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The Ecology and Management of Moist Mixed-Conifer Forests in Eastern Oregon and Washington: a Synthesis of the Relevant Biophysical Science and Implications for Future Land Management

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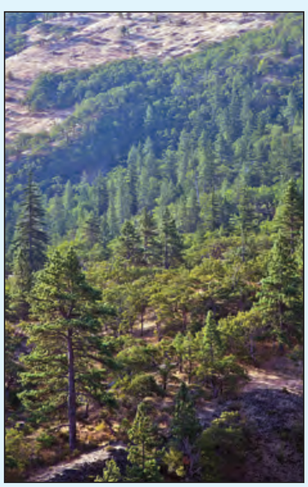

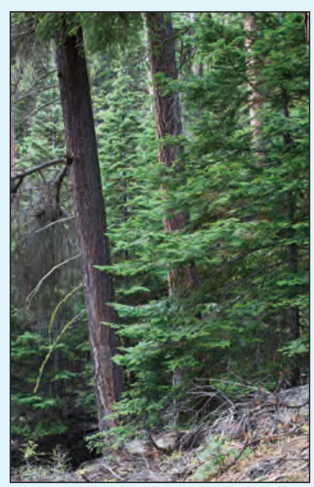

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Cover photographs: (upper left and lower right) Okanogan-Wenatchee National Forest, Washington, by John Marshall; (lower left) Columbia River Gorge, Oregon, by Miles Hemstrom; (upper right) Deschutes National Forest, Oregon, by Tom Spies.

Abstract

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Land managers in the Pacific Northwest have reported a need for updated scientific information on the ecology and management of mixed-conifer forests east of the Cascade Range in Oregon and Washington. Of particular concern are the moist mixed-conifer forests, which have become drought-stressed and vulnerable to high-severity fire after decades of human disturbances and climate warming. This synthesis responds to this need. We present a compilation of existing research across multiple natural resource issues, including disturbance regimes, the legacy effects of past management actions, wildlife habitat, watershed health, restoration concepts from a landscape perspective, and social and policy concerns. We provide considerations for management, while also emphasizing the importance of local knowledge when applying this information at the local and regional level.

Keywords: Disturbance ecology, landscape restoration, land management, resilience, stewardship.

Executive Summary

Millions of hectares of Western forests have been negatively affected by drought and by insect and disease outbreaks, and are overloaded with fuel, priming them for unusually severe and extensive wildfires. In light of these trends, public support for forest restoration has grown. One priority of the USDA Forest Service is to restore resiliency to forest and range ecosystems, enabling them to cope with an uncertain future. Natural resource managers and policymakers are awash in information from a growing body of science, with little time to sort through it, let alone assimilate the many different sources and interpretations of the best available science.

One priority of the USDA Forest Service is to restore resiliency to forest and range ecosystems, enabling them to cope with an uncertain future.

Regional research and management executives requested a review of the large body of scientific information on eastside moist mixed-conifer (MMC) forests within the context of the broader forest landscape in eastern Oregon and Washington. This focus was motivated by a lack of up-to-date management guidelines, scientific synthesis, and consensus among stakeholders about management direction in the diverse MMC type.

Understanding complex ecological and social processes and functions across landscapes requires an integrated assessment that combines multiple scientific disciplines across spatial and temporal scales. Accordingly, this synthesis compiles existing research, makes connections across disparate sources, and addresses multilayered natural resource issues. It has been prepared to assist land managers in updating existing management plans and on-the-ground projects that are intended to promote resilience in MMC forests and riparian areas. We consider management flexibility at the local scale critically important for contending with specific legacy effects of management and the substantial ecological variation in MMC forest conditions, as well as for adapting management to local social and policy concerns.

Our hope is that this synthesis will serve as a reference that provides a condensed and integrated understanding of the current state of knowledge regarding MMC forests, as well as an extensive list of published sources where readers can find further information. We also hope to enhance cross-disciplinary communication and enrich dialogue among Forest Service researchers, managers, and external stakeholders as we address common restoration concerns and management challenges for MMC forests in eastern Oregon and Washington.

Key sections of this synthesis include:

- A description of MMC forests and their context in the broader landscape of eastern Oregon and Washington.
- Key concepts of restoration and the landscape perspective.
- A comprehensive summary of pre-Euro-American settlement conditions in MMC forests.

- A description of the socioeconomic context in the region.
- A summary of human impacts on MMC forests.
- Broad management implications of research findings.
- A practical list of management considerations for diagnosing restoration needs and designing landscape approaches.

Moist Mixed-Conifer Forests

Mixed-conifer forests are a major component of the dry-to-wet conifer forest complex that is widely distributed across eastern Oregon and Washington. Among other factors, these forests are important for carbon sequestration, watershed protection, terrestrial and aquatic wildlife habitat, and outdoor recreation, and they provide economic opportunities through provisioning of a wide variety of forest products. MMC forests cover a large area east of the crest of the Cascade Range in Oregon and Washington, where grand fir, white fir, and Douglas-fir are the dominant late-successional tree species. MMC forests can be considered intermediate between drier conifer forests where pine was historically dominant and fire was typically frequent and low in severity, and wetter or cooler mixed-conifer forests where fire was less frequent and burned at higher severities. The MMC forest type is in a central position along a complex moisture, composition, and disturbance gradient of conifer forests in this region. This forest type is diverse and difficult to define, but potential vegetation types and current conditions can be used to help identify places where stands and landscapes need restoration. Historically, the forest landscape (from dry to wet) was a mosaic driven by variation in climate, soils, topography, and low- to mixed- and occasional high-severity fire.

Decades of wildfire suppression and exclusion, domestic livestock grazing, and selective and clearcut timber harvesting have interacted to alter the structure, composition, and disturbance regimes of these forests. MMC forests have become denser, have lost large individuals of fire-resistant tree species, and, on many sites, have become dominated by dense patches of shade-tolerant tree species that are less resistant to fire and less resilient to drought. These vegetation changes and management activities have shifted fire regimes toward less frequent, but larger and more severe fires, which tend to simplify the landscape into fewer, larger, and less diverse patches resulting in more homogenous conditions. Currently, many mixed-conifer forests are denser and more uniform in structure, and contain more live and dead fuel than they did historically. But the relative effects of human-caused changes, like fire suppression and timber harvesting, on these forests differ widely across the region. Thus, it is important to develop local, first-hand knowledge of the historical and contemporary disturbance regimes of these forests.

Key Management Considerations

As we reviewed the scientific literature, our primary objective was to synthesize the large body of information into succinct findings that are supported by credible research and relevant to practitioners and others interested in management of MMC forests. Some of our findings are that:

- **Historical range of variation (HRV) is useful as a guide but not as a target.** Returning to it is no longer feasible or practical in some places because of changing climate, land use, and altered forest structure and composition. The contemporary concept of restoration goes beyond the oft-stated goal of reestablishing ranges of resource conditions that existed at some time in the past (e.g., prior to Euro-American settlement). Our ecological process-oriented approach supports restoration of conditions that may have occurred in the past under certain circumstances. However, the objective of ecological restoration is to create a resilient and sustainable forest under current and future conditions. It must be forward looking. Managers have some capacity to influence the future range of variability (FRV) to achieve desired future ecosystem conditions for a landscape.
- **Disturbance regimes have been significantly altered after 150 years of Euro-American land use.** Wildfires (along with insects, pathogens, and weather) were the dominant disturbance process shaping historical forest structure and composition. Low-, mixed-, and high-severity fires occurred in MMC forests, varying in size and occurrence across ecoregions. Small and medium fires were the most numerous, but large fires accounted for the majority of the area burned. Forests today neither resemble nor function as they did 150 years ago.
- **Moist mixed-conifer forests are more vulnerable to large, high-severity fire and insect outbreaks.** Widespread anthropogenic changes have created more homogenized conditions in this forest type, generally in the form of large, dense, and multilayered patches of fire-intolerant tree species. These changes have substantially altered the resilience mechanisms associated with MMC forests.
- **Patterns of vegetation structure and composition in an eastside forest landscape shaped by intact disturbance regimes are diverse and differ over space and time.** Resilience in these forests depends on this ecological heterogeneity. Euro-American settlement and early management practices put these landscapes on new and rapidly accelerating trajectories of change

in vegetation composition and structure. Despite the change and variability, topography, soils, and elevation constrain these vegetation patterns and provide a template for understanding and managing landscape patterns. For example, south-facing aspects and ridges tended to burn more often and less severely than north-facing aspects and valleys. Landscape restoration strategies can capitalize on these tendencies.

- Several wildlife species of conservation concern require structural complexity typical of mature and old forests, which are currently limited or at risk. With no action, maintaining adequate area and spatial patterns of old-forest habitats will be a challenge with the anticipated increases in severe fire and insect infestations expected in response to changed forest conditions and climate change. Restoration at a landscape scale will encounter challenges in retaining existing old-forest patches while transitioning to a more heterogeneous and resilient forest condition.
- Community-based collaborative groups can facilitate restoration in eastside national forests. One of the major constraints to increasing the pace and scale of restoration treatments on lands administered by the National Forest System (NFS) in eastern Oregon has been the lack of social agreement about how to achieve it. The Forest Service promotes collaboration as a means for helping diverse stakeholder groups come together and find an agreeable path forward. The creation of local groups and the Forest Service's Collaborative Forest Landscape Restoration Program both offer innovations and demonstrate opportunities to improve capacity for restoration through collaborative processes.

Potential Applications

In the midst of complicated social and political forces, forest managers make decisions that require the application of complex scientific concepts to project-specific conditions. Decisions often must balance risks (e.g., elimination of fuels hazards vs. preservation of old-forest conditions) while acknowledging and allowing for uncertainties. Decisionmakers also must weigh tradeoffs associated with alternative courses of action to obtain multiple-use policy and land management objectives. We acknowledge this difficult task and the concurrent need to have access to and thoughtfully apply the best available scientific information.

It is not the role of the research community to direct management decisions. However, synthesis of research, identification of core scientific findings, and discernment of management implications in specific contexts are appropriate roles.

Research also has a role of working alongside managers to learn from successes and failures. We provide considerations for management and emphasize that their application to local and regional landscapes requires the skill and knowledge of practitioners to determine how best to apply them to a local situation with its particular management history. Legacy effects do matter, and one size does not fit all.

In chapter 5, we synthesize principle findings gleaned from the body of scientific literature (summarized in chapter 4) as they pertain to management of MMC forests. These constitute the "take-home" messages that are intended to assist land managers in the execution of their work.

The social agreement and institutional capacity for restoring MMC forests is every bit as important as the scientific foundation for doing so. The ability to institute the kinds of changes managers will consider is directly a function of the capacity of the larger community to form working partnerships and a common vision.

Some of the potential changes in forest management evoked within this document represent a departure from "business as usual." Land managers will decide how to proceed, and this will depend in large part on budget, policy, local circumstances, and ultimately the judgment of line officers. However, there are some ideas and observations from past work, both research and management, that suggest the need for some prudent adjustments in management approach.

Contents

Chapter 1—Introduction

> There is and always will be uncertainty and unpredictability in managed ecosystems, both as humans experience new situations and as these systems change because of management.
>
> Surprises are inevitable. Active learning is the way in which this uncertainty is winnowed. Adaptive management acknowledges that policies must satisfy social objectives but also must be continually modified and be flexible for adaptation to these surprises. Adaptive management therefore views policies as hypotheses—that is; most policies are really questions masquerading as answers. Since policies are questions, then management actions become treatments in the experimental sense. The process of adaptive management includes highlighting uncertainties, developing and evaluating hypotheses around a set of desired system outcomes, and structuring actions to evaluate or "test" these ideas.
>
> —Lance Gunderson (2000), *Annual Review of Ecology and Systematics*

Purpose and Scope of This Synthesis

Fire-prone mixed-conifer forests east of the crest of the Cascade Range in Oregon and Washington (hereafter, "the east side") provide clean water, recreation, wildlife habitat, and many other important ecosystem goods and services (fig. 1). However, over the last century, many of these forests have become denser and less resilient to disturbance as a result of human activity, altered disturbance regimes, and climate warming. In the last few decades, many of these forests have also become further drought stressed and increasingly vulnerable to high-severity fire (Westerling et al. 2006) and insect outbreaks as a result of climate change (Preisler et al. 2012).

In the Pacific Northwest, research and management executives from the USDA Forest Service recently highlighted the need to update management guidance, scientific synthesis, and consensus among stakeholders regarding these challenges. This synthesis responds to this need. In particular, executives asked us to focus on eastside moist mixed-conifer (MMC) forests (see definition in chapter 2), within the context of the broader forest landscape on the east side. We have written this synthesis for a diverse audience of forest planners, managers, and engaged citizens.

For this effort, we convened a team of government and university scientists to review the literature, synthesize knowledge and management options, and compile a bibliography. Team members were selected based on their experience and expertise, spanning a range of disciplines including aquatic ecology and fish biology, climate change, disturbance ecology, fire and forest ecology, landscape ecology, silviculture, social science, and wildlife biology.

John Marshall

Figure 1—A moist mixed-conifer forest, Glass Creek Watershed, Okanogan-Wenatchee National Forest, eastern Washington.

We focused the synthesis on the ecology of MMC forests but acknowledge that decisionmakers must address major issues surrounding the needs and values of human communities, so these decision processes are addressed as well. The forests of the east side represent complex patchworks of temperature, moisture, productivity, climate, and disturbance regimes, and their associated forest types (fig. 2), so we occasionally discuss forest types that adjoin the MMC type to improve overall context. The synthesis specifically addresses:

- Vegetation, landscape, and disturbance ecology.
- Wildlife habitats and populations; and aquatic ecosystems and associated species.
- Landscape approaches and perspectives.
- Silvicultural approaches.
- Climate change influences and climate futures.

Research findings summarized here can make a valuable contribution to restorative management, but it will be up to managers and program leaders to consider the unique ecological conditions of each landscape and to determine how this information guides specific land management decisions. To that end, findings here are intended to conceptually frame—not prescribe—land management. Such guidance is available in the new Forest Service Planning Rule (36 CFR Part 219), an important new administrative rule that applies current science to planning and

John Marshall

Figure 2—A mixed-conifer landscape, Middle Fork of Gold Creek, Okanogan-Wenatchee National Forest, eastern Washington.

We consider management flexibility at the local site level critically important for contending with specific legacy effects of management and the substantial ecological variation in moist mixed-conifer forest conditions, as well as for adapting management to local policy concerns.

management. We consider management flexibility at the local site level critically important for contending with specific legacy effects of management and the substantial ecological variation in MMC forest conditions, as well as for adapting management to local policy concerns.

In related dry mixed-conifer and ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forest types, research and social license to begin restoration is relatively more advanced (e.g., see Franklin et al. 2013, Jain et al. 2012), and managers are proceeding with restoration treatments (e.g., see Hessburg et al. 2013, OWNF 2012). However, we include the pine and dry mixed-conifer types in our discussions because these dry and MMC types are entwined spatially and ecologically in eastern Oregon and Washington.

Current Management Context and Restoration Mandate

Federal and state forest managers are charged with maintaining and restoring diverse, resilient (see "Glossary"), and productive forests in these fire-prone landscapes. Restoration can increase the resilience of eastside MMC forests, and reduce the ecological costs of high-severity wildfire and economic costs of fire suppression activities and postfire rehabilitation, along with the ecological and socioeconomic threats to adjoining state, private, and tribal lands. But achieving these goals is challenging without sufficient investment. Twentieth-century harvesting of large trees has reduced commercial operability in many eastside forests (e.g., see Rainville et al. 2008), and the value of remaining commercial products is often insufficient to cover the costs of restoration.

Developing and implementing sound management strategies requires up-to-date knowledge of biological and physical processes that regulate forest ecosystem structure and function. We also require information on how humans have modified these processes, especially disturbance processes, which are a key to sculpting habitat and structural patterns (e.g., see Spies 1998). In this context, a number of factors are particularly relevant to eastside MMC forests:

- The processes and patterns of eastside forests have been altered over the past 100 to 150 years (depending upon location) by the combined cumulative effects of mining, livestock grazing, road and railroad construction, conversion of grasslands and shrublands to agriculture, timber harvesting, fire suppression and exclusion, urban and rural development, invasions of alien plant and animal species (including pathogens, insects, and aquatic organisms), expanding infestations of native diseases and insects, and anthropogenically induced climate change.

