning (i.e., restoration thinning) to restore or create ecological diversity in forest plantations are not well understood at stand or landscape scales and are known only from relatively short-term studies. Long-term research is needed to understand how ecosystems and the biota respond to these management actions and to learn more about the possible ecological costs and benefits of these actions in stands older than 80 years that might have undesirable densities or uniformity of trees. Similarly, long-term effects of postfire management warrant further study at large and long-term scales.
7. Given tradeoffs associated with restoration actions or inactions for different types of habitats and successional stages, research is needed to explore options for managing for a dynamic mosaic of vegetation and habitats at landscape scales under climate change. For example, how much do the pace, scale, and pattern of restoration activities at landscape scales affect fire severity and patterns of successional stages under a changing climate?
8. It will also be important to better understand the tradeoffs associated with use of both coarse- and fine-filter approaches to conservation. Dynamic landscape modeling is needed, and where feasible, landscape-scale experiments and demonstration areas will be important to advancing our understanding of this issue.

Conclusions and Management Considerations

Timber harvest, fire exclusion, fire suppression, and the loss of burning by American Indians have profoundly changed the moist and dry forests of the NWFP area. Although the motivation for the NWFP arose from clearcutting of old growth and loss of spotted owl habitat in moist forests, the dry zone forests, which occupy about 43 percent of the Plan area, have actually experienced more pervasive ecological changes as a result of human activity. Key changes in dry forests are loss of large, fire-resistant trees to logging, large departures in amounts and patterns of surface and canopy fuels, widespread shifts in proportions of seral stages, and changes in the patch sizes of those seral stages. These changes have affected all species and all processes; some in favorable ways (e.g., more habitat for dense forest species) and others in unfavorable ways (e.g., loss of open old-growth forests and ecological resilience to fire and drought). Changes in the moist forests are also significant, but they have been affected to a lesser and different degree by fire exclusion. Here, intensive timber harvest has been the primary impact on biodiversity by dramatically reducing the amount of dense old-growth forests and fragmenting habitats for species associated with these older forests. Fire exclusion in moist forests has had an important but different and less visible effect: the loss of diverse early-seral vegetation and associated reduction in landscape diversity.

The 2012 planning rule adds a new context for NWFP national forests that will undergo plan revision in the coming years: management for ecological integrity (ecosystem characteristics) and species conservation using coarse-filter approaches; fine-filter approaches are to be used for a limited number of species where coarse-filter approaches may not be sufficient. Coarse-filter approaches based on managing for ecological integrity (as opposed to coarse-filter approaches based on one vegetation type, i.e., dense old growth) would promote basic ecological processes, including major disturbances that regulate successional and fuel patterns (i.e., “habitat” for fire). Ecosystem-dynamics approaches are needed to rebuild more functional ecosystems, reduce threats to and possible listing of additional species, and provide a more ecologically viable approach to maintaining existing listed or sensitive species within the context of meeting other ecological and socioeconomic goals.

Management Considerations Summarized

- The 2012 planning rule sets a new context for ecosystem management under the NWFP: it focuses on ecological integrity based on maintaining and restoring disturbance and other ecological processes. Natural range of variation is a guide but not necessarily a target. This is a broader focus than the original coarse-filter approach of the NWFP, which focused primarily on one type of forest condition: dense, multilayered older forest.
- The goals and standards and guides for LSRs of the moist forests with infrequent fire are a relatively good match for managing for ecological integrity and resilience, especially in the face of climate change and invasive species.
- Focusing restoration (e.g., variable-density thinning) in LSRs in moist forests on plantations makes sense from a conservation perspective, and can provide jobs and economic returns. However, there will be tradeoffs with some ecological goals (e.g., amounts of dead wood) that may need mitigation.
- Fire suppression has had an effect on vegetation conditions in moist forests, especially in the drier part of the zone where fire was historically more frequent and mixed-severity effects more common. The effect is not the same as in dry forests. Fire exclusion in moist forests has reduced the amount of early-successional vegetation in the landscape, reduced diversity of structure in old-growth forests that were subject to partial stand-replacement fire, and thus reduced landscape-scale diversity. Managers may want to consider restoring fire or using fire surrogates to promote early-successional forests and landscape-scale diversity in plantations and forests more than 80 years old in the matrix. Managing for diverse early-seral stages would require a landscape-scale approach to ensure that old-growth goals are not compromised.
- The goals, standards, and guides for LSRs in dry forests are inconsistent with management for ecological integrity and resilience to climate change and fire. Dense late-successional older forests would have been historically uncommon in dry forests,

and their current higher abundance is a function of fire exclusion and suppression. Fires have become much less frequent than historically, but, when they burn, they are more likely to include large patches of high-severity fires that kill fire-resistant older trees and alter landscape-scale patch patterns. In the absence of fire, the forest structure and composition are shifting toward denser forests and shade-tolerant species that are less resistant to fire and drought.

- Management actions that promote resilience in dry forest landscapes include reducing the continuity of surface and canopy fuels to reduce patch sizes and thus the extent of high-severity fires and using prescribed fire or managing wildfire for ecological benefits where appropriate. Landscape-level strategies are needed to provide for dense forest conditions as indicated by the NWFP in environments where they are more likely to persist in the face of fire and climate change.
- Alternative approaches to the LSR network and standards and guides may better meet both coarse- and fine-filter goals by incorporating the evolving understanding of the ecological dynamics of dry forests and threats from climate change and invasive species that apply to both moist and dry forests.

Our main findings and conclusions are listed below by general topic. We also indicate which of the following questions the conclusion applies to:

Guiding Questions

1. What are the structures, dynamics, and ecological histories of mature and old-growth forests in the NWFP area, and how do these features differ from those of other successional stages (e.g., early and mid successional)?
2. How do these characteristics differ by vegetation type, environment, physiographic province, and disturbance regime?
3. What is the scientific understanding about using historical ecology (e.g., historical disturbance regimes and natural range of variation) to inform management, including restoration?

4. What are the principal threats to conserving and restoring the diversity of old-growth types and to other important successional stages (e.g., diverse early seral), and to processes leading to old growth?
 5. What does the competing science say about needs for management, including restoration, especially in dry forests, where fire was historically frequent?
 6. How do the ecological effects of treatments to restore old-growth composition and structure differ by stand condition, forest age, forest type, disturbance regime, physiographic province, and spatial scale?
 7. What are the roles of successional diversity and dynamics, including early- and mid-seral vegetation, in forest conservation and restoration in the short and long term?
 8. What is the current scientific understanding concerning application of reserves in dynamic landscapes?
 9. How do recent trends of forests in the NWFP reserve network relate to both original NWFP goals, those of the 2012 planning rule, and climate change adaptation needs?
 10. What is the current understanding of postwildfire management options and their effects?
- b. We developed a fire regime map (fig. 3-6) to provide a framework for planning and managing these diverse forests. Four major fire regimes are recognized, two in the moist forests and two for the dry forests.
 - c. The major regimes of the moist forests are:
 - i. Infrequent (greater than 200 years), high severity
 - ii. Moderately frequent to somewhat infrequent to (50 to 200 years) mixed severity.
 - d. The major regimes of the dry forests are:
 - i. Frequent (15 to 50 years) mixed severity
 - ii. Very frequent (5 to 25 years) low severity
 - e. Of these four regimes, the two mixed-severity regimes are the most variable and complex. All severities of fire occur in all regimes, but the regimes differ in proportion and spatial pattern of high-severity fire.
2. Old-growth forest structural elements common to all forests of the region include relatively large and old live, decadent and dead trees, and spatial heterogeneity of forest structure and composition. Other characteristics such as multiple canopy layers, shade-tolerant associates, and large amounts of dead and down wood are not necessarily characteristic of all old-growth forest types under the historical disturbance regimes of the region. Large-tree elements can also be found in younger forests, and patches of early-seral vegetation that developed following high-severity disturbance in older forests.

Ecology of Old-Growth and Other Vegetation Types (Questions 1 and 2)

1. Knowledge of historical disturbance regimes and successional dynamics is essential for conserving, restoring, and promoting resilience of old-growth forests and other successional stages to climate change, fire, and other disturbances.
 - a. All seral stages contribute to maintaining native forest biodiversity, ecosystem function, and other ecosystem services. Moist forests and dry forests have fundamentally different disturbance regimes, developmental pathways, and ecological potentials.
3. Definitions of old growth that recognize old-growth structural features as a continuum across stands of various ages and disturbance histories are more ecologically realistic and useful for restoration planning than a definition that has only one threshold with the result that forests are either old growth or not.
4. Current definitions of old growth used in monitoring are based on current forest inventory plots. This means that definitions for dry forests, which have

been heavily influenced by fire exclusion, are not reflective of historical forest structure and composition that were typical of this environment. Better definitions or reference conditions that reflect the variety of old growth are needed for conservation and restoration of old-growth and landscape dynamics for dry forest types, as well as communities with significant hardwood components.

5. Older forests differ in tree density, spatial heterogeneity, and species composition between moist and dry forest zones and across their associated disturbance regimes. Dense, multilayered old forests were typical of infrequent/high-severity fire regimes in moist forests parts of the region, while relatively open forest of pine, Douglas-fir, and other conifers were typical of very frequent/low- and mixed-severity regimes in dry zone forests. Dense multilayered older forest in dry forest landscapes occurred in fire refugia such as topographic settings where fire was infrequent. Old-growth forest structure and composition were most diverse in the mixed-severity regime of the moist forests and the mixed-severity regime of the dry forests.
6. Early-seral and “pre-forest” vegetation was an important component of many landscapes. Early-seral vegetation that results from high- and mixed-severity disturbance provides distinctive biodiversity and ecosystem function. Grasses, herbs, shrubs, hardwoods, and legacy live and dead trees that develop during these stages can influence forest development, biotic communities, and ecosystem function for decades to centuries.
7. Landscape diversity also varied across the disturbance regimes. In the infrequent/high-severity regime of the moist forests, the dominant landscape pattern was medium to coarse grained with very small to very large patches of older forests of complex structure, patches of younger more homogeneous forests, and rare to common (depending on climate period) very large patches of early-successional vegetation. Patches of hardwoods and

shrubs would have occurred along many streams.

In the mixed-severity regime of the moist forests, the landscape would have been a relatively dynamic mosaic of well-connected and dispersed mature and older forests and differently aged and sized patches of younger forests and preforest vegetation forests, often containing remnant live and dead large trees.