- In many areas (e.g., those not burned in recent decades), today's dry and MMC forests are denser, have more small trees and fewer large fire-tolerant trees, and are dominated by shade-tolerant and fire-intolerant tree species. These conditions have reduced the fire and drought tolerance of these forests. Large increases in surface and canopy fuel loads are widespread, resulting in greater risk of large and often severe wildfires, especially during extreme fire weather conditions. Furthermore, high stand densities increase competition for growing space among trees, thereby reducing the amount of water and nutrients available to individual trees and increasing their susceptibility to some insect and disease disturbances. These changes threaten the long-term sustainability of dry and MMC forests in general, and in particular the long-term survival of remaining large and very large trees that persist in overcrowded conditions.
- Climate change is already transforming forests in eastern Oregon and Washington because it is linked to ongoing drought as well as elevated levels of tree mortality attributed to insects, diseases, and wildfire. This transformation is occurring at a brisk pace, and the window of opportunity for effecting change in this trajectory is relatively short (likely a few decades).

Mandated conservation of threatened or endangered species, some of which have been threatened by landscape alterations (first bullet above), has added ecological and regulatory complexity to forest management. For example, in the eastern Cascades, restoration of dry and MMC forests, fire regimes, and fuel patterns is often constrained by the need to minimize disturbance in areas around active nest locations of the northern spotted owl (NSO) (*Strix occidentalis caurina*), to conserve its dense, late-successional and old-forest nesting, roosting, and foraging habitats.

Currently, many landscapes of the east side have late-successional forest patterns that are out of sync with past and current fire regimes and are ecologically unsuited to providing large contiguous areas of late-successional habitat over time. In addition, much of the structural diversity in those stands is relatively short-lived because of the dominance of shade-tolerant tree species. Conservation of habitat for old forest species including NSO, northern goshawk (*Accipiter gentilis*), and pileated woodpecker (*Dryocopus pileatus*) needs to address the sustainability and continuous recruitment of that habitat. The revised recovery plan for the NSO (USFWS 2011) has called for an active management approach for sustaining and recruiting old forest habitats in fire-prone forests of eastern Washington and Oregon. Similar considerations are also an issue for species outside of the range of the NSO, particularly northern goshawks and pileated woodpeckers.

Forest environments throughout the east side are ecologically and physiographically variable: rates of change in forest conditions and effects of historical influences vary with forest type, cultural geography, and physiographic region. Although broad-scale direction can have value at times, one-size-fits-all solutions generally will not work. Instead, considerations of local conditions, land use histories, and biophysical and landuse classifications can help address this variability.

Managers report that maps of forest types across the east side are inconsistent in their classifications of vegetation associations. This creates a significant challenge for managing that vegetation for myriad uses and habitats. Managers also lack adequately detailed characterizations of how each forest type has been affected by natural, human, and climatic influences.

The relative merits of active versus passive management of forests to achieve ecological and socioeconomic goals are subject to fierce debate, which is driven by wide-ranging and often competing or conflicting societal values and is not likely to subside. Many citizens have expectations for sustainable delivery of ecosystem services from forests on public lands. Delivery is confounded by uncertainties regarding the long- versus short-term effects of various management practices on goods, services, and values.

A central purpose of restoration is to reestablish the adaptive and resilient capacities of landscapes and ecosystems in each unique physiographic region.

In the Western United States, Forest Service priorities focus on restoring ecosystems, managing wildland fires, and strengthening communities while providing jobs (Chief of the Forest Service address to the Pinchot Institute, 2013). Restoration on public lands[1] implies reenabling forests and grasslands and their associated species to adequately cope with increased climate-related stresses, and enhancing their recovery from climate-related disturbances, while continuing delivery of forest-derived values, goods, and services to citizens. **A central purpose of restoration then is to reestablish the adaptive and resilient capacities of landscapes and ecosystems in each unique physiographic region. A second, related purpose is to restore the adaptive capacity and social resilience of associated human communities, while restoring ecological systems.** The first purpose is served by restoring ecological patterns and processes that are in synchrony with the biota, geology, and climate. The second is served by enabling human communities to derive benefit from forest goods and services while conducting restoration and maintenance activities.

[1] The USDA Forest Service has defined restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. Ecological restoration focuses on reestablishing the composition, structure, pattern, and ecological processes necessary to facilitate terrestrial and aquatic ecosystem sustainability, resilience, and health under current and future conditions" (USDA FS 2013).

The concept of restoration includes, but is larger than, the goal of reestablishing the historical range of variability (HRV). A process-oriented approach supports restoring ranges of conditions that have occurred in the past (e.g., eradicating invasive species and reconnecting fragmented habitats of threatened or endangered species), in environments where the future climate will strongly resemble the recent climate, and where the results are socially understood and acceptable. Where the future climate is not expected to resemble the recent climate, the objective of ecological restoration would be to create resilient forests and rangelands that are adapted to the future climate. This idea is captured in the related notion of future range of variability, or FRV, as coined by several authors (Binkley and Duncan 2010, Hessburg et al. 2013, Keane et al. 2009, Moritz et al. 2011, Weins et al. 2012). In addition to being forward-looking, contemporary concepts of ecological restoration may emphasize social context and the connectivity and interactions among people, societies, and the biophysical elements of ecosystems. For some landscapes, it will simply be impossible and inadvisable to return them to prior conditions (Harris et al. 2006), while for others it may be well advised.

To effectively implement restoration goals across eastside mixed-conifer forests, managers can consider applying already completed assessments of conditions across the eastside landscape (e.g., the Interior Columbia Basin Ecosystem Management Project and Eastside Forest Ecosystem Health assessments), perhaps expanding on them, and then prioritizing the location and nature of treatments that would be most beneficial to ecosystems and to people. This will involve both regional and local landscape assessment.

In some places, there may be a need to increase the pace and scale of restoration to address a variety of immediate threats—including fire, climate change, bark beetle infestations, and others—for the health of public forest ecosystems, watersheds, and natural-resource-dependent communities. However, for these efforts to be sustainable, both ecological and socioeconomic vantage points would be best considered in the context of regional and local landscape patterns and processes, because in the long run the natural system must support the social system.

Structure of This Report—Where to Find Sections of Interest

Overview of contents—This report progresses from a review of the relevant science to a presentation of management considerations.

Chapter 2 discusses how MMC types fit within a regional coniferous mosaic. In this chapter, the reader will understand our definition of MMC forests, what forest types are typically included within this classification, and generally where they are located.

Chapter 3 presents ecological concepts that are foundational to landscape restoration.

Chapter 4 explains and summarizes the detailed scientific information that constitutes our synthesis. We provide a large number of citations to guide the reader through the scientific literature. Major findings are also summarized in chapter 4. Appendix 3 is provided to complement chapters 4 and 5.

Chapter 5 is the core of this report, in which the reader will find relevant management concepts gleaned from the scientific literature. We also provide a list of considerations that can help guide a landscape evaluation process. This list arises from recent experiences of eastside land managers who have adopted a landscape perspective and are conducting landscape evaluations. Silvicultural options and innovations are discussed as they relate to landscape prescriptions and their component stand-level prescriptions. We also present ideas about adjustments that may help increase institutional capacity to implement the ideas contained in this report. We close this section with an overview of the socioeconomic conditions that underlie most land management decisions.

Chapter 6 provides a brief discussion of the important institutional considerations that influence how these concepts might be implemented. This chapter also presents a summary discussion on the socioeconomic issues that are clearly in the foreground of any management strategy that seeks to restore lands and resources.

Chapter 7 provides brief conclusions and summary thoughts about how our findings may be incorporated into land management and project-level plans.

Chapter 2—Definition of Moist Mixed-Conifer Forests and the Regional Context

Moist mixed-conifer (MMC) forests are diverse and cover a large area east of the crest of the Cascade Range in Oregon and Washington, where grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) are the potential late-successional (climax) tree species (fig. 3). These forests grow in environments that are subsets of the white fir–grand fir and Douglas-fir "series," broad potential vegetation types used by Forest Service managers (Powell et al. 2007, Simpson 2007) to characterize the land's general ecological and vegetative capability.

Depending upon the environment and local site climate, patches of MMC forest typically occur where the current vegetation (not necessarily the same as potential natural vegetation or the vegetation that would develop under historical fire regimes) is a mixture of shade-intolerant ponderosa pine (*Pinus ponderosa*

Tom Spies

Figure 3—A mixed-conifer forest with overstory of ponderosa pine and patchy understory of post-1900 grand fir on the Sisters Ranger District, Deschutes National Forest, central Oregon.

Lawson & C. Lawson) or western larch (*Larix occidentalis* Nutt.), and shade-tolerant Douglas-fir, white or grand fir, and occasionally Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), as in the moist grand fir zone of the Blue Mountains (Powell 2007). In areas of complex, dissected topography, the MMC forest intermingles with ponderosa pine and dry mixed-conifer types and wetter or cooler conifer types (fig. 4) (Powell et al. 2007, Simpson 2007). Other conifers occasionally associated with MMC forest include lodgepole pine (*Pinus contorta* Douglas ex Loudon), western white pine (*Pinus* monticola Douglas ex D. Don), sugar pine (*Pinus lambertiana* Douglas), Shasta red fir (*Abies magnifica* A. Murray bis), Pacific silver fir (*Abies* amabilis (Douglas ex Loudon) Douglas ex Forbes), and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.). Dry ponderosa pine and dry mixed-conifer conditions tend to occupy lower montane settings, ridgetops, and southern exposures, whereas MMC conditions typically occur in mid to upper montane settings; on northerly and sometimes southerly aspects, especially in the upper elevations; in valley bottoms; and in lower headwall positions.

The focus of this report is not so much about forests of a particular moisture class, but mixed-conifer forests where fire exclusion has altered forest composition, structure, and function from their historical range of variability. Wetter mixed-conifer forests with longer fire return intervals may also have restoration needs, but these typically are not associated with changed fire regimes and thus are not examined in this report. The historical vegetation of MMC forest was controlled by frequent to moderately frequent fires (<20 to 50 years) that burned with mixed severity, containing both low- and high-severity patches. In most parts of the MMC forest, this fire frequency has been suspended, and disturbance regimes have been altered through a combination of historical drivers including grazing, loss of Native American fire ignitions, and active fire suppression. Consequently, the current MMC forest vegetation contains a significantly greater component of shade-tolerant tree species (e.g., white or grand fir or understory Douglas-fir) than occurred in the historical vegetation. Under historical or more fire- and drought-resilient states, these shade-tolerant species would have been less common in the understory in many areas, and large fire-resilient ponderosa pine, Douglas-fir, and western larch would have dominated the canopy layer. See appendix 3 for additional information about the variety of historical forest conditions that occurred in the Blue Mountains forest province early in the 20th century. Data are adapted from the Interior Columbia Basin Assessment (Hessburg et al. 1999a, 2000a).

Dry mixed-conifer and ponderosa pine sites (ponderosa pine series, or drier grand-fir/white or Douglas-fir subseries) typically experienced frequent fire return intervals (10 to 25 years), and exhibited a relatively open forest structure. Wetter

Figure 4—Dry mixed-conifer and pine-oak, grading into moist mixed-conifer in a drainage, Columbia River Gorge, near Rowena, Oregon.

and cooler mixed-conifer sites experienced longer fire return intervals (>50 years), and greater frequency of higher severity fire, and would have had a component of older shade-tolerant trees in the overstory, with dense areas of multistoried forest. Because few detailed fire history studies exist for the mixed-conifer forest in general, we use potential vegetation types (PVTs, see discussion below) as a surrogate for the fire regime and the degree to which mixed-conifer forests have departed in terms of composition and structure. These potential natural vegetation types, which include both series and approximations of subseries (e.g., dry, dry-moist, moist, and moist to wet variants of grand, white, and Douglas-fir series) can be a starting place for identifying the environments and locations of MMC forest (table 1). The PVTs used by managers (e.g., series, subseries, and plant associations) are often the best available source of local information, but are only an approximate surrogate for the fire regime of a site or local landscape. Figure 5 illustrates the continuum of forest types found along an elevational gradient on the east side of the Cascade Range and the relationship of forest type to general fire regimes.

Table 1—Area of major forest potential vegetation types in eastern Oregon and Washington

	All ownerships	Federal lands			Nonfederal
		Wilderness	Other	Total	
			Hectares		
Potential vegetation type:					
Douglas-fir (dry)	1 818 129	57 050	740 450	797 500	1, 020 629
Grand fir/mixed conifer (cool/moist)	1 654 634	169 229	929 871	1 099 100	555 534
Grand fir (warm/dry)	433 195	24 283	335 693	359 977	73 218
Mixed conifer (cold/dry)	98 149	3056	73 966	77 022	21 127
Mixed conifer (dry)	1 096 662	15 495	442 618	458 114	638 548
Lodgepole pine	180 526	1646	114 442	116 087	64 439
Mountain hemlock	505 756	171 993	225 136	397 129	108 627
Pacific silver fir	172 084	59 154	84 780	143 934	28 150
Ponderosa pine	1 804 724	10 645	1 040 868	1 051 514	753 210
Subalpine fir	616 773	172 152	343 853	516 005	100 768
Western hemlock	304 794	10 214	180 064	190 278	114 516
Other potential vegetation types	18 675 361			6 136 634	12 538 727
Total	27 360 787			11 343 294	16 017 493

Source: the Integrated Landscape Assessment Project.

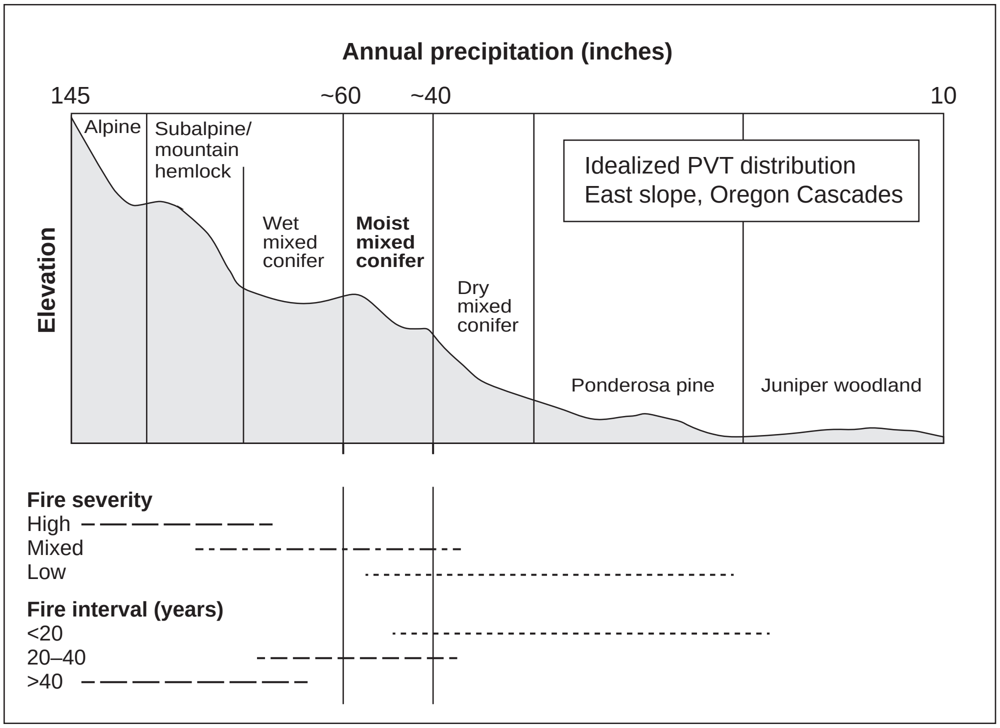


Figure 5—Continuum of forest types found along an elevational gradient on the east side of the Cascade Range and the relationship of forest type to general fire regimes. PVT = potential vegetation type.

Variation in slope, topographic position, and landscape context can create a high degree of variation in fire regimes within the same potential vegetation type.

Variation in slope, topographic position, and landscape context can create a high degree of variation in fire regimes within the same PVT. For example, for a given PVT, areas with steep or concave slopes often experience more high-severity fire than gentler and more convex slopes. Likewise, MMC PVT patches embedded within large dry forest patches or adjacent to grass or shrub patches may experience a higher fire frequency than they would if they were embedded in MMC forest. Context matters, and is critical to interpreting the native fire regime. Final determination of MMC forest for restoration purposes should be based on landscape context, local environment, and disturbance history. Figure 6 provides a conceptual model of the complex ecological setting in which we find MMC forests. There are many biological and physical conditions that will influence what kind of vegetation is found at a site.