8. The forest landscape of the frequent/mixed-severity regime of the dry forests would have been a complex mosaic of forest structural types that was very strongly controlled by frequent fire. In the very frequent fire regimes, the forested part of the landscape would have been a fine- to medium-grained mosaic of older trees and very small to small patches of early-successional conditions. The open nature of the forest combined with the fine grain of patches often led to blending of areas of old trees with understory vegetation (forbs, grasses, shrubs) otherwise typical of early-seral conditions. In steep, dissected topography (e.g., northwest California), the mosaic of forest conditions would have been more strongly expressed as a function of topography and fine-scale variability in disturbance regimes and successional pathways.

Value of Ecological History (Question 3)

1. Knowledge of ecological history is essential for conducting and guiding conservation and restoration. Using HRV in forest structure, composition, and landscape patterns can be a useful guide for conservation and restoration efforts. However, returning forests and landscapes to a narrowly defined state of historical conditions and dynamics will not be possible nor desirable in many landscapes given anthropogenic forest change (e.g., land ownership patterns and forest management) and climate change. Approximations of historical regimes and forest conditions or management for resilience to fire as a recurring ecological process and climate change will be a more realistic and sustainable goal for many areas.

Conservation and Restoration Needs (Questions 4 and 5)

1. While the restoration needs differ between the moist and dry forests, logging and plantation silviculture have affected forests in all of the regimes. In the moist forests, clearcutting and plantation establishment for timber production reduced the area of old-growth forests and fragmented the landscape across millions of acres of forest lands. Intensive timber management has reduced stand-level diversity, reduced dead wood and snag abundance, increased the amount of sharp edges, and increased road densities. Clearcutting and plantation establishment affected the drier forests as well, but a more pervasive effect may have been the partial harvest of old pines that significantly reduced the abundance of large, fire-resistant trees, leaving existing older forests with far fewer large live and dead trees than they would have had under natural disturbance regimes. Moreover, often the larger overstory trees are species (e.g., Douglas-fir, grand fir, or white fir) that are not as resistant to fire.
2. Fire exclusion effects are also present in all regimes but are significantly different between the dry and moist forest zones. In the dry forests, lack of fire has greatly increased tree density and reduced resilience to fire, drought, insects, and disease. Specifically, the area of multilayered, closed-canopy older forest has increased outside the historical range over the past 100 years despite logging and recent fires. Fire suppression has also had an effect in the moist forests, but there has generally been little impact on fuel accumulation (except where logging has occurred and slash has not been treated) and fire risk as these productive forests naturally have high fuel loads. Instead, the effects of fire suppression in moist forests have been to reduce the area of high-severity fire (relative to historical dynamics), and, consequently, the area of diverse early-successional vegetation. Thus, lack of fire

in the moist, mixed-severity-regime forests has likely reduced landscape diversity.

3. Fire exclusion and succession toward shade-tolerant, fire-sensitive species may be leading to more fire-resistant older forest vegetation in some dry forests under a wider range of fire weather conditions. Forests in these areas are more shaded, dry out more slowly, have lower windspeeds, and have more compact fuel beds that are less able to carry fire than more open pine-dominated older forests. However, under extreme weather, these forests are less resistant and resilient because they are more likely to burn with high severity than historically, when forests were more open and contained less fuel. As climate changes, such extremes (e.g., drought and high winds) are expected to increase.

Competing Science Related to Need for Restoration (Question 5)

1. Some have argued that restoration is not needed because most ponderosa pine and dry mixed-conifer forests have been mischaracterized as simply having a low-severity fire regime. Instead, they contend that these forests were historically denser than most other studies indicate and are better characterized as having a more variable-severity fire regime, with significant components of mixed- and high-severity fire. Baker (2012) and others cite Hessburg et al. (2007) in support of their arguments; however, the results of Hessburg have been misinterpreted in these papers and do not fully support claims about the importance of high-severity fire in dry forests. In addition, recent research (Levine et al. 2017) indicates that the method used by Baker (2012) overestimates tree densities. We believe the preponderance of evidence supports the view that prior to Euro-American settlement, pine and dry mixed- and some moist mixed-conifer forests had relatively low tree densities and that large patches of high-severity fire were not common in dry forests

with very high frequency (<25 years) and low severities. However, larger patches of high-severity fire were an important component of dry forests (e.g., mixed conifer) with frequent (15 to 50 years) mixed-severity regimes.

Trends in Forests in the NWFP Reserve Network (Question 9)

1. At the scale of the NWFP area, losses of older forest owing to logging and wildfire over the 20 years of the NWFP have been relatively small and compensated for by significant gains from succession offsetting almost two-thirds of the losses from high-severity disturbance. However, dynamics differ geographically and with scale, and some areas, especially the Klamath region in Oregon and California, have had much higher net losses as a result of very large high-severity patches (mainly from a single large fire [Biscuit]). The NWFP reserve strategy, which focused on closed-canopy older forests is currently meeting many of the expectations of the NWFP, but it appears unlikely that this network will support the original conservation goals and new goals of the 2012 planning rule in dry forests under climate change. Threats include more frequent and larger patches of high-severity fire, which are promoted by high canopy fuel continuity and elevated surface fuel loads.

Reserve Approaches in Dynamic Landscapes (Questions 8 and 9)

1. Reserves are a valuable strategy for conserving biological diversity in the face of development and many extractive land uses. The literature indicates that goals and management guidelines for reserves need to be clearly defined. Management within reserves also may be needed in many cases to address past management effects or restore ecological processes and ecosystems that have been altered by past land use, including timber management, fire exclusion, and invasive species.

2. The options that were developed in FEMAT (1993) and set the foundation for the NWFP were based on the best available science at the time, but that science emphasized moist zone forest ecology and did not adequately deal with the substantially different ecology of forests and landscapes of the dry forest zone (Spies et al. 2006b). Although the LSRs are currently providing for late-successional/old-growth forest conservation, new science and increased understanding of fire regimes and climate change indicate that focusing only on dense older forest as the primary conservation goal across the entire NWFP area will likely have unintended negative consequences in terms of diversity of successional stages, resilience to fire and climate change, and biotic disturbance.
3. The current LSR standards, guidelines, and spatial patterns for dry forests do not appear to be consistent with emphasis on ecological integrity and other approaches for conserving biodiversity under the 2012 planning rule. In addition, threats from climate change and invasive species including the barred owl would appear to justify a reassessment of the reserve network in both dry and moist forests (see chapter 12). Development and evaluation and testing of new, highly integrated conservation approaches is encouraged to deal with changing knowledge, new perspectives on fire regimes, climate change, invasive species, and recognition of tradeoffs among biodiversity goals (e.g., coarse filter and fine filter) and between the ecological and social dimensions of forest ecosystem management (see chapter 12 for more information).

Restoration Approaches (Questions 6 and 7)

1. Restoration is more about creating landscapes for the future that are resilient to future fires and changes in climate and support native species than it is about recreating past conditions. We can use historical ecology at the community and landscape scales to understand how various patch- and landscape-level patterns will respond under these new conditions. Restoration strategies include:

- a. Variable-density thinning in plantations to increase ecological heterogeneity and accelerate growth of large trees and tree crowns.
 - b. Variable-density thinning from below and prescribed fire in dense older forests in very frequent/low-severity and frequent/mixed-severity regimes to increase resilience of those forests to fire and climate change through restoring more diverse structures and compositions of older forests.
 - c. Careful use of prescribed fire and managing wildfires away from the wildland-urban interface in dry forests and mixed-severity regimes of moist forests to restore key ecological processes while protecting critical areas of dense, older forest conditions.
 - d. Creating diverse early-successional habitat where feasible given other ecological goals and social constraints. This could include partial cutting (retention silviculture) and prescribed fire (e.g., “ecological forestry”) in plantations and perhaps in forests over 80 years old (which is allowed in the NWFP in the matrix of moist forests and within LSRs in dry forests) where this practice would be consistent with other landscape goals (e.g., resilience to fire and climate change, habitat for spotted owls, creating landscape-scale successional diversity).
 - e. Using landscape-level strategies based on disturbance regimes, topography, spatial pattern, and departure from desired historical conditions.
2. The scientific understanding of using 80 years as a threshold for restoration of stands within LSRs in moist forests has not improved much since the NWFP was established. The 80-year rule from the NWFP was based on expert opinion of stand development from data collected in natural forests of different ages. Eighty years is a one-size-fits-all threshold that does not recognize that stand age is only a rough proxy for stand structure and development potential, both of which can vary greatly based on site conditions and disturbance history.

Depending on the structure and composition of stands, and landscape context and objectives, restoration treatments in forests over 80 years could promote old-growth characteristics or reduce them (e.g., reduce number of large dead trees). However, in general, and given a lack of new information, treatments of stands over 80 years in moist forests would still be expected to have less benefit for reaching old-growth structure than restoration in stands under 80.

3. There is no new ecological science that undercuts the guideline of using alternative silviculture to meet both wood production and ecological goals in stands over 80 years in the NWFP matrix of the moist forests. Studies of retention silviculture suggest that some biodiversity elements of older forests can be retained in stands managed for a combination of timber and structural and compositional diversity.
4. All management (including restoration activities and lack of activities) involve ecological tradeoffs:
 - a. Commercial thinning can provide short-term early-seral habitat and accelerate the development of large live trees and habitat diversity for some species but may have a short-term impact on habitat quality for other late-successional species and can reduce amounts of deadwood in the future (although deadwood may be higher than the historical range owing to fire exclusion).
 - b. Thinning and restoring fire to forests with a history of very frequent fire can increase resilience to wildfire and increase habitat for species that use more open older forests and are dependent on fire, but these actions can degrade habitat quality for species that use dense older forests, which may have developed owing to fire exclusion.
 - c. Excluding fire from dry forests will increase surface and canopy fuel continuity and increase size of patches of high-severity fire when fires escape suppression and burn under extreme conditions.

- d. Excluding fire and disturbance from dry forests will typically increase stand density and shift species composition toward late-successional species and species that use dense forests and lower the resilience of these forests to fire and drought.
- e. Excluding fire from moist forests (especially in the drier parts of the moist forests) likely reduces landscape-scale vegetation diversity and the area of diverse early-successional forest and may increase the sizes of high-severity fire patches.
- f. The effects of stand-level management actions may be different when examined at different spatial scales and time periods. Multiscale and multitemporal analysis can help reveal how management effects differ with spatial and temporal scale.
- g. Tradeoffs among goals are particularly strong in managing road networks, because existing road networks can negatively affect some native species and ecosystem processes, but they also can support landscape restoration, fire management, and active management to support other ecological and socioeconomic goals.