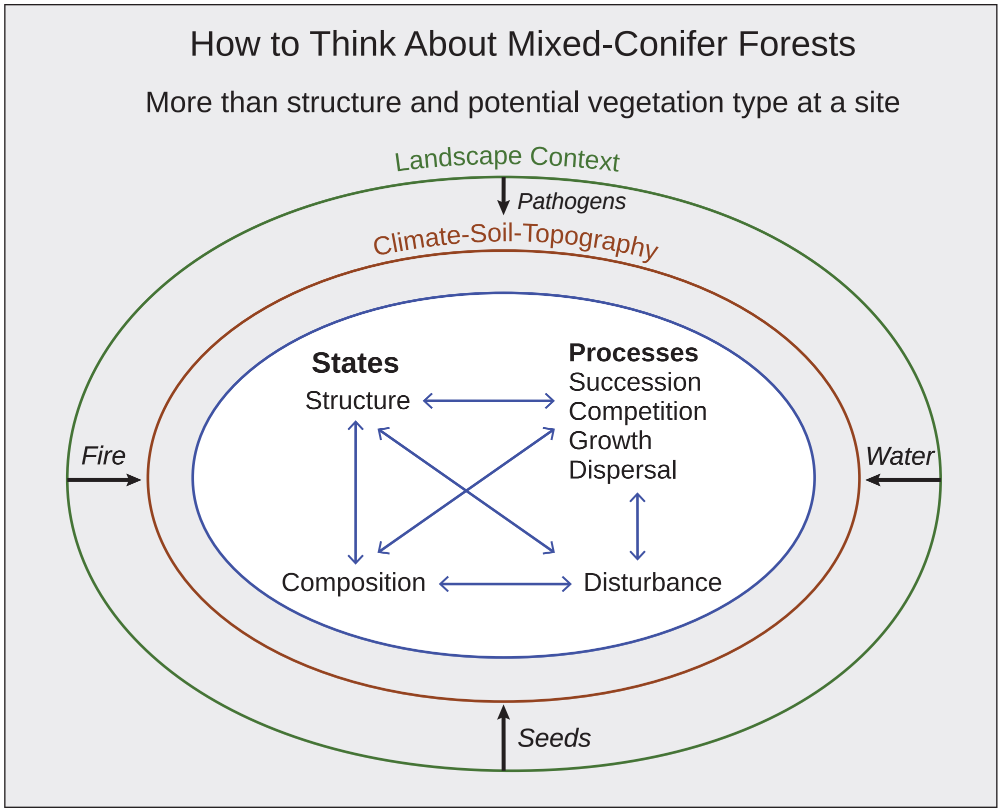


Figure 6—Conceptual model of the complex ecological setting of moist mixed-conifer forests.

The MMC type is widely distributed across the east side (fig. 7) occupying particular environments and elevations along the east slope of the Cascade Range and large patches within the northern and central portions of the Blue Mountains. It often is sandwiched between the drier mixed-conifer and pine types and cooler and wetter mixed-conifer types. Figure 7 is intended to give a general depiction of the distribution of MMC forest. No standard, peer-reviewed maps of dry, moist, or wet PVTs exist for the entire region. See "Ecological Composition, Patterns, and Processes Prior to Euro-American Settlement (Before About 1850)" in chapter 4 for more details on sources of regional information and details on vegetation in general.

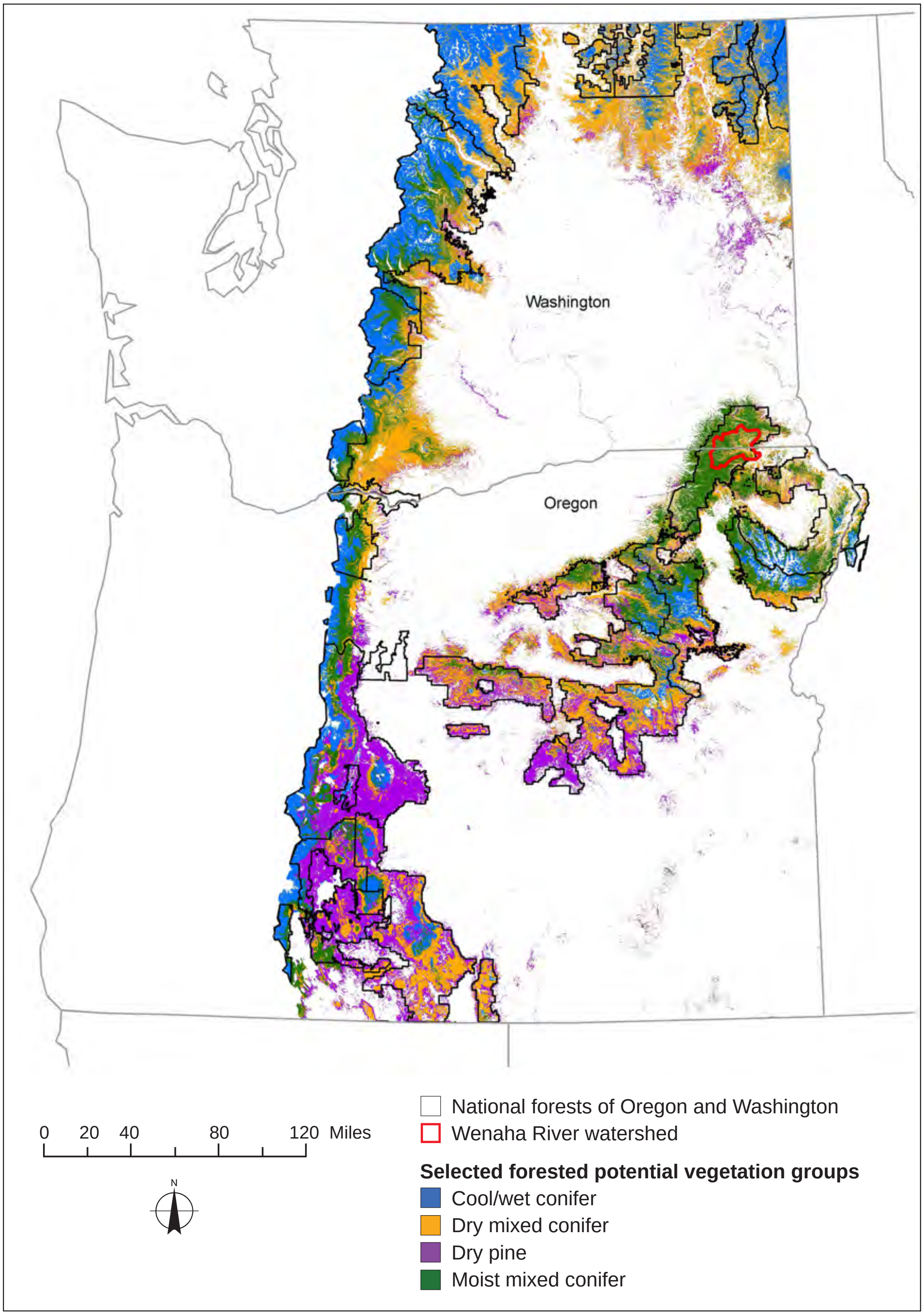


Figure 7—Map of selected potential vegetation types that support mixed-conifer forests in eastern Oregon and Washington, using data from the Integrated Landscape Assessment Project (Pacific Northwest Research Station and Oregon State University).

Chapter 3—Ecological Principles of Restoration and Landscapes

Restoration of ecological processes and patterns requires a multi-scale spatial and temporal perspective. Historically, the forests in eastern Oregon and Washington were diverse and complex in their species composition and structure (e.g., tree sizes, ages, density, layering, clumpiness). These patterns influenced the frequency, severity, and spatial extent of native insect, disease, wildfire, and abiotic disturbance processes such that signature "disturbance regimes" were apparent. However, a century or more of management has significantly altered patterns of structure and composition, and as a direct consequence, the associated disturbance regimes are highly altered as well. Figure 8 illustrates the large disparity in historical vs. current species composition and structure of forests in eastern Oregon and Washington.

Moist mixed-conifer (MMC) forests were and continue to be hierarchically structured systems with complex interactions between spatial scales. Part of what concerns us today is uncertainty driven by these cross-scale disturbance interactions. For example, very large wildfires, insect outbreaks, and changes in winter

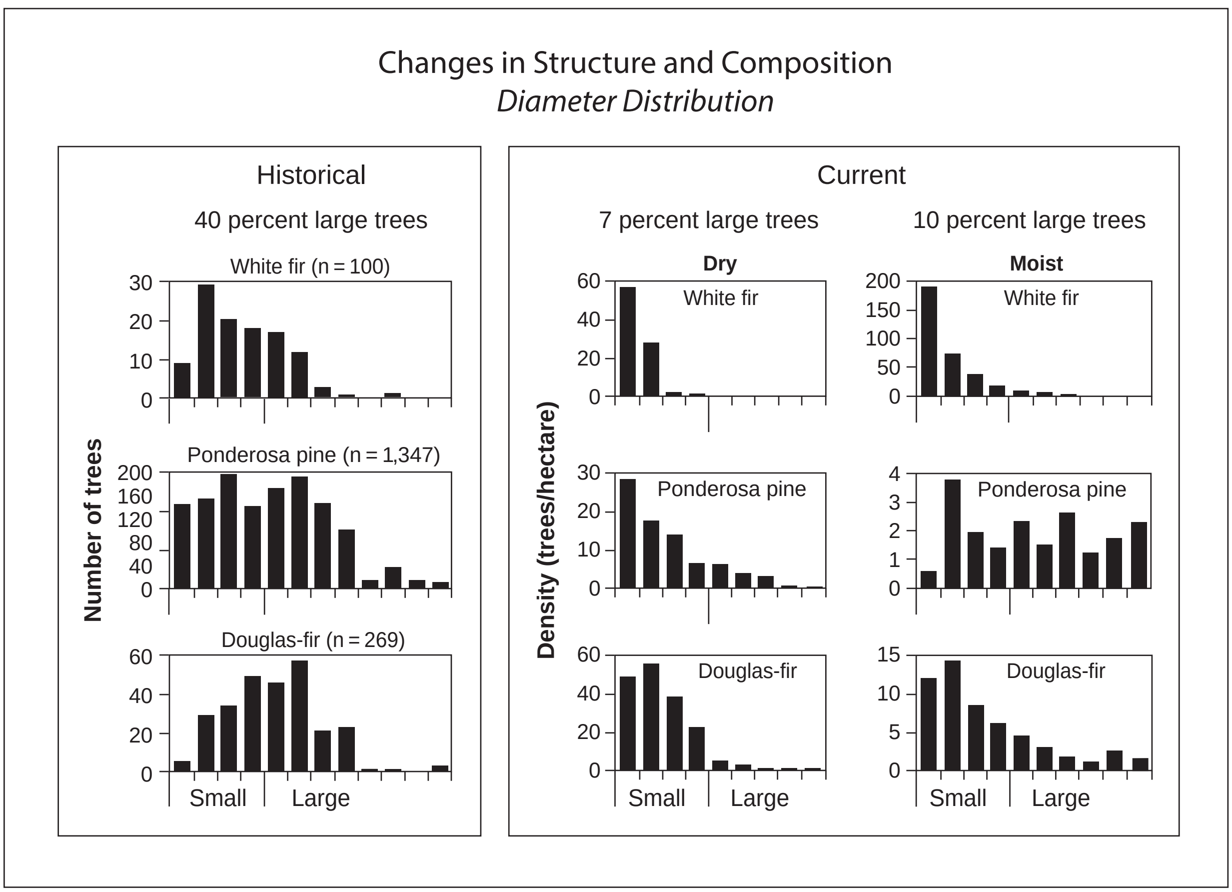


Figure 8—Disparity in historical vs. current species composition and structure of forests in eastern Oregon and Washington.

snowpack and hydrology occurring at a regional scale threaten to alter meso-scale patterns and disturbance processes of local landscapes. Simply working at meso-scale (local landscapes, e.g., watersheds and subwatersheds) does not address these concerns. The answer lies in regional- or ecoregional-scale solutions.

Forest ecosystem responses to disturbances or weather changes are often nonlinear, or involve complex feedback loops or time lags, particularly when we allow for longer observation periods.

At a meso-scale, patterns of forest structure and composition emerge, which are primarily maintained by interactions among environments, topography, weather, soils, geomorphology, and disturbances. But other broader and finer scale patterns also exist, and these are also influential to maintaining and changing meso-scale vegetation patterns over space and through time.

Forest ecosystem responses to disturbances or weather changes are often nonlinear, or involve complex feedback loops or time lags, particularly when we allow for longer observation periods. Thus, some interactions are relatively more unpredictable and may not manifest in any sort of change in the short term, until some kind of threshold is reached (Malamud et al 1998, Moritz et al. 2013, Peterson 2002). The challenge for scientists and managers seeking to restore landscape resilience is to develop a better understanding of ecological complexity that can more readily be translated into practical strategies for restoration.

Our review of recent theory, observation, and understanding in the field of landscape ecology shows it is critical to consider long-term spatial and temporal phenomena prior to drawing conclusions or developing simplified decision rules based solely on temporally short or narrow geographic observation windows.

The Concept of Resilience

The concept of resilience promises a robust alternative to management goals based on static conditions or simple applications of the historical range of variability.

The resilience of current and future forest ecosystems is a major concern of land managers today. The concept of resilience promises a robust alternative to management goals based on static conditions or simple applications of the historical range of variability (HRV). However, resilience is not an easily defined concept in a practical sense (Gunderson 2000). Furthermore, operational metrics of resilience have received little attention to date (Carpenter et al. 2001). To become more operational, resilience must be defined in terms of specific system attributes and in relation to specific disturbances or perturbations. It is important to understand that resilience is a relative term and is constrained by space and especially time. Eventually change will be significant enough that a previously resilient system will reset itself into a new state. Thus, some bounding of space and time is necessary to define resilient states.

Folke (2006) and Gunderson (2000) have identified three conceptual domains of resilience: (1) engineering resilience, which focuses on recovery or return time to a stable equilibrium (e.g., return to a particular forest structure or composition); (2) ecological resilience, which focuses on maintaining function and persistence with multiple equilibria (e.g., HRV); and (3) socioecological resilience, which focuses on reorganization, adaptive capacity, and multiscale interactions among the many community members, stakeholders, and responsible government organizations that have an interest in the outcome of land management. Although we focus mainly on ecological resilience in this report, we acknowledge that the ecological system is imbedded in a socioeconomic system that interacts with ecological systems.

The degree of alteration of an ecosystem and its dynamics (patterns of change over time and space) must be understood before we can consider if and how the system can be restored (fig. 9).

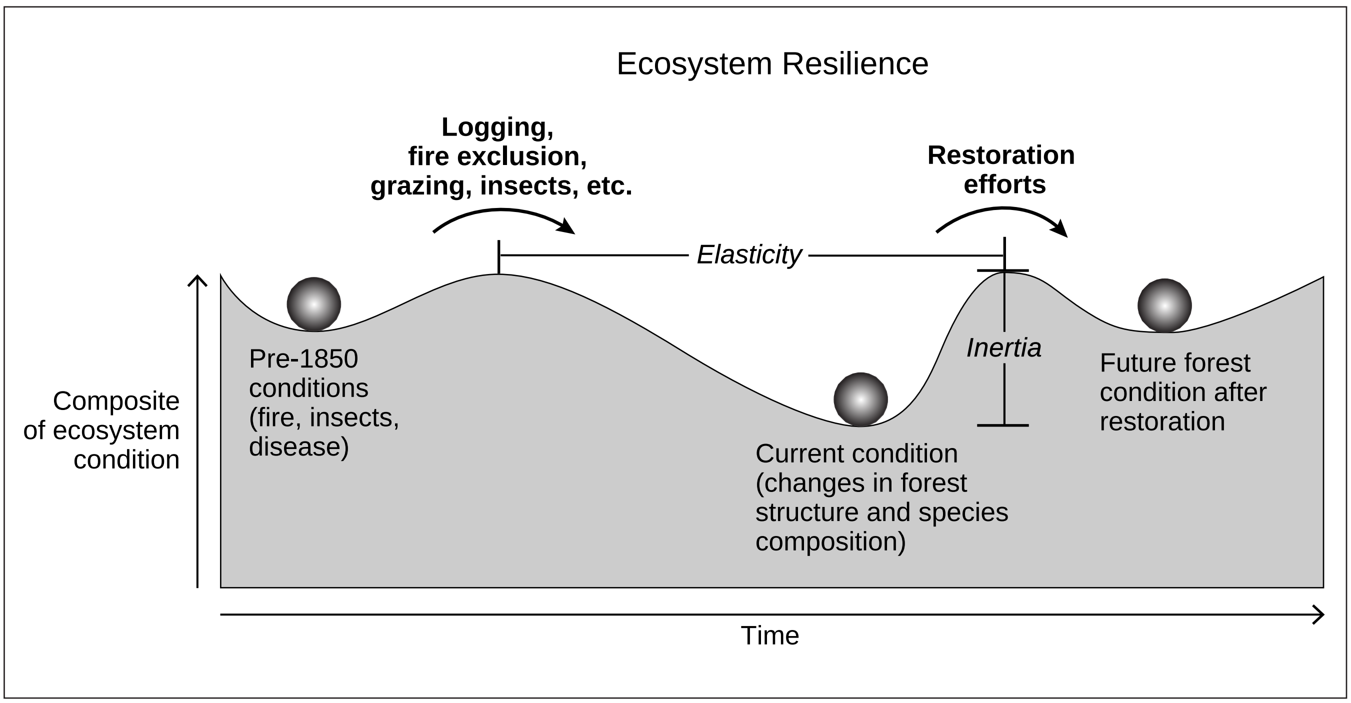


Figure 9—Ball and cup heuristic of ecosystem stability (adapted from Gunderson 2000). Valleys represent the boundaries in which ecosystems are coping with disturbances, balls represent the ecosystem, and arrows represent disturbances. Some disturbances push the system out of a past “stable” state into a different ecosystem condition. The influences of inertia and elasticity are indicated in the diagram.

Westman (1978) suggested five characteristics that depict the potential resilience of a system:

Inertia: The resistance of a system to disturbance.

Elasticity: The speed with which a system returns after disturbance.

Amplitude: A measure of how far a system can be moved from a previous state and still return.

Hysteresis: The lagging of an effect behind its cause, such as delayed response of the system to a disturbance.

Malleability: The difference between the pre- and postdisturbance conditions. The greater or lesser the malleability, the lesser or greater will be the system's resilience.

These technical terms from systems analysis illustrate the complexity of system responses to change and the challenge of restoring dynamic systems. The characteristics of resilience described here are not all measured with equal ease; some may simply be immeasurable in a short period because of a lack of historical data or reliable ecosystem models (Westman 1978). Given these limitations on data availability and the overall understanding of ecosystem interactions, we are reminded of a compelling need to employ an adaptive management approach in both research and monitoring, where we can track the results of management efforts and follow disturbances and recovery efforts over a long term. This would give managers the ability to inform subsequent management decisions with what was learned from previous management decisions and thus make appropriate adjustments. It also suggests the value and need for ecosystem models and tools that can include consideration and measurement of both inertia and resilience. Early versions of these tools have given us a better understanding of the complex dynamics of forest ecosystems and, in turn, how we can craft management strategies to achieve desired outcomes. In short, management for ecosystem resilience necessitates iterative steps to allow for adjustments at each juncture of trial and learning.

Resilience does not always result in desirable conditions on the land (Folke 2006). Degraded and nonnative vegetation can also be resilient in its own way; e.g., landscapes dominated by cheatgrass (*Bromus tectorum* L.), which is generally considered undesirable, can be resilient in the face of restoration efforts by land managers. In this example, managers have experienced significant difficulties attempting to restore sagebrush steppe ecosystems invaded by cheatgrass (Chambers et al. 2009, D'Antonio et al. 2009).

Landscapes often have considerable inertia because of alterations of pattern and process after more than a century of human activity. For example, Wallin et al. (1994) found that landscape patterns generated by past forest management or disturbance can take decades or centuries to restore (see also Heinselman 1973). Hysteresis (in a large dose) can operate in altered landscapes to create undesirable resilience or a delay in desired response to management. For example, the widespread accumulation of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) and, to a lesser degree, grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.) across many eastside landscapes now means that disturbance patches created by management or wildfire are more likely to regenerate to Douglas-fir and grand fir than they would have in the recent past. These relations can create significant challenges in executing successful restoration because of the large scale of the effect.

Ecological Principles for Landscape Planning and Management

Our perspective on the scientific principles underlying restoration of landscape resilience in eastside landscapes is based on three central ideas: (1) the biophysical environment (i.e., vegetation, climate, geology, and topography) and disturbances interact to control system behavior at several key spatial and temporal scales; (2) Euro-American activities have altered these ecological interactions and reduced landscape resilience; and (3) increasing resilience of desired conditions requires management actions that restore processes and patterns at these key scales. Without a broad geographic and ecosystem perspective that includes the past, present, and future role of humans and the climate, it will be impossible to restore resilient forests across a wide range of ecoregions and landscapes. Such a systems view should enable more effective application of treatments to meet restoration or resilience goals.

We use the terms **local** and **regional** to describe landscapes. We define local landscapes as variably sized areas, typically ranging in size from one to several subwatersheds (hydrologic unit code [HUC] 6 or 12-digit watersheds) (Sieber et al. 1987) (see also the National Hydrography Dataset at http://nhd.usgs.gov/), or a watershed (HUC 5 or 10-digit watershed). These subwatersheds collectively reside in a single ecological subregion (sensu Hessburg et al. 2000a), and exhibit characteristic patchworks of successional stages and topography consistent with the climate, biota, physical processes, and disturbance regimes of that subregion. Subwatersheds (i.e., HUC level 6) typically range in size from about 10,000 to 40,000 ac (4050 to 16 200 ha), but larger and smaller subwatersheds also occur. We define a regional landscape as the complete collection of all local landscapes that comprise an ecological subregion.

We are motivated by eight core ecological principles that are foundational to restoring eastside forests, including moist mixed-conifer forests.

We define (local or regional) landscape resilience as the capacity of the ecosystems to absorb disturbance and climate change while reorganizing and changing but essentially retaining the same function, structure, identity, and feedbacks (adapted from Walker et al. 2004). Resilient landscapes maintain a dynamic range of species, vegetation patterns, and patch size distributions (broad- or meso-scale) that emerge under the constraints of the climate, geology, disturbance regimes, and biota of the area.

In this chapter, we outline eight core ecological principles that are foundational to restoring eastside forests, including MMC forests. We expand on these in later chapters; however, our aim here is to highlight key ideas that motivate our thinking.

Physical and biological elements of an ecosystem interweave, creating distinctive patterns on a landscape. Climate, interacting with vegetation, disturbance, topography, soils, and geomorphology created domains of ecosystem behavior at local and regional landscape scales. During every historical climate period, a range of patterns and patch sizes of forest successional stages likely emerged. This emergent natural phenomenon is referred to as the natural range of variation (NRV), also referred to as the HRV. As the climate shifted, so did the NRV. The notion of a static NRV is a common misperception and provides a misleading objective for land managers. It does not exist. Prolonged periods of warming or cooling, wetting or drying, or combinations of these have occurred repeatedly over time. Whenever these changes have happened, they have pushed the NRV in new directions. But sudden and extensive shifts in the NRV were typically constrained, except under the most extreme climatic circumstances, by the lagged landscape memory encoded in existing patterns of living and dead vegetation. This is the quality of a natural system that we represent as landscape resilience.

Vegetation dynamics and fire regimes of MMC forests were and are among the most variable. In the historical forest, this forest type exhibited low-, mixed-, and high-severity fires; the amount of each severity type varied with the climate regime, and by physiographic region. Patterns of vegetation structure and composition and fire frequency and severity changed gradually across landscapes with variation in climate and the impacts of settlement and early management. But before management, fire regimes of MMC forests were variable, depending on topography and ecoregion. In some locations, fires occurred relatively frequently (every 10 to 30 years), in others, fire frequency was more variable (25 to 75 years). In the former, fire severity would have been primarily low- and mixed-severity, with surface fire effects dominating. In the latter case, fire severity would be primarily mixed- and high-severity, with active and passive crown fire effects dominating. In some ecoregions, the fire regimes and tree composition of the dry ponderosa pine, dry mixed conifer, and MMC types were quite similar and there were no clear lines of demarcation. In others, the differences in these types in terms of aspect orientation, tree density, layering, and species mixes were pronounced. With the advent of fire suppression, understories of the ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) and dry mixed-conifer forests were in-filled largely by ponderosa pine and shade intolerant Douglas-fir, respectively, but the MMC forests were in-filled by grand or white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.) and Douglas-fir. Grand fir and Douglas-fir understories may have been transient in some historical MMC forests: if they got established during a period without fires, they may or may not have been eliminated by subsequent frequent low-severity fires. During longer intervals between fires, significant fuel ladders may have developed and mixed-severity fire effects would have been typical. On wet mixed-conifer forest sites, shade-tolerant tree species were persistent and fire intervals were long enough to allow development of old shade-tolerant trees and larger patches of dense multilayered forests within stands and landscapes.

Eastside forest patterns and influential processes are constantly shifting over space and time. However, topography, soils, and elevation constrain these patterns and provide a relatively simple template for understanding and managing landscapes for them. As a first approximation, the topographic and edaphic patterns of landscapes provide a natural template for pattern modification and restoration. For example, spatial patterns of ridges and valleys, and north- and south-facing aspects, strongly represent characteristic patterns and size distributions of historical vegetation patches. North-facing aspects and valley bottoms historically supported the densest and most complex forest structures, and when fires occurred, these sites experienced more severe fire behavior than south-facing aspects and ridges, owing to site climate and growing season factors. The same is true today. In contrast, south-facing aspects and ridges tended to burn more often and less severely than north-facing aspects and valleys. Wildfire conditions in summer were typically drier and fine fuels were conditioned for burning, even during average summer burn conditions.

Ecosystems and their component parts are organized in an interactive, hierarchical arrangement. Processes associated with the regional landscape exert a measure of control over patch dynamics of local landscapes, and ultimately, some fine-scale patterns and processes within patches. For this reason, no forest type, its disturbance regime, or its variation may be thought of in isolation. Some landscapes are dominated by one topographic aspect (e.g., north- or south-facing); consequently, vegetation on minor aspects may be different than would otherwise be expected from knowledge of their site conditions alone. In landscapes with more southerly aspects and ridges, corresponding northerly aspects typically also see more frequent fires and lower than typical severity. Knowledge of specific context and scale can help guide patch-level decisions.

Postsettlement human activities have resulted in increasingly homogenized forests and, in turn, significant changes in the scope and effects of natural disturbances. Widespread human-caused changes to vegetation structure, composition, and fuelbeds have created more homogenized conditions in the MMC forest, generally in the form of large, dense, and multilayered patches of intermediate-aged fire-intolerant tree species. These changes have increased the area and frequency of large, high-severity fire patches, and the extent and frequency of other biotic disturbances (e.g., bark beetle and budworm outbreaks). The changes have also substantially altered the resilience mechanisms associated with the native forests. Climate and the characteristic disturbance regimes and landscape patterns regulated the composition, frequency, and size of the largest patches. Prior to Euro-American settlement, local and regional landscapes had developed over a period long enough for coarse and fine-scale patterns and species composition to be in some degree of synchrony with their physical environments and the climate system. Much of this synchrony has been lost through the cumulative effects of relatively recent human activities on the landscape. Current and future fire regimes and landscape patterns are on new trajectories trending away from a previously established resilience. To restore this coupling between patterns and processes (wildfire, insects, pathogens, and weather), forest structure, composition, and landscape patterns must be modified at a scale that is consistent with the scale of the current vulnerabilities. Pattern modifications should be consistent with the inherent disturbance regimes of large landscapes and forest types, and with the climate regime.

Rare ecological events can have a disproportionately large effect on ecosystems. Rare, large-scale events (disturbance, climatic, biotic, geologic) can significantly affect future landscape dynamics, especially if their frequency, size, or severity are unprecedented for the climatic, biotic, and environmental conditions. Large wildfires, dramatic climatic extremes, rapid changes in plant and animal species distributions, and large insect outbreaks are examples of natural and human-caused events that are rare but can have a strong and lasting effect on future landscape patterns and processes. Typically, these events are hard to predict and some are outside the control of managers. Nevertheless, they shape current landscapes and can be anticipated when developing and gaging the extent and timing of risk mitigation strategies. To a modest degree, managers can, through cumulative smaller actions, prepare landscapes in a manner that reduces the likelihood or impact of these large and rare events. However, to be effective, the timing and extent of the actions must match the level of inertia that supports the large-scale events. For example, large areas of eastern Oregon and Washington are susceptible to chronic western spruce budworm (*Choristoneura occidentalis*) infestation owing to the wide prevalence of Douglas-fir, grand fir, and white fir in large, dense, multilayered patches. Changing this situation would require the reduction in the prevalence, complex layering, and density of these host tree species over a very large area to match the scale of the vulnerability to this disturbance.

Resilience depends on ecological heterogeneity and varies with spatial and temporal scale. At no time were all patches of a landscape resistant to fires or other disturbances. At any given time, some patches within a landscape were always susceptible to insect attack, stand-replacing fires, pathogen infections, or a combination of these. In some ecoregions, as much as 25 to 35 percent of the forest had been recently burned by high-severity fire, and a significant area was in an early-seral condition (grass, shrub, or seedling/sapling) or was recently burned and recovering (e.g., see the tables in app. 3). This is how forest habitats with complex structure and age classes continuously emerged on the landscape and were retained despite ongoing disturbances. In other ecoregions, where surface fire effects stemming from low- and mixed-severity fires were clearly dominant, fine-scale patterns in forest composition, structure, and tree age created a fine-scale mosaic of susceptibility to disturbance. This is how forest habitats with fine-scale structure and age classes continuously emerged on this landscape. The interplay of fine- and coarse-scale drivers (e.g., disturbance, topography, soils, and microclimate) across the regional landscape created fine-, meso-, and coarse-scale forest habitats with complex structure and age classes. In this way, local and regional landscapes, but not all stands or patches, were resilient.

Completely natural or historical landscape patterns cannot be the goal for current and future climate and landscape conditions. However, the past (e.g., HRV) can be an important guide to creating resilient forests. Knowledge of how forests and landscapes changed in response to disturbances and climate variation in the past is valuable for providing future forests and landscapes that have desired ecological patterns and processes. Where human-driven changes (e.g., fire suppression, grazing, past logging) have significantly altered forests relative to HRV, it will take significant inputs of human energy (i.e., ecologically motivated management) to create desired landscape conditions.

The size, diversity, and complexity of eastside mixed-conifer landscapes necessitate a prioritized approach to management. Though generalized, this concept, as well as those listed above, are critical to restoring resilience in these forests. These concepts have a strong ecological foundation, focus on restoring a more natural coupling of pattern and process, and can help managers create conditions that conserve options, are adaptable, and can be implemented with available skills and abilities common on the staff of a forest or district (or equivalent). Developing this characterization of a forest, although new to contemporary forest management, is not overly difficult and will enable a much more effective treatment strategy at the stand level where managers typically do their on-the-ground work.

Simple partitioning of the landscape into basic topographic positions is a straightforward method for parsing the forest into subunits with different inherent growth potential and disturbance regimes.

Topography as a Template for Landscape Heterogeneity

Previous research efforts have highlighted the predictive importance of topography (and more broadly, geomorphology) in landscape management (e.g., see Underwood et al. 2010). Studies and assessments from mixed-conifer forests (e.g., Hessburg et al. 2005, 2007; North et al. 2009; Taylor and Skinner 2003) have established that patterns of forest condition and fire behavior are strongly affected by topographic and physiographic features. Variability in soils also contributes to landscape heterogeneity, usually at a finer spatial scale. There are simple rules-of-thumb that can be gleaned from this work and applied to landscape management.

Simple partitioning of the landscape into basic topographic positions, such as drainage bottoms, ridgetops, or south- and north-facing slopes, is a straightforward method for parsing the forest into subunits with different inherent growth potential and disturbance regimes. Aspect patches of all sizes can be used to tailor treatments to the landscape; these can be readily generated in a geographical information system (GIS). There are now a number of easy-to-use GIS tools for doing this on any landscape using standard digital elevation model (DEM) data. The following provides a brief overview of some of the insights that topography provides.

South-facing aspects and ridges tended to burn more often and less severely than north-ridges, and fine fuels were typically conditioned for burning, even during average summer burn conditions. One can imagine that ridges, with their more exposed conditions and open grown forests, provided a rather elaborate network of natural fuel breaks owing to high lightning ignition frequency and limited fuel accumulations. This was typically not the case on north-facing aspects and in valleys, hence their reduced fire frequency. Although exceptions to these generalizations abound, landscape restoration can capitalize on these general tendencies without using a one-size-fits all approach. Instead, landscape restoration can apply a rule-of-thumb approach, as follows.