Post-Wildfire Management (Question 10)

1. Salvage logging after wildfire does not typically generate ecological benefits for species and processes associated with patches of high-severity wildfire. However, in some cases (e.g., where fire exclusion has led to dense forests), post-wildfire management may be justified, including:
 - a. Planting key tree species after wildfires in uncharacteristically large patches of high-severity fire that may otherwise be slow to regenerate where seed sources are lacking
 - b. Thinning high-density post-wildfire regeneration as appropriate to increase heterogeneity and resilience to drought and wildfire.
 - c. Salvaging postfire pole and small-sized trees that have grown in during the period

of fire exclusion in dry zone forests, where these may constitute a significant fuel bed for reburns in the future, while retaining the medium, large, and very large trees as dead snags and down logs.

2. Actions can be taken to mitigate many of the potentially undesirable effects of salvage logging, particularly by retaining many areas that are not salvaged to ensure heterogeneity and availability of those distinctive postfire communities.

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Appendix 1: Crosswalk of Simpson (2013) Potential Vegetation Zones With Existing Vegetation From the Classification and Assessment With Landsat of Visible Ecological Grouping (CALVEG)

Table 3-8—Potential vegetation zones with existing vegetation from CALVEG^a

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%), ultra mafic 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%)

^a Percentages 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>.

Appendix 2: Fire Regime Mapping Method

Wildfire studies in Pacific Northwest forests have shown strong correlations between fire occurrence and area burned with summer temperature and precipitation (Dalton et al. 2013, Littell et al. 2009, McKenzie et al. 2004). Accordingly, we used climate variables for temperature and precipitation that coincided with the regional fire season as covariates in this mapping method. Our climate data source was the parameter-elevation regressions on independent slopes model (PRISM) climate normal data (PRISM 2015) for the period 1971–2000. We included a third variable for density of lightning-ignited wildfires data from 1970 to 2002 (Brown et al. 2002) because fires in some regions may be limited by lack of ignitions during dry periods. Each mapping variable was classified into categories based on the equal divisions of the distributions in the forested areas. Thus, each class covered a relatively equal proportion of

the forested landscape. Temperature was divided into five classes, and the other two variables were divided into three classes (table 3-9).

Potential vegetation zones (potential vegetation types) were summarized across all combinations of variable classes. Review of these data (e.g., temperature, precipitations, lightning ignition, density, and vegetation types) and expert opinion were used to assign each variable combination to one of four fire regimes: (1) infrequent (>200-year return interval) stand replacing; (2) somewhat infrequent to moderately frequent (50- to 200-year return interval), mixed severity; (3) frequent (15- to 50-year return interval), mixed severity; and (4) very frequent (5- to 25-year return interval), low severity (table 3-10). The final map product was filtered to remove pixel noise using a 3 by 3 majority filtering process.

Table 3-9—Variable map classification scheme based on quantile (by forested area) breaks

Rank	July–August mean monthly maximum temperature	May–September mean monthly precipitation	Lightning ignition density 1970–2002
	<i>°C</i>	<i>Millimeters</i>	<i>Ignitions/km²</i>
Very low	15–23	NA	NA
Low	23–25	6–32	<0.05
Moderate	25–27	32–54	0.05–1.2
High	27–30	54–189	>1.2
Very high	30–37	NA	NA

NA = not applicable.

- 1.1. Infrequent (>200-year return intervals) stand replacing (Landfire group V)
 - a. Potential vegetation type (PVT): wetter/colder parts of western hemlock, Pacific silver fir, mountain hemlock. Cover types: Douglas-fir, western hemlock, Pacific silver fir, noble fir, mountain hemlock
 - b. Area dominated by large to very large patches (10^3 to 10^6 ac) of high-severity fire, low and moderate severity also occur. Small- to medium-size patches were most frequent.
 - 1.2. Moderately frequent to somewhat infrequent (50- to 200-year return intervals) mixed severity (Landfire regime group III)
 - a. PVT: drier/warmer parts of western hemlock, Pacific silver fir and others. Cover types: Douglas-fir, western hemlock, Pacific silver fir, noble fir.
 - b. Mixed severity in space and time, typically including large (10^3 to 10^4 ac) patches of high-severity fire and areas of low- and moderate-severity fire. Small patches of high severity would be frequent.
1. Dry forests, primarily east side of Washington and Oregon, southwest Oregon, northwest California
 - 1.1. Frequent (15- to 50-year return intervals), mixed severity (Landfire regime group I and III)
 - a. PVT: Douglas-fir, grand fir, white fir, tanoak. Cover type: Douglas-fir, white fir, red/noble fir, western white pine
 - b. Mixed-severity fire with medium to large (10^2 to 10^4 ac) patches of high-severity fire
 - 1.2. Very frequent (5- to 25-year return intervals) low severity (Landfire regime group I)
 - a. PVT: ponderosa pine, dry to moist grand fir, white fir. Cover types: ponderosa pine, Douglas-fir, mixed pine, oak
 - b. Dominated by low-severity fire with fine-grained pattern ($<10^\circ$ to 10^2 ac) of high-severity fire effects, large patches of high-severity fire rare in forests except in earlier seral stage (e.g., shrub fields).