Managing south-facing aspects and ridgelines. In application, southerly aspects and ridges could be managed to support fire-tolerant species in clumped and gapped distributions by: (1) favoring very large-, large-, and medium-sized trees so that they occupied, e.g., 40 to 50 percent of the tree cover in the majority of these patches, and represented, e.g., 50 to 60 percent of their total area; (2) stocking to support a dominance of surface fire behavior stemming from low- and mixed-severity fires, and tree densities that support endemic but not epidemic bark beetle populations (i.e., resulting in the mortality of only a few trees over time); (3) maintaining tree species composition that strongly discourages the spread and intensification of root diseases, while allowing their presence; and (4) maintaining stocking on south slopes and ridges by low and free thinning and similar methods, especially by prescribed burning at regular intervals. Size class and canopy cover dominance would certainly vary from place to place, but ranges of conditions could be calibrated from historical reconstructions and modified as needed by incorporating expected climate changes. Prescribed burning and thinning activities could discriminate against the most severe dwarf mistletoe infestations. This would adequately mimic historical fire influence, but allow some of the most ecologically beneficial aspects of dwarf mistletoe infestation. Where fire- and drought-tolerant species are not dominant on south slopes and ridges, managers can regenerate them using methods that are best adapted to local site conditions.

Managing north aspects and valley bottoms. In application, north aspects and valley bottoms tend to support a mix of fire-tolerant and fire-intolerant tree species in relatively dense, often multilayered arrangements. Stocking can support surface and crown fire behavior stemming from mixed- and high- with occasional low-severity fires. Landscape patchiness of denser north aspect and valley bottom forest conditions can help constrain the frequency, severity, and duration of defoliator and bark beetle outbreaks. Ideal stocking on north slopes and valleys would reflect species compositions that encourage or allow the spread of root disease as a natural process; mixed fire-tolerant and fire-intolerant species compositions should adequately restrain the spread of root diseases while allowing ecologically beneficial fine-scale habitat and forage conditions stemming from root disease centers. Stocking on north slopes and valleys may be maintained by free selection thinning or similar methods, especially where density is quite high and layering is simple. Where fire-tolerant species are not present in north-facing slopes and valley bottom settings, they may be regenerated using methods that are adapted to local site conditions, depending on other local habitat constraints. Some patches may be dominated by drought- and fire-intolerant species without harm to the larger landscape.

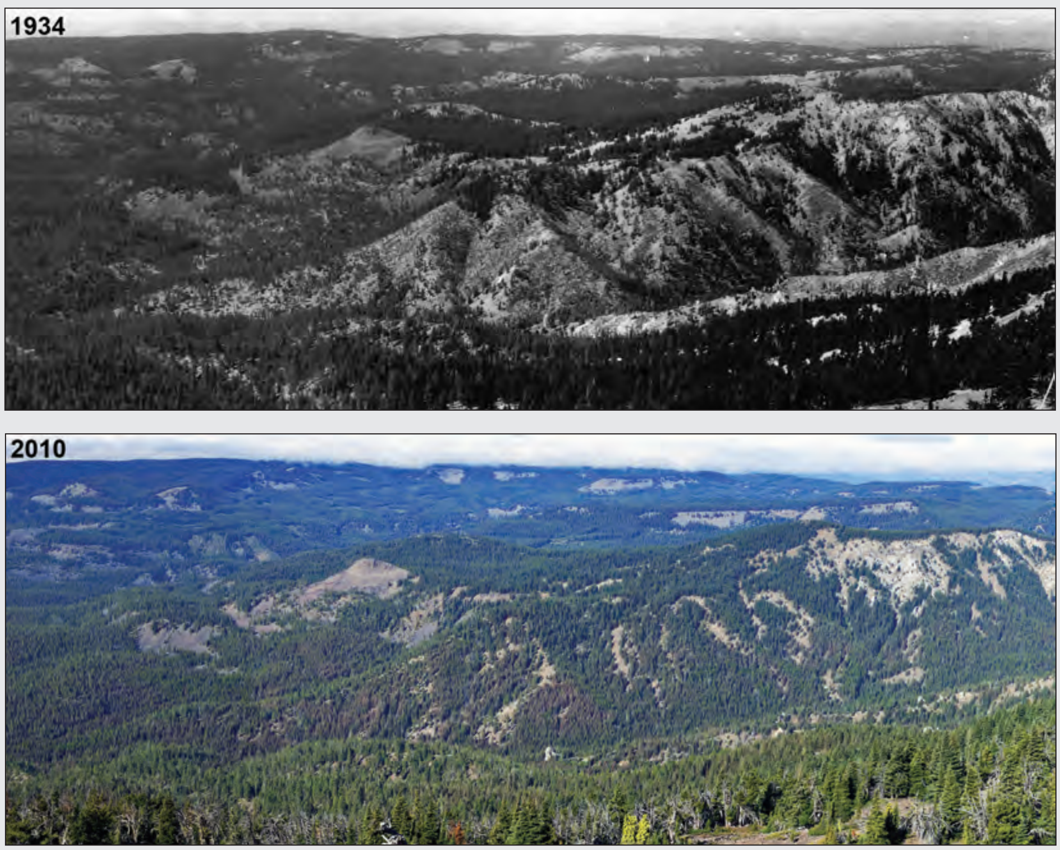
Forest types and their fire regimes are interconnected. Disturbance regimes and their variations in each forest type offered a regulating influence in adjacent but differing forest types. For this reason, no forest type and its disturbance regime and variation may be thought of in isolation. Some landscapes have significantly more north- or south-facing aspects than others;

(continued on next page)

(from page 27)

consequently, variations emerge that increase landscape complexity and these should be noted. In landscapes with more southerly aspects and ridges,north-facing aspects typically see more frequent fires and lower than typical severity. The converse is also true. These ideas can help shift the focus away from an overly simplistic topographically driven landscape template.

Figure 10 illustrates this intrinsic landscape pattern driven by topographic position with two photographs of Mission Peak on the Wenatchee National Forest. The photo from 1934 shows very little tree growth on the south-facing slopes and ridgetops with dense forest on the north-facing slopes and drainage bottoms. Because of fire exclusion, the forests have filled in as shown by the 2010 photo.



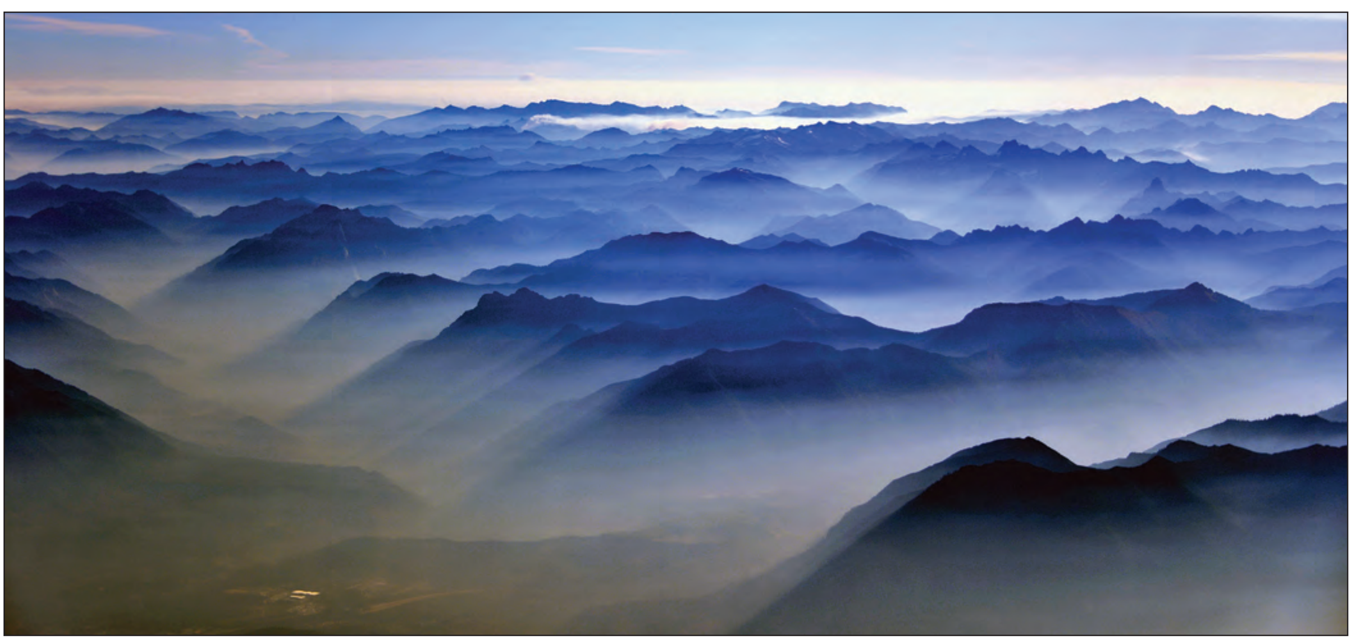
John Marshall

Figure 10—The Mission Peak area on the Okanogan-Wenatchee National Forest; a comparison between 1934 and 2010 of forest density and structure.

Chapter 4—Scientific Foundations

Ecological Composition, Patterns, and Processes Prior to Euro-American Settlement (Before About 1850)

The mountains of Oregon and Washington exert strong orographic control on climate, vegetation, disturbance, and land use across the region. Steep precipitation and temperature gradients have a significant influence on the vegetation east of the Cascade Divide (Franklin and Dyrness 1988) (fig. 11). Moist mixed-conifer (MMC) forest patches frequently occur within a broader continuum of mixed-conifer forest types (fig. 12) and within topographic locations juxtaposed with mixed conifer or ponderosa pine vegetation types, or grassland and shrubland patches (fig. 13) (Hessburg et al. 1999a, 2000b; Spies et al. 2006). This mosaic of landscape composition and physiognomic conditions can alter the disturbance regime as well as the functioning of these patches across broader spatial scales. For example, the frequent fire regimes of grassland and shrubland influence adjacent forests by increasing their fire frequency. This occurs because grass and shrub patches often function as "conveyor belts," readily spreading wildfire to adjacent patches. Likewise, dry mixed-conifer patches that experience frequent surface fires can also influence adjacent MMC patches. However, when MMC forest is surrounded by cold or wet forest types, the fire regime may be influenced by this context; and fires may tend to be less frequent and more severe.



B. Nguyen

Figure 11—At the regional scale, the moisture gradient is structured by large-scale topographic features, such as the Cascade Mountains, and a prevailing south-southwest storm track.

Illustration by Stevie Ruda

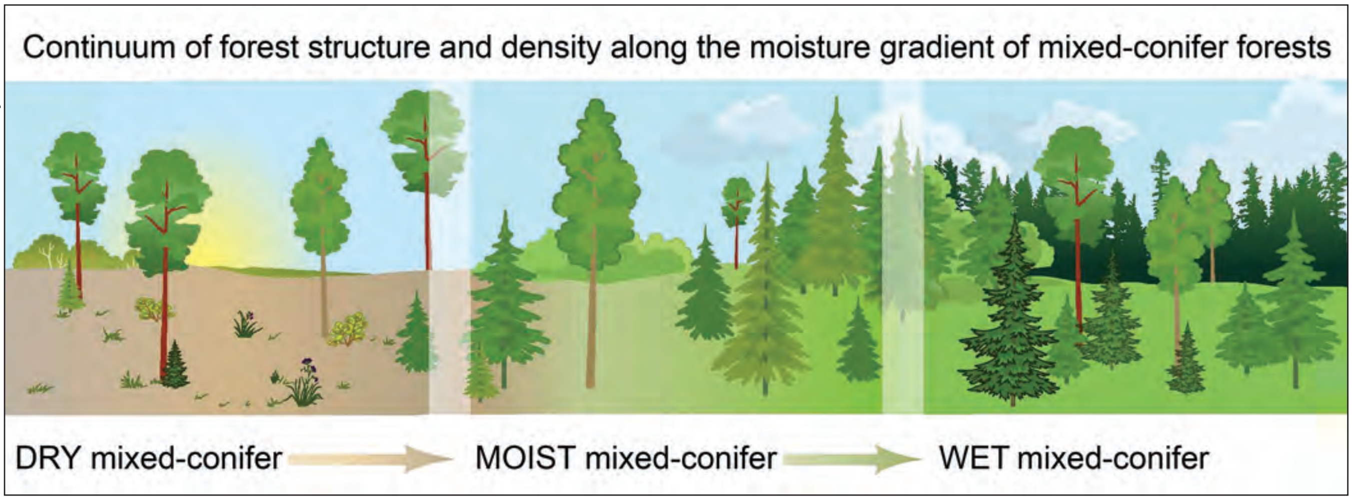


Figure 12—A simple schematic illustrating a typical landscape gradient of different forms of mixed conifer, from the dry type through the moist types to the wet types.

Miles Hemstrom

Figure 13—A common gradient of grasslands/shrublands to dry forests and moist forests. Blue Mountains near Pendleton, Oregon.

Where Is the MMC Forest Located?

No standardized, peer-reviewed maps of the potential vegetation subseries covering the MMC forest are published for the entire region. The Region 6 Area Ecology Program has developed plant association classifications and maps for individual subregion ecology areas, but the names and environmental conditions of the plant associations differ. Currently the region is using the vegetation classification represented in the Integrated Landscape Assessment Program (a partnership jointly managed by the Forest Service Pacific Northwest Research Station and Oregon State University's Institute of Natural Resources) for defining MMC forest distribution and abundance (fig. 7, table 2). The map we presented is intended solely for the purpose of initial regional-scale planning and analysis and has limited spatial accuracy for project planning. Local classifications and maps that are spatially accurate at a patch scale are more appropriate for defining the distribution of the MMC forest for project level planning and management.

Local classifications and maps that are spatially accurate at a patch scale are more appropriate for defining the distribution of the moist mixed-conifer forest for project level planning and management.

Other classification strategies also exist (e.g., Henderson et al. 2011, Simpson 2007) for vegetation zones that have resulted in different depictions of this forest type across the region. Confounding this issue, national forests often have their own local maps of potential vegetation types (PVTs) that they use for management, and the scale of these maps varies from forest to forest. These maps do not correspond to the regional maps shown here. We suggest that any future regionwide strategy for MMC forest be underpinned with a single map that uses a regionally coherent classification standard and mapping protocol. The map would accurately identify the major areas that support the MMC type, and its relation to other types. This regional map would enable cross-forest data sharing and landscape assessment and management coordination. Local forest-level maps would then be needed to estimate departure from historical dynamic and threats to forest resilience.

Mixed-Conifer Plant Association Classification

The plant associations of series associated with mixed-conifer forest can be placed into different moisture/fire regime groups using published and unpublished information (table 2). We present these as a first approximation for defining the occurrence of different mixed-conifer-disturbance regime types and aid to communication. As stated elsewhere, potential vegetation types (e.g., series, subseries, and associations) are only approximate indicators of fire regime, a concept which is best applied at landscapes scales (larger areas that are mosaics of local climate, topography, soils and vegetation).

(continued on next page)

(from page 31)

We grouped plant associations of the grand fir, Douglas-fir, and Shasta red fir (*Abies magnifica* A. Murray bis) series into subgroups based on moisture and historical fire regime: dry-low severity, moist-dry-low- to mixed-severity, moist-mixed severity, and moist to wet-high-severity (table 2). Associations in the subalpine fir and Shasta red fir series were included if ponderosa pine is an overstory component in the series or if plant association guides mentioned that low-severity fires were part of the historical fire regime. We assigned plant associations to a mixed-conifer moisture-fire regime group based on published information and expert opinion about the historical disturbance regime, environmental setting, and species composition. Plant associations in dry environments where historical fire regimes were predominately low severity and ponderosa pine was common in both the overstory and understory were classified as dry mixed-conifer.

Plant associations were classified as MMC forest if the following conditions were met: (1) mixed-severity fire regime; (2) intermediate environmental setting; and (3) Douglas-fir or grand fir are the most common understory tree species and are co-dominant in the overstory with ponderosa pine or other early-seral dominant, or those tree species are predicted to be dominant in the overstory and understory community based on the primary indicator plant species. Plant associations that could not be classified as dry or moist because of broad variability in historical disturbance regimes and species composition were classified as moist-dry. We expect the variability in the moist-dry group represents associations where dry and MMC forest intermingle on the landscape. The moist to wet classification was given to all plant associations with an infrequent, high-severity fire regime and species composition that were inconsistent with our definition of eastside mixed-conifer forest. Some of these associations would be classified as “moist” in some subregions but their historical fire regimes are still dominated by high-severity fire at relatively longer intervals. In these latter associations, ponderosa pine was usually absent in all strata; early-seral species including lodgepole pine, Douglas-fir and western larch were often even-aged; and shade-tolerant fire-intolerant species were common in the overstory. Plant associations in these moist to wet environments may still merit restoration due to past timber management activities, but fire does not play the same role as it does in dry and MMC forests, and it is unlikely that fire exclusion has modified structure, composition in these wetter types.

Table 2—Mixed-conifer plant associations grouped by moisture and approximate fire regime (app. 2)

National forest (NF)/region[a]	Vegetation series/ plant association	Extent[b]	Species composition[c]	Source[d]
Dry with low-severity fire*—characterized by low-severity fires and relatively short return intervals of less than 35 years. Natural fire regimes in order of prevalence are I, III A, III B, and IV A.*				
Blue and Ochoco Mountains	Douglas-fir/elk sedge	Widespread	PSME, PIPO, ABGR	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains	Grand fir/elk sedge	Widespread	ABGR, PIPO, PSME	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains	Grand fir/pinegrass	Widespread	ABGR, PIPO, PSME, LAOC, PICO	Johnson and Clausnitzer 1992 Johnson and Simon 1987
Blue and Ochoco Mountains	Grand fir/birchleaf spirea	Intermediate	ABGR, PSME, LAOC, PIPO	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains	Douglas-fir/mountain snowberry	Minor	PSME, PIPO, JUOC	Johnson and Clausnitzer 1992 Johnson and Simon 1987
Blue and Ochoco Mountains	Douglas-fir/common snowberry	Widespread	PSME, PIPO, LAOC, JUOC	Johnson and Clausnitzer 1992 Johnson and Simon 1987
Blue and Ochoco Mountains	Douglas-fir/pinegrass	Widespread	PSME, PIPO, ABGR	Johnson and Clausnitzer 1992 Johnson and Simon 1987
Blue and Ochoco Mountains	Douglas-fir/birchleaf spirea	Intermediate	PSME,PIPO	Johnson and Simon 1987
Colville NF	Ponderosa pine–Douglas-fir/bluebunch wheatgrass	Minor	PIPO, PSME	Williams et al. 1995
Colville NF	Douglas-fir/snowberry	Intermediate	PSME, PIPO, LAOC	Williams et al. 1995
Colville NF	Douglas-fir/mountain snowberry	Minor	PSME, PIPO, LAOC	Williams et al. 1995
Colville NF	Douglas-fir/pinegrass	Widespread	PSME, LAOC, PIPO	Williams et al. 1995
Deschutes, Mount Hood NFs	Douglas-fir/bitterbrush	Intermediate	PIPO, PSME, CADE, JUOC	Simpson 2007
Deschutes, Mount Hood NFs	Douglas-fir/greenleaf manzanita	Intermediate	PIPO, PSME, CADE	Simpson 2007
Fremont-Winema, Mount Hood NFs	Douglas-fir/common snowberry	Intermediate	PSME, PIPO, QUGA, JUOC	Simpson 2007
Fremont-Winema, Mount Hood NFs	Douglas-fir/mahala mat	Intermediate	PIPO, PSME, CADE	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/pinegrass	Minor (in east Cascades)	PIPO, PSME, ABGR	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/common snowberry	Widespread	PIPO, ABGR, PSME	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/greenleaf manzanita	Widespread	PIPO, ABGR, PICO	Simpson 2007
Fremont-Winema NF	Grand fir/wooly wyethia	Minor	PIPO, ABGR, PICO	Simpson 2007
Fremont-Winema NF	Grand fir/mahala mat	Intermediate	PIPO, ABGR, CADE, JUOC	Simpson 2007
Mount Hood NF	Douglas-fir/elk sedge	Widespread	PIPO, PSME, QUGA, ABGR	Topick et al. 1988
Mount Hood NF	Douglas-fir/western fescue	Intermediate	PSME, PIPO, ABGR, QUGA	Topick et al. 1988
Mount Hood NF	Douglas-fir/snowberry	Intermediate	PIPO, PSME, ABGR	Topick et al. 1988
Mount Hood NF	Douglas-fir/pinemat manzanita	Minor	PIPO, PSME, ABGR	Topick et al. 1988
Mount Hood NF	Grand fir/elk sedge	Widespread	PSME, ABGR, PIPO	Topick et al. 1988
Mount Hood NF	Grand fir/snowberry	Most widespread	PSME, ABGR, PIPO	Topick et al. 1988
Okanogan NF	Ponderosa pine–Douglas-fir/bluebunch wheatgrass	Widespread	PIPO, PSME	Williams and Lillybridge 1983

Table 2—Mixed-conifer plant associations grouped by moisture and approximate fire regime (app. 2) (continued)

National forest (NF)/region[a]	Vegetation series/ plant association	Extent[b]	Species composition[c]	Source[d]
Okanogan NF	Douglas-fir/pinegrass	Most widespread	PSME, LAOC, PIPO, PICO, POTR5	Williams and Lillybridge 1983
Okanogan NF	Douglas-fir/bearberry	Intermediate	PSME, PICO, PIPO, LAOC	Williams and Lillybridge 1983
Okanogan NF	Douglas-fir/mountain snowberry	Intermediate	PSME, PIPO, LAOC	Williams and Lillybridge 1983
Wenatchee NF	Oregon white oak/bluebunch wheatgrass	Minor	QUGA, PIPO	Lillybridge et al. 1995
Wenatchee NF	Oregon white oak/pinegrass–elk sedge	Minor	QUGA, PIPO, PSME	Lillybridge et al. 1995
Wenatchee NF	Oregon white oak/California hazel–common snowberry	Minor	QUGA, PIPO, PSME, POTR5	Lillybridge et al. 1995
Wenatchee NF	Ponderosa pine/pinegrass–bluebunch wheatgrass	Minor	PIPO, PSME	Lillybridge et al. 1995
Wenatchee NF	Ponderosa pine/bitterbrush–bluebunch wheatgrass	Intermediate	PIPO, PSME	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/elk sedge	Intermediate	PSME, PIPO	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/bitterbrush–bluebunch wheatgrass	Widespread	PIPO, PSME	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/bitterbrush–pinegrass	Widespread	PIPO, PSME	Lillybridge et al 1995
Wenatchee NF	Douglas-fir/bluebunch wheatgrass	Widespread	PSME, PIPO	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/pinegrass–bluebunch wheatgrass	Intermediate	PIPO, PSME	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/pinegrass	widespread	PSME, PIPO, PICO, LAOC	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/snowberry–bluebunch wheatgrass	Intermediate	PSME, PIPO	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/snowberry–pinegrass	Widespread	PSME, PIPO	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/snowberry	Widespread	PSME, PIPO	Lillybridge et al 1995
Wenatchee NF	Douglas-fir/mountain snowberry	Intermediate	PIPO, PSME	Lillybridge et al. 1995
Wenatchee NF	Douglas-fir/shiny-leaf spirea–pinegrass	Widespread	PSME, PIPO	Lillybridge et al. 1995
Wenatchee NF	Grand fir/pinegrass	Most widespread	PSME, PIPO, ABGR, PICO	Lillybridge et al. 1995
Wenatchee NF	Grand fir/mountain snowberry	Minor	PSME, PIPO, ABGR	Lillybridge et al. 1995
Wenatchee NF	Grand fir/pinemat manzanita	Widespread	PSME, PIPO, ABGR, PICO, LAOC	Lillybridge et al. 1995
Warm Springs Indian Reservation	Ponderosa pine–Douglas-fir/bitterbrush	Intermediate	PSME, PIPO CADE	Marsh et. al 1987
Warm Springs Indian Reservation	Ponderosa pine–Douglas-fir/ bitterbrush-ceanothus	Widespread	PIPO, PSME, CADE	Marsh et. al 1987
Warm Springs Indian Reservation	Ponderosa pine–Douglas-fir/snowberry	Intermediate	PSME, PIPO CADE	Marsh et. al 1987
Warm Springs Indian Reservation	Ponderosa pine–Douglas-fir/bitterbrush (Mutton)	Intermediate	PIPO, JUOC, QUGA	Marsh et. al 1987
Warm Springs Indian Reservation	Ponderosa pine–Douglas-fir/snowberry (Mutton)	Intermediate	PSME, PIPO CADE	Marsh et. al 1987

Table 2—Mixed-conifer plant associations grouped by moisture and approximate fire regime (app. 2) (continued)

National forest (NF)/region[a]	Vegetation series/ plant association	Extent[b]	Species composition[c]	Source[d]
Warm Springs Indian Reservation	Ponderosa pine–Douglas-fir/greenleaf manzanita–ceanothus (Mutton)	Intermediate	PSME, PIPO, CADE, QUGA	Marsh et. al 1987
Warm Springs Indian Reservation	Ponderosa pine–Douglas-fir/snowberry (Mutton)	Intermediate	PSME, PIPO QUGA	Marsh et. al 1987
Warm Springs Indian Reservation	Ponderosa pine–Douglas-fir/prairie smoke avens–mule's ears (Mutton)	Intermediate	PIPO, PSME, JUOC, QUGA	Marsh et. al 1987
Moist-dry with low- to mixed-severity fire*—Characterized by low to moderate-severity fire with intervals intermediate between the dry and moist group. Large high-severity fires rarely occur. Natural fire regimes in order of prevalence are III A, III B, I, IIIC, and IV A.*				
Blue and Ochoco Mountains	Douglas-fir/oceanspray	Intermediate	PSME, PIPO	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains	Grand fir/grouse huckleberry	Intermediate	ABGR, LAOC, PSME, LAOC, PICO, PIPO	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains				
Wallowa-Snake Province	Douglas-fir/Rocky Mountain maple–mallow ninebark	Intermediate	PSME, PIPO	Johnson and Simon 1987
Blue and Ochoco Mountains				
Wallowa-Snake Province	Douglas-fir/big huckleberry	Intermediate	PSME, PIPO, LAOC	Johnson and Clausnitzer 1992 Johnson and Simon 1987
Blue and Ochoco Mountains; Strawberry Mountains	Douglas-fir/pinemat manzanita/ elk sedge	Minor	PSME, PIMO, ABLA	Johnson 2004
Blue and Ochoco Mountains				
Wallowa-Snake Province	Douglas-fir /mallow ninebark	Widespread	PSME, PIPO, LAOC	Johnson and Clausnitzer 1992
Colville National Forest	Douglas-fir/ninebark	Most widespread	PSME, PIPO	Williams et al. 1995
Colville National Forest	Douglas-fir/ninebark–twinflower	Widespread	PSME, LAOC, PIPO	Williams et al. 1995
Colville National Forest	Douglas-fir/big huckleberry	Intermediate	PSME, LAOC, PICO	Williams et al. 1995
Colville National Forest	Douglas-fir/dwarf huckleberry	Minor	PSME, PICO, LAOC,PIPO, PIEN	Williams et al. 1995
Colville National Forest	Grand fir/ninebark	Widespread	PSME, LAOC, PIPO, ABGR	Williams et al. 1995
Colville National Forest	Grand fir/dwarf huckleberry	Minor	PICO, PSME, LAOC, ABGR	Williams et al. 1995
Deschutes, Mount Hood NFs	Douglas-fir/chinquapin	Widespread	PSME, PIPO, CADE, PILA	Simpson 2007
Deschutes, Mount Hood NFs	Douglas-fir/oceanspray	Minor	PSME, PIPO, QUGA, JUOC	Simpson 2007
Deschutes, Mount Hood NFs	Grand fir/oceanspray	Intermediate	PSME, PIPO, ABGR	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Douglas-fir/starflower	Widespread	PSME, PIPO, CADE	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Douglas-fir/creeping snowberry	Intermediate	PSME, PIPO, CADE, QUGA	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Douglas-fir/prince's pine	Intermediate	PSME, PIPO, CADE	Simpson 2007
Fremont-Winema NF	Grand fir/creeping snowberry	Intermediate	PIPO, ABGR, PICO, PSME	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/golden chinquapin	Widespread	PSME, PIPO, ABGR	Simpson 2007

Table 2—Mixed-conifer plant associations grouped by moisture and approximate fire regime (app. 2) (continued)

National forest (NF)/region[a]	Vegetation series/ plant association	Extent[b]	Species composition[c]	Source[d]
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/western starflower	Widespread	PSME, ABGR, PIPO	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/common prince's pine	Widespread	PIPO, ABGR, PICO, PSME	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/starry false-solomonseal	Widespread	PIPO, ABGR, PICO	Simpson 2007
Mount Hood NF	Douglas-fir/oceanspray/elk sedge	Widespread	PSME, PIPO, QUGA, ABGR	Topick et al. 1988
Mount Hood NF	Grand fir/starflower	Widespread	PSME, ABGR, PIPO	Topick et al. 1988
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/long-stolen sedge	Intermediate	PIPO, PICO, ABGR	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/pinemat manzanita	Intermediate	ABCO, PICO, PIPO, PIMO	Simpson 2007
Mount Hood NF	Grand fir/chinkquapin	Minor	ABGR, PSME, PICO, LAOC, PIPO	Topick et al. 1988
Okanogan NF	Douglas-fir/ninebark	Minor	PSME, LAOC, PICO, PIPO	Williams and Lillybridge 1983
Okanogan NF	Douglas-fir/boxwood	Minor	PSME, PIPO, POTR5	Williams and Lillybridge 1983
Okanogan NF	Douglas-fir/huckleberry	Intermediate	PSME, LAOC, PICO, PIPO	Williams and Lillybridge 1983
Okanogan NF	Subalpine fir/pinegrass	Intermediate	PSME, ABLA,PICO, LAOC, PIPO	Williams and Lillybridge 1983
Wenatchee NF	Douglas-fir/huckleberry (three minor huckleberry associations)	Minor	PSME, LAOC, PICO, PIPO	Lillybridge et al. 1995
Wenatchee NF	Grand fir/pinegrass– lupine	Intermediate	PSME, PIPO, ABGR, PICO, LAOC	Lillybridge et al. 1995
Wenatchee NF	Grand fir/heartleaf arnica	Intermediate	PSME, ABGR, LAOC, PIPO, PIEN	Lillybridge et al. 1995
Wenatchee NF	Grand fir/oceanspray–pinegrass	Intermediate	PSME, ABGR, PIPO, LAOC	Lillybridge et al. 1995
Wenatchee NF	Grand fir/Cascade Oregon grape–pinegrass	Intermediate	PSME, ABGR, PICO, LAOC, PIPO	Lillybridge et al. 1995
Wenatchee NF	Grand fir/common snowberry–pinegrass	Intermediate	PSME, ABGR, PIPO	Lillybridge et al. 1995
Warm Springs Indian Reservation	Mixed conifer/snowberry/elk sedge	Intermediate	PIPO, PSME, ABGR	Marsh et. al 1987
Warm Springs Indian Reservation	Mixed conifer/ceanothus	Widespread	PIPO, PSME, ABGR, CADE	Marsh et. al 1987
Warm Springs Indian Reservation	Mixed conifer/snowberry	Widespread	PSME, PIPO ABGR	Marsh et. al 1987
Warm Springs Indian Reservation	Mixed conifer/boxwood	Intermediate	ABGR, PSME, PIPO, LAOC	Marsh et. al 1987
Warm Springs Indian Reservation	Grand fir/boxwood	Intermediate	ABGR, PSME, PIPO, LAOC	Marsh et. al 1987
Moist with mixed-severity fire*—Characterized by moderate to high-severity fire with lesser amounts of low-severity fire in comparison to the moist-dry group. Fire return intervals are usually longer than 50 years. Natural fire regimes in order of prevalence are III B, III C, III A, IV B, and IV A.*				
Blue and Ochoco Mountains				
Wallowa-Snake Province	Grand fir/big huckleberry	Widespread	ABGR, LAOC, PSME, PIEN, ABLA, PIPO, PICO	Johnson and Clausnitzer 1992
Johnson and Simon 1987				
Wallowa-Snake Province	Grand fir/Rocky Mountain maple	Intermediate	ABGR, PIEN, PSME, LAOC, PIMO	Johnson and Clausnitzer 1992 Johnson and Simon 1987

Table 2—Mixed-conifer plant associations grouped by moisture and approximate fire regime (app. 2) (continued)