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Infrequent—high severity	Very low	Low	Low	0	0	0	0	0	12	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	Moderate	Low	14	1	0	0	0	2	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	Moderate	High	0	0	0	0	0	0	3	7	31	1	0	1	0	0	0
Infrequent—high severity	Very low	High	Low	32	0	2	0	0	0	2	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	High	Moderate	2	0	1	0	0	0	32	15	13	3	0	0	0	0	0
Infrequent—high severity	Very low	High	High	0	0	0	0	0	0	5	10	9	0	0	0	0	0	0
Infrequent—high severity	Low	Low	Low	0	3	0	0	0	8	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Low	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Low	Moderate	Low	4	41	4	0	0	2	0	0	0	0	1	0	0	0	0
Infrequent—high severity	Low	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Low	Moderate	High	0	0	0	0	0	0	2	18	13	6	1	6	0	0	0
Infrequent—high severity	Low	High	Low	37	18	23	0	1	0	2	0	0	0	1	0	0	0	0
Infrequent—high severity	Low	High	Moderate	9	0	7	1	2	0	30	4	0	0	0	0	0	0	0
Infrequent—high severity	Low	High	High	0	0	1	0	0	0	7	21	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Low	Low	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Moderate	Low	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Moderate	High	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	High	Low	1	1	11	0	0	0	0	0	0	0	4	0	0	0	0
Infrequent—high severity	Moderate	High	Moderate	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	High	High	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Infrequent—high severity	High	Low	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	High	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	High	High	Low	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	High	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very high	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very high	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very high	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very high	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very low	Low	Moderate	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes (continued)

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Moderately frequent— mixed severity	Very low	Low	High	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very low	Moderate	Moderate	0	0	0	0	0	0	3	4	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very low	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very low	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very low	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	Low	High	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	Moderate	Moderate	0	0	1	0	0	0	2	1	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	Moderate	High	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	High	Moderate	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Low	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Moderate	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Moderate	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Moderate	Moderate	Low	0	14	7	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Moderate	Moderate	Moderate	0	0	4	0	0	0	1	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Moderate	Moderate	High	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes (continued)

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Moderately frequent— mixed severity	Moderate	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Moderate	High	Moderate	1	0	7	0	0	0	2	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Moderate	High	High	0	0	3	0	0	0	4	3	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	Low	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	Low	Moderate	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	Moderate	Low	0	21	5	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	Moderate	Moderate	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	Moderate	High	0	0	3	0	0	0	0	2	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	High	Moderate	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	High	High	High	0	0	3	0	0	0	0	1	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very high	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very high	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very high	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very high	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very high	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent— mixed severity	Very low	Low	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very low	Low	Moderate	0	0	0	0	0	1	0	0	1	0	0	1	0	0	0

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes (continued)

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Frequent—mixed severity	Very low	Low	High	0	0	0	0	0	0	0	0	4	0	0	1	0	0	0
Frequent—mixed severity	Very low	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very low	Moderate	Moderate	0	0	0	0	0	3	0	0	9	0	0	0	0	0	0
Frequent—mixed severity	Very low	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very low	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	Low	Low	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	Low	Moderate	0	0	0	0	0	0	0	0	0	0	1	2	0	0	0
Frequent—mixed severity	Low	Low	High	0	0	0	0	0	0	0	0	10	67	1	4	0	0	0
Frequent—mixed severity	Low	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	Moderate	Moderate	0	0	0	0	0	4	0	0	2	0	0	2	0	0	0
Frequent—mixed severity	Low	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Frequent—mixed severity	Low	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Moderate	Low	Low	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Moderate	Low	Moderate	0	0	0	0	0	1	0	0	0	0	1	3	4	0	0
Frequent—mixed severity	Moderate	Low	High	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0
Frequent—mixed severity	Moderate	Moderate	Low	0	0	0	0	0	2	0	0	0	0	12	0	0	0	0
Frequent—mixed severity	Moderate	Moderate	Moderate	0	0	0	0	2	4	0	0	0	0	0	1	0	0	0
Frequent—mixed severity	Moderate	Moderate	High	0	0	0	0	1	0	0	0	0	0	0	8	0	0	0
Frequent—mixed severity	Moderate	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Moderate	High	Moderate	0	0	0	13	3	1	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Moderate	High	High	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Frequent—mixed severity	High	Low	Low	0	0	0	0	1	28	0	0	0	0	3	2	1	0	8
Frequent—mixed severity	High	Low	Moderate	0	0	0	0	1	1	0	0	0	0	3	7	19	0	1
Frequent—mixed severity	High	Low	High	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	High	Moderate	Low	0	0	0	0	0	2	0	0	0	0	4	0	0	0	0
Frequent—mixed severity	High	Moderate	Moderate	0	0	0	0	8	4	0	0	0	0	2	2	0	0	0
Frequent—mixed severity	High	Moderate	High	0	0	0	0	3	0	0	0	0	0	0	10	0	0	0
Frequent—mixed severity	High	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	High	High	Moderate	0	0	0	75	11	0	0	0	0	0	1	0	0	0	0
Frequent—mixed severity	High	High	High	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes (continued)

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Frequent—mixed severity	Very high	Low	Low	0	0	0	0	1	12	0	0	0	0	12	0	0	21	59
Frequent—mixed severity	Very high	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very high	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very high	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very high	Moderate	Moderate	0	0	0	0	13	0	0	0	0	0	2	1	0	5	0
Frequent—mixed severity	Very high	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Frequent—mixed severity	Very high	High	Moderate	0	0	0	11	5	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very high	High	High	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Very low	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Low	Low	High	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0
Very frequent—low severity	Low	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Moderate	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Moderate	Low	High	0	0	0	0	0	0	0	0	3	22	2	10	9	0	0
Very frequent—low severity	Moderate	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	High	Low	Moderate	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0
Very frequent—low severity	High	Low	High	0	0	0	0	3	0	0	0	1	0	7	17	48	0	0
Very frequent—low severity	High	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	High	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Very frequent—low severity	High	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Very high	Low	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Very high	Low	Moderate	0	0	0	0	10	0	0	0	0	0	14	2	14	55	26
Very frequent—low severity	Very high	Low	High	0	0	0	0	22	0	0	0	0	0	20	8	3	13	4
Very frequent—low severity	Very high	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Very frequent—low severity	Very high	Moderate	High	0	0	0	0	5	0	0	0	0	0	5	3	0	5	1
Very frequent—low severity	Very high	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Temp. = temperature; Precip. = precipitation; ABAM = *Abies amabilis*; ABLA = *Abies lasiocarpa*; ABMAS = *Abies magnifica* var. *shastensis*; ABGRC = *Abies grandis/concolor*; CHLA = *Chamaecyparis lawsoniana*; LIDE = *Libocedrus decurrens*; OAK = *Quercus* spp.; PIPO = *Pinus ponderosa*; PINUS = *Pinus* spp.; PISI = *Picea sitchensis*; PSME = *Pseudotsuga menziesii*; SESE = *Sequoia sempervirens*; THPL = *Thuja plicata*; TSME = *Tsuga mertensiana*; TSHE = *Tsuga heterophylla*.

Appendix 3: Summary of Fire History Studies in the Northwest Forest Plan

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone

Vegetation zone	Study	Extent/time period <i>Hectares</i>	Method	Frequency/ return interval <i>Years</i>	Rotation	Low/ moderate/ high <i>Percent</i>	High- severity patch size <i>Hectares</i>	Interpreted regime	Mapped regime
Redwood:	Stuart 1987	300 ha 1898–1940	Scars	7.8	—	—	—	Low	F-MS
	Finney and Martin 1989	~600 ha 1300–1860	Scars	10.1	—	—	—	Low	I-HS
	Brown and Swetnam 1994	<1000 ha 1714–1962	Scars	9.9	—	—	—	Low	F-MS
	Brown et al. 1999	Unknown	Age, scars	7–13	—	—	—	Low	F-MS
	Brown and Baxter 2003	20 316 ha 1550–1930	Scars	6–20	—	—	—	Low	F-MS
Western hemlock:	Means 1982	Unknown	Scars	100	—	—	—	Mixed	MF-MS
	Fahnestock and Agee 1983	Western Washington pre-1934	Age class from historical survey records	—	598	—	—	High	I-HA
	Stewart 1986	<1 ha ~1200–1982	Age, scars	50 ^a	—	—	—	Mixed	MF-MS
	Yamaguchi 1986	Unknown Post-1480	Age, scars	40–150	—	—	—	Mixed	I-HS
	Teensma 1987	11 000 ha 1482–1952	Age, Scars	114	78	—	—	Mixed	MF-MS
	Agee et al. 1990a or b?	3500 ha 1573–1985	Age, Scars	137	—	—	—	Mixed	I-HS
	Morrison and Swanson 1990	1940 ha 1150–1985	Age, Scars	96	95	—	<110 ha	Mixed/high	MF-MS
						0–86/ 0–60/ 0–100			
	Garza 1995	3540 ha Pre-1910	Age, scars	93–158	134	24–41/9–23/ 25–54		Mixed	MF-MS

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone (continued)

Vegetation zone	Study	Extent/time period	Method	Frequency/ return interval	Rotation	Low/ moderate/ high	High- severity patch size	Interpreted regime	Mapped regime
		<i>Hectares</i>		<i>Years</i>		<i>Percent</i>	<i>Hectares</i>		
	Impara 1997	~140 000 ha 1478–1909	Age, scars	85	271	—	—	Mixed	MF-MS/ I-HS
	Wetzel and Fonda 2000	2500 ha 1400–1985	Age, growth release	21.3 ^b	—	—	—	Low	I-HS
	Agee and Krusemark 2001	26 000 ha Pre-1900	Age, live residual structure from air photos	—	296	7–9/ 18–31/ 62–90	—	High	MF-MS
	Robbins 1999	~1562 km ² 1700–1990	Age, scars	49 (2–191)	—	—	—	Low/mixed	MF-MS
	Olsen and Agee 2005	~7000 ha 1650–1900	Age, scars	2–167	—	—	—	Mixed	MF-MS
	Weisberg 2009	14 504 ha 1550–1849	Age, scars	—	162	—	—	Mixed	MF-MS
	Wendel and Zabowski 2010	1873 ha 1568–2007	Age, scars	127	140	—	—	Mixed/high	I-HS
Silver fir:									
	Hemstrom and Franklin 1982	~53 000 ha 1200–1850	Age	—	465	—	—	High	I-HS
	Fahnestock and Agee 1983	Western Washington pre-1934	Age class from historical survey records	—	834	—	—	High	I-HS
	Agee et al. 1990a	3500 ha 1573–1985	Age, scars	108–137	—	—	—	Mixed	I-HS
	Morrison and Swanson 1990	1940 ha 1150–1985	Age, scars	239	149	0–80/ 0–78/ 0–100	<50	Mixed/high	MF-MS

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone (continued)