National forest (NF)/region[a]	Vegetation series/ plant association	Extent[b]	Species composition[c]	Source[d]
Blue and Ochoco Mountains	Grand fir/Columbia brome	Intermediate	ABGR, PSME, LAOC, PIEN	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains	Grand fir/grouse huckleberry–twinflower	Intermediate	ABGR, PIEN, LAOC, PSME	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains				
Wallowa-Snake Province	Grand fir/twinflower	Widespread	ABGR, PIEN, PSME, LAOC, PICO, ABLA, PIPO, PIMO	Johnson and Clausnitzer 1992 Johnson and Simon 1987
Wallowa-Snake Province	Grand fir/queenscup beadlilly	Widespread	ABGR, LAOC, PSME, PIEN	Johnson and Clausnitzer 1992
Johnson and Simon 1987				
Colville National Forest	Grand fir/big huckleberry/queenscup beadlilly	Intermediate	PSME, LAOC, ABGR, PICO,	Williams et al. 1995
Fremont-Winema, Deschutes NFs	Mountain hemlock/golden chinquapin	Widespread	ABCO, ABPR, PICO, PIMO, PIPO, PSME, TSME	Simpson 2007
Fremont-Winema, Deschutes NFs	Shasta red fir/golden chinquapin	Intermediate	ABPR, PICO, PILA, PIMO, PIPO	Simpson 2007
Fremont-Winema, Deschutes NFs	Shasta red fir/common prince's pine	Intermediate	ABPR, PICO, PIMO, PIPO	Simpson 2007
Fremont-Winema, Deschutes NFs	Shasta red fir/pinemat manzanita	Intermediate	ABPR, PICO, PIMO, PIPO	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/twinflower	Widespread	PSME, ABGR, PIPO	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/queenscup beadlily	Widespread	ABGR, PSME, PIPO, TSME	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	Grand fir/vanilla leaf	Widespread	ABGR, PSME, PIPO, CADE, TSME	Simpson 2007
Fremont-Winema, Deschutes, Mount Hood NFs	grand fir/wild ginger	Intermediate	PSME, ABGR, PIPO, ABMA/ABPR	Simpson 2007
Mount Hood NF	Grand fir/twinflower	Widespread	ABGR, PSME, PIPO	Topick et al. 1988
Mount Hood NF	Grand fir/vanilla leaf	Widespread	PSME, ABGR, PIPO, PICO, LAOC, TSHE, ABPR	Topick et al. 1988
Mount Hood NF	Grand fir/skunk leaved polemonium	widespread	ABGR, PSME, PICO, LAOC, PIEN, PIPO	Topick et al. 1988
Okanogan NF	Subalpine fir/vaccinium	Widespread	LAOC, PSME, PICO, PIEN, ABLA	Williams and Lillybridge 1983
Wenatchee NF	Grand fir/shiny-leaf spirea–bracken fern	Intermediate	PSME, PIPO, ABGR, PIMO	Lillybridge et al. 1995
Wenatchee NF	Grand fir/vine maple–queencup beadlily	Intermediate	PSME, ABGR, PIPO	Lillybridge et al. 1995
Wenatchee NF	Grand fir/Cascade Oregon-grape	Widespread	PSME, ABGR, LAOC, PIPO	Lillybridge et al. 1995
Wenatchee NF	Grand fir/vanilla leaf	Intermediate	ABGR, PSME, LAOC, PIMO, PIPO	Lillybridge et al. 1995
Wenatchee NF	Grand fir/vine maple–prince's pine	Intermediate	PSME, ABGR, PIPO, PICO,	Lillybridge et al. 1995
Warm Springs Indian Reservation	Grand fir/snowberry	Widespread	ABGR, PSME, PIPO,	Marsh et. al 1987
Warm Springs Indian Reservation	Grand fir/vine maple	Widespread	PSME, ABGR, PIPO	Marsh et. al 1987
Warm Springs Indian Reservation	Grand fir/big leaf huckleberry	Widespread	ABGR, PSME, LAOC, PIMO	Marsh et. al 1987
Warm Springs Indian Reservation	Grand fir–Lodgepole pine/ boxwood/ pinegrass	Intermediate	PICO, ABGR PIPO, ABLA,	Marsh et. al 1987

Table 2—Mixed-conifer plant associations grouped by moisture and approximate fire regime (app. 2) (continued)

National forest (NF)/region[a]	Vegetation series/ plant association	Extent[b]	Species composition[c]	Source[d]
Moist to wet with high-severity fire*—Higher severity fire tends to dominate in most events and return intervals are usually greater than 100 years. Natural fire regimes in order of prevalence are III C and IV B.*				
Blue and Ochoco Mountains	Grand fir/oakfern	Minor	ABGR, PIEN, PSME, LAOC, ACGL, TABR	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains	Grand fir/sword fern–ginger	Intermediate	ABGR, PSME, LAOC, PIEN	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains				
Wallowa-Snake Province	Grand fir/Pacific yew/ queen's cup beadlilly	Intermediate	ABGR, PIEN, PSME, LAOC, TABR	Johnson and Clausnitzer 1992
Johnson and Simon 1987				
Blue and Ochoco Mountains	Grand fir/Pacific yew/ twinflower	minor	ABGR, PSME, LAOC, PIEN, TABR	Johnson and Clausnitzer 1992
Blue and Ochoco Mountains	Grand fir/false bugbane	Intermediate	ABGR, PIEN, ABLA, LAOC, PICO, TABR	Johnson and Clausnitzer 1992
Colville National Forest	Grand fir/Douglas maple/queencup beadlilly	Intermediate	PSME, LAOC, ABGR	Williams et al. 1995
Mount Hood National Forest	Grand fir–Engelmann spruce/ starry solomonplume	Minor	ABGR, PIEN, PSME, PICO	Topick et al. 1988
Mount Hood NF	Grand fir/vanilla leaf–vine maple	Widespread	PSME, PIPO, ABGR, CONU	Topick et al. 1988

Note: This table is intended as an interim classification subject to further revision as new information becomes available. Note that terms used to describe climatic environment are relative terms and have different definitions among regions. Therefore, this regional classification may differ from local classifications.

[a] National forest/region = subregion of the plant association.

[b] Extent = geographic distribution and approximate area of a plant association relative to other associations within the region.

[c] Species composition = all tree species recorded in each association, ordered by prominence within each association.
ABGR = *Abies grandis* (grand fir); ABLA = *Abies lasiocarpa* (subalpine fir); ABPR = *Abies procera* (noble fir);
ACGL = *Acer glabrum* (Rocky Mountain maple); CADE = *Calocedrus decurrens* (incense cedar); CONU = *Cornus nutallii* (Pacific dogwood); JUOC = *Juniperus occidentalis* (western juniper); LAOC = *Larix occidentalis* (western larch);
PICO = *Pinus contorta* (lodgepole pine); PIEN = *Picea engelmannii* (Engelmann spruce; PILA = *Pinus lambertiana* (sugar pine); PIMO = *Pinus monticola* (western white pine); PIPO = *Pinus ponderosa* (ponderosa pine); PSME = *Pseudotsuga menziesii* (Douglas-fir); QUGA = *Quercus garryana* (Oregon white oak); TABR = *Taxus brevifolia* (Pacific yew); TSHE = *Tsuga heterophylla* (western hemlock); TSME = *Tsuga mertensiana* (mountain hemlock).

[d] Source = citation for plant association description.

Vegetation Classification and Disturbance Regimes

Effective restoration depends on having an accurate classification of vegetation and its disturbance regime. There are various schemas for classifying vegetation for purposes of ecological stratification, including current vegetation, historical vegetation, and potential natural vegetation (PNV). PNV in application is organized into a taxonomic hierarchy consisting of series, subseries, and plant associations, which are named for dominant overstory and undergrowth plants (Powell 2007). The standard system of vegetation classification and mapping used by the Forest Service and other federal agencies is based on this taxonomic hierarchy. Mixed conifer is composed of grand fir, white fir and Douglas-fir series, but the distinction of dry, moist and wet mixed-conifer types (i.e., disturbance regimes) requires the use of lower levels in the hierarchy, including subseries and groups of plant associations. We provide a first approximation for how associations could be grouped into four moisture/disturbance regime classes, (equivalent to subseries; table 2), but caution that this classification is interim and subject to revision as more knowledge becomes available. In this synthesis we use the term PVT in a general way to refer to any potential natural vegetation class regardless of level in the hierarchy (series, subseries, and plant associations).

A PVT is the native, late-successional (or "climax") plant community that reflects the regional climate and the dominant plant species of an area that would occur on a site in the absence of disturbance (Pfister and Arno 1980). This approach assumes that disturbances still occur, but that the composition of the late-successional community occurring when disturbance is absent is a reasonable indicator of the biophysical environment. Hence, PVTs may be considered as approximate surrogates for the environmental conditions of a site (in terms of moisture, solar, nutrient, and temperate regimes) that exert some control on productivity, habitat potential, regeneration rates, and the frequency and severity of disturbance regimes.

Concepts of orderly succession and climax vegetation are questionable in mixed-conifer and other fire-prone eastside environments (O'Hara et al. 1996). These forests actually exhibit non-equilibrium dynamics, where the potential climax conditions were typically never realized due to myriad interacting and multi-scale disturbance processes, especially fire, that both advanced and retarded succession. Potential natural vegetation classifications can also take into account the reality that vegetation structure and composition is altered and controlled by periodic wildfire (Winthers et al. 2005). Much of the dry and MMC forests in Oregon and Washington would belong to these "fire climax" potential natural vegetation types. Despite these complexities of nature and the limits of our conceptual models, PVTs can be a useful starting place for identifying the ecological potential, fire regime, and restoration needs for an area.

At finer scales (below subseries) the plant association is a unit defined on the basis of a characteristic range of species composition, diagnostic species occurrence, habitat conditions, and physiognomy (USDA FS 2005). A number of plant associations are included in the MMC type and some grade into the drier or wetter mixed-conifer types. See the list in table 2 for an interim assignment of plant associations to our mixed-conifer moisture/disturbance regime types.

Tree species of the MMC forest are characterized as either early-, mid-, or late-successional (late-seral), and as either shade-intolerant or tolerant, respectively. The most common early-seral forest tree species are western larch (*Larix occidentalis* Nutt.), ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson), lodgepole pine (*Pinus contorta* Douglas ex Loudon), western white pine (*Pinus monticola* Douglas ex D. Don), trembling (quaking) aspen (*Populus tremuloides* Michx.; especially where a seasonally high water table exists), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), and mixed conditions of these types. With the exception of Douglas-fir, each of these species is shade-intolerant, all readily regenerate after fire, and western larch, ponderosa pine, and Douglas-fir can become exceptionally fire-tolerant as they advance in age, owing to an ever thickening bark (Agee 1993). Dominant late-seral tree species are grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.; in central and southern Oregon), and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), where local environmental conditions (e.g., cold-air drainage or pockets) occur that will promote dominance of these species. All are shade-tolerant and fire-intolerant. Exceptions can occasionally be found in mature grand fir and white fir, which may also possess a thick bark and occasionally occur in park-like stands (Hessburg et al. 1999a, 2000a).

Under native fire regimes, the moist mixed-conifer forest would have displayed highly variable structure and composition regionally, making it among the most diverse and complex of all forest types.

Under native fire regimes, the MMC forest would have displayed highly variable structure and composition regionally, making it among the most diverse and complex of all forest types (e.g., see the tables in app. 3 for the Blue Mountains province). The MMC forest landscape included a wide range of cover type and structural class combinations that would derive from a mixture of low- to mixed-severity fires with lesser amounts of high-severity fire in the mix (e.g., see app. 3, Hessburg et al. 2007, Perry et al. 2011). Recent studies in MMC forest in central Oregon suggest that the amount of low- and mixed-severity fire and forest structure (e.g., density) and tree layer composition in pre-Euro-American times may not have differed much from nearby dry mixed-conifer PVT (Hagmann et al. 2013, Merschel 2012).

The PVT is not the same as the current or actual vegetation, which may exist in an early-, mid-, or late-successional state or in a nonnative and human-altered state (e.g., agricultural patch, urban or rural development area, pasture, or road right-of-way). The difference between the current and potential vegetation types can be important to restoration and should be known before undertaking restoration activities. The current vegetation type needs to be considered to determine which restoration trajectories are most probable, given local site conditions, landscape context, disturbance history, and potential climate change effects.

Finally, with changing climate, PVT classifications based on the relationship of potential late-successional communities to their environment will be related to current and recent past climate conditions, but not to the future climate. Consequently, managers are best served by using a variety of sources of information about a site, including potential climate change effects, when making management decisions. Individual trees or populations exposed to climate conditions outside their climate niches may be maladapted, resulting in compromised productivity and increased vulnerability to disturbance. In some locations the future will not resemble the past with respect to potential vegetation, and assemblages of tree species that we currently recognize may be quite different in the future. This will be especially true where there is a rapid change in environment and vegetation (e.g., current ecotone edges). We suggest that a current PVT map be developed to the full extent of subwatersheds of each national forest, using regionally standardized protocols for PVT definitions and mapping.

Although it is generally accepted that the PVTs differ in their disturbance regimes, only a handful of dendroecological and landscape reconstruction studies have characterized variability in wildfire regimes in eastern Washington and Oregon (Agee 1994; Hessburg et al. 2007; Heyerdahl et al. 2001, 2013; Perry et al. 2011; Wright and Agee 2004), and have been conducted in the eastern Washington Cascades and Blue Mountains. More work is needed to better understand fire regime variability and relations with physiography. Studies of dry and MMC forests, which were characterized by highly variable low-, mixed-, and high-severity fire regimes, have been conducted in three eastern Washington ecoregions (Hessburg et al. 2007), and data are available to complete similar studies in the Blue Mountains and eastern Cascades of Oregon.

Similarly, syntheses of existing studies indicate relatively high variation in fire frequency within major plant series, especially at longer return intervals. For example, fire return intervals in the grand fir series range from 17 to 100 years (Agee 1994). Agee (1994, p. 17) stated that "the [warmer and drier] mixed-conifer forests of the Douglas-fir, white fir, and grand fir series show the most frequent fire activity of all eastside forests, although cooler, wetter sites of the grand fir series have longer fire return intervals. Frequent fires in drier plant associations of these series are likely due to higher productivity of fine dead fuels needed to carry another fire compared to the ponderosa pine series." Hessburg et al. (2007) corroborated this finding. In dry mixed conifers, they found that low- and mixed-severity fires dominated by surface fire effects affected most of the area of this type. In the MMC type, they found that mixed-severity fires were also dominant, but crown fire effects stemming from mixed-and high-severity fires were more obvious.

High variability in fire regimes as a function of environmental and vegetation context means that the correspondence of a MMC PVT with a particular narrow range of frequency and severity of fire and other disturbances should be viewed with caution. Variation in effects of fire suppression and logging on forest structure and composition is relatively large within the MMC type, especially as it is expressed in different geographic areas of the region. Thus, it is important to develop first-hand knowledge of the historical and contemporary disturbance regimes of local landscapes, and of their expression by PVT. Other factors to be considered when assessing the local disturbance regime include topography, soil physical and chemical characteristics, overall geomorphology, disturbance and management history, and climatic variability. Figures 14 and 15 illustrate the inherent heterogeneity that typically exists on these complex landscapes. Figure 14 shows the complex spatial configuration that results simply from basic topographic position at the southern end of the Wallowa-Whitman National Forest. Figure 15 illustrates the more complex patchwork of landscape units that result from a combination of topography, simplified categories of precipitation, and soils in the Wenaha River watershed on the Umatilla National Forest. This is a simple example of the concept of using a Land Type Association (LTA) that incorporates a number of landscape features into an ecological classification system based on the associations of biotic and environmental factors, which include climate, physiography, water, soils, hydrology, and potential natural communities.