Vegetation zone	Study	Extent/time period	Method	Frequency/ return interval	Rotation	Low/ moderate/ high	High- severity patch size	Interpreted regime	Mapped regime
		<i>Hectares</i>		<i>Years</i>		<i>Percent</i>	<i>Hectares</i>		
	Garza 1995	3540 ha Pre-1910	Age, scars	154–246	—	24–57/20–22/ 45–50	—	Mixed	MF-MS
Mountain hemlock:									
	Dickman and Cook 1989	18 000 ha Post-1400	Age	—	—	—	>3200	High	I-HS
	Fahnestock and Agee 1983	Western Washington pre-1934	Age class from historical survey records	—	598	—	—	Mixed	I-HS
	Agee et al. 1990a	3500 ha 1573–1985	Age, scars	137	—	—	—	Mixed	I-HS
Subalpine:									
	Fahnestock and Agee 1983	Western Washington pre-1934	Age class from historical survey records	—	800	—	—	High	I-HS
	Agee et al. 1990a	3500 ha 1573–1985	Age, scars	109	—	—	—	Mixed	I-HS
Douglas-fir and grand fir/white fir:									
	Leiberg 1903	Southern Oregon ~1900	Historical land survey	—	—	—	~14 000	High	MF-MS/ F-MS
	Weaver 1959	Unknown	Scars	47	—	—	—	Mixed	VF-LS
	Agee et al. 1990a	3500 ha 1573–1985	Age, scars	52–93	—	—	—	Mixed	I-HS
	Agee 1991	197 ha 1760–1930	Age, scars	16	37–64	—	—	Low/mixed	VF-LS
	Bork 1985	~100 ha Pre-1900	Scars	8	—	—	~400	Low	VF-LS
	Wills and Stuart 1994	~20 ha 1745–1849	Age scars	10.3–17.3	—	—	—	Low	VF-LS

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone (continued)

Vegetation zone	Study	Extent/time period <i>Hectares</i>	Method	Frequency/ return interval <i>Years</i>	Rotation	Low/ moderate/ high <i>Percent</i>	High- severity patch size <i>Hectares</i>	Interpreted regime	Mapped regime
	Taylor and Skinner 1998	1570 ha 1627–1849	Age, scars	14.5	19	59/ 27/ 14	—	Low/ mixed	VF-LS
	Van Norman 1998	45 000 ha 1480–1996	Age, scars	123	—	—	—	Mixed	MF-MS
	Brown et al. 1999	2000 ha 1820–1945	Age, scars	7.7–13	—	—	—	Low	F-MS
	Everett et al. 2000	3240–12 757 ha ~1700–1860	Scars	6.6–7	11–12.2	—	2.4–40	Low	F-MS
	Stuart and Salazar 2000	~120 ha 1614–1944	Age, scars	27 (12–161)	—	—	—	Low	VF-LS
	Taylor and Skinner 2003	2325 ha Pre-1905	Age, scars	11.5–16.5	19	—	—	Low/mixed	VF-LS
	Wright and Agee 2004	~30 000 ha 1562–1995	Scars	19–24	—	—	10–100	Low/mixed	MF-MS
	Hessburg et al. 2007	~72 000 ha ~1930	Historical aerial photos	—	—	18/ 58/ 24	~10 000	Mixed	MF-MS/ F-MS
	Baker 2012	140 400 ha ~1770–1880	Live structure from historical inventory	—	496 ^c	18/ 59/ 23	—	Mixed	F-MS/VF-LS
Ponderosa pine:	Weaver 1959	Unknown	Scars	11–16	—	—	—	Low	VF-LS
	Soeriaatmadja 1966	1500–5000 ha Unknown	Scars	3–36	—	—	—	Low	VF-LS
	West 1969	Unknown	Age	—	—	—	<0.26	Low	VF-LS

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone (continued)

Vegetation zone	Study	Extent/time period	Method	Frequency/ return interval	Rotation	Low/ moderate/ high	High- severity patch size	Interpreted regime	Mapped regime
		<i>Hectares</i>		<i>Years</i>		<i>Percent</i>	<i>Hectares</i>		
	Bork 1985	~100 ha Pre-1900	Scars	4–7	—	—	—	Low	VF-LS
	Morrow 1985	2 ha Pre-1900	Age	—	—	—	<0.35	Low	VF-LS
	Hessburg et al. 2007	~106 000 ha 1930–1940	Live structure from historical aerial photos	—	—	30/ 58/ 12	—	Low/mixed	VF-LS
	Baker 2012	123 500 ha ~1770–1880	Live structure from historical inventory	—	705 ^c	40/ 44/ 16	—	Low/mixed	VF-LS

— = No value in cell

^a Interpreted regimes are classified on fire frequency classes: low <35 years, mixed 35 to 200 years, high >200 years). In cases in which fire frequency was not available, we considered fire rotation and the percentage of high-severity fire. Mapped regimes are predicted from combinations of summer precipitation, summer temperature, and lightning frequency and follow the four class regimes used in this chapter: F-MS is frequent–mixed severity, L-HS is infrequent–high severity, MF-MS is moderately frequent–mixed severity, and VF-LS is very frequent–low severity. 1 ha = 2.47 ac.

^b Stewart noted 15 fires over a 750-year period.

^c Estimated at a 200-ha (494 ac) scale.

^d Rotation for high-severity only.



A northern spotted owl in the McKenzie River Basin in Oregon.
Photo by John and Karen Hollingsworth, U.S. Fish and Wildlife Service.