General patterns of landforms, climate, and vegetation—
The ecological dynamics of the region are set within a geological and geomorphic template that controls climate, vegetation, and disturbance. Plate tectonics and volcanism, along with glacial and fluvial processes over geologic time, have produced distinct land surface forms, environments, and biotic assemblages across Oregon and Washington. These bio-geo-climatic contexts provide the basis to classify province-scale ecoregions (fig. 16) (Bailey 1995, Omernik 1987), and ecological subregions (Hessburg et al. 2000b) that reside within them. The location of the region on the windward Pacific Coast results in a predominantly marine-type

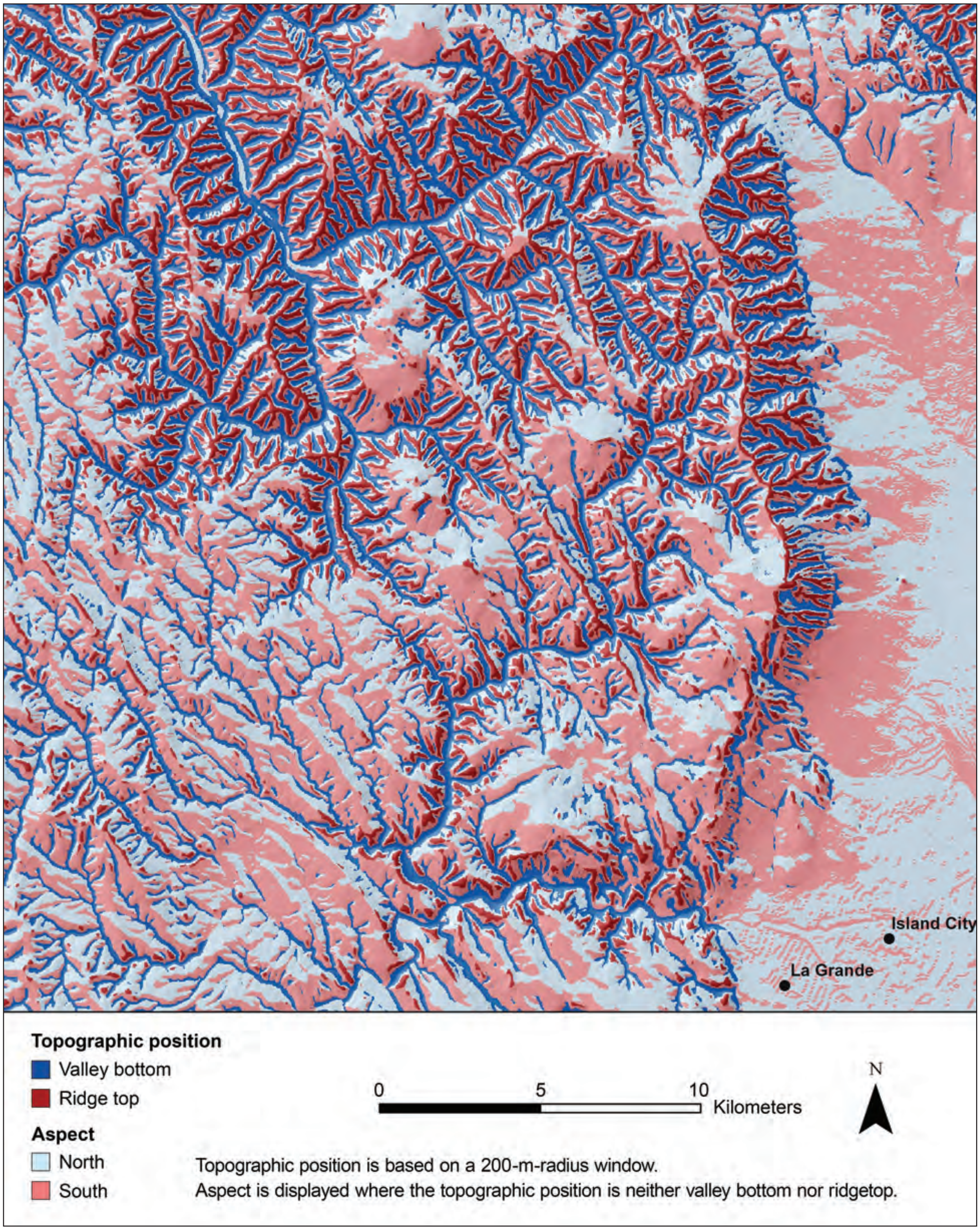


Figure 14—A map of the lower Grande Ronde Watershed on the Wallowa-Whitman National Forest in northeast Oregon, illustrating the spatial heterogeneity of the watershed simply as a function of basic topography.

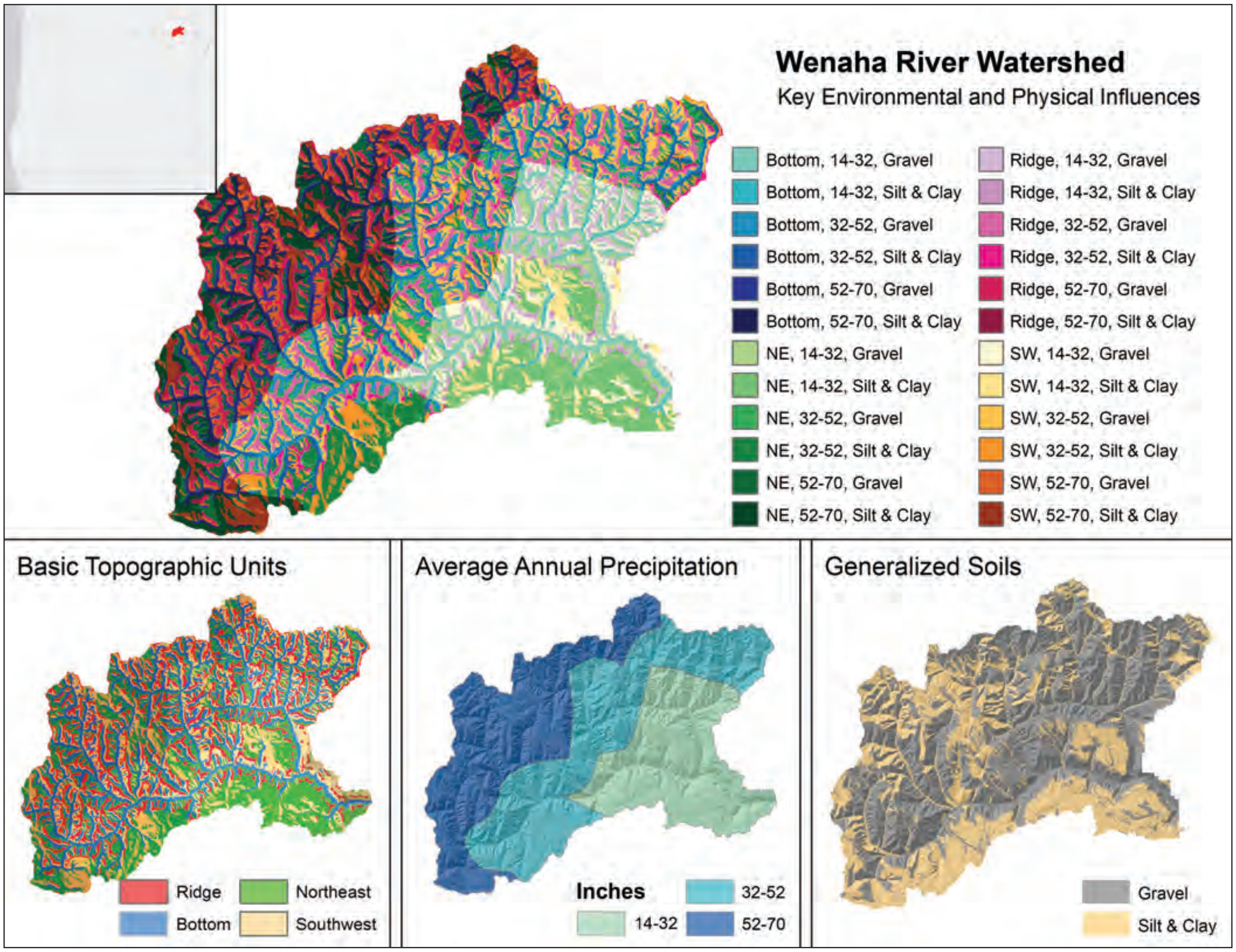


Figure 15—A map of the Wenaha River watershed on the Umatilla National Forest in northeast Oregon, illustrating the landscape complexity derived from an integration of simplified topography, soils, and climate spatial data.

climate west of the Cascade Mountains, while east of the Cascades, the climate possesses both continental and marine characteristics (Western Regional Climate Center, http://www.wrcc.dri.edu). In fall and winter, a low-pressure system in the North Pacific Ocean brings moist and mild (i.e., pacific) westerly airflow across the region, resulting in a wet season that begins in mid to late October, reaches a peak in winter (January to March), then gradually decreases later in spring. Blocking pressure in the North Pacific in summer brings a prevailing westerly and northwesterly flow of comparatively dry, cool, and stable air into the Pacific Northwest.

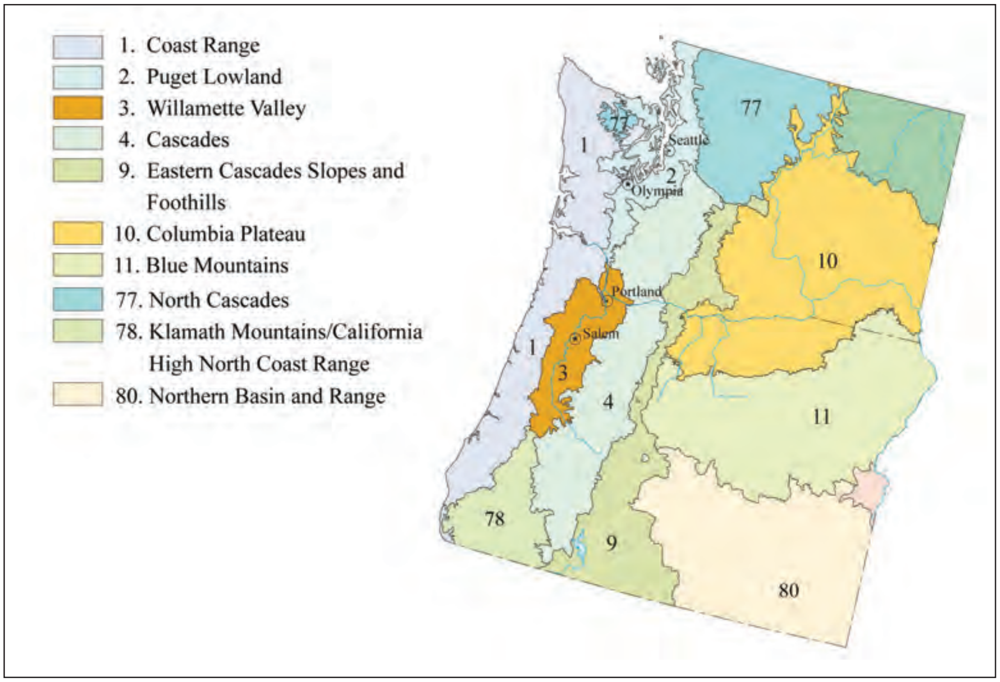


Figure 16—Ecological regions of Oregon and Washington based on the LEVEL III Ecoregions of the Continental United States, Environmental Protection Agency (revised in April 2013).

The orographic effects of the deeply dissected mountain ranges in the region result in occasionally heavy precipitation on the west slope of the Cascades and at higher elevations and relatively dry conditions on the east slope and at lower elevations (fig. 17). Temperatures vary from mild maritime conditions west of the Cascades crest to continental and temperate or Mediterranean conditions east of the Cascades.

Plant life forms and community types vary predictably across these landforms and climatic gradients. The Douglas-fir/western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) habitat type dominates west of the Cascade crest, except in the Willamette River, Umpqua River, and Rogue River valleys in the rain shadow of the Coast Mountains and portions of the Puget Trough (in the rain shadow of the Olympic Mountains), where oak savannas and dry coniferous forests occur (Franklin and Dyrness 1973). From the Cascades crest east to the west slope of the Rocky Mountains, there is a strong climate-induced stratification of vegetation types from valley bottoms to mountain tops. Sagebrush and grassland habitats dominate the Columbia Basin, the Great Basin, and the Snake River Plain. Ponderosa pine and juniper (*Juniperus* L.) habitats make up the lower forest and woodland ecotone. Douglas-fir occupies intermediate elevations, western larch and grand fir occupy still higher elevations, and Engelmann spruce and subalpine fir are found near upper tree line. Lodgepole pine inhabits subalpine environments and other low and mid-montane

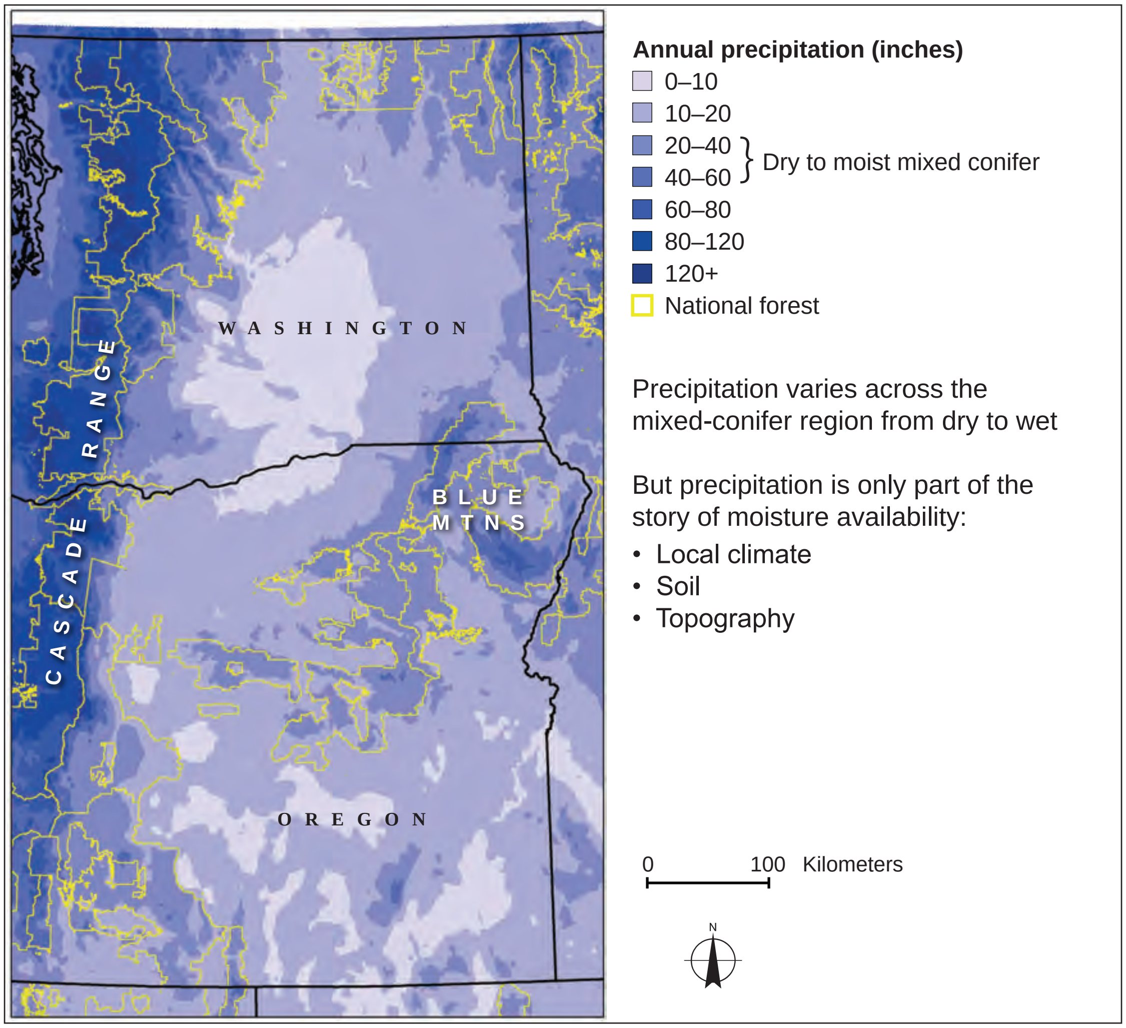


Figure 17—Generalized annual precipitation map of eastside Oregon and Washington. Note the significant rain shadow created by the Cascade Range and relatively high precipitation in some areas of the Blue Mountains.

environments that are prone to cold air pooling. Gradients extending from dry, low-elevation rangeland vegetation to moist conifer forests are especially sharp in the East Cascades, Okanogan Highlands, and Blue Mountains. In these locations, topographic controls (e.g., aspect, slope position) on vegetation composition, density, and productivity are also especially pronounced.

At the coarsest level, net primary productivity (NPP) is limited by temperature at the highest elevations, by moisture in the driest locations east of the Cascades crest, and by insolation (direct sunlight) west of the Cascades crest (Running et al. 2004). As a consequence, NPP is highest in the Coast and Cascade Ranges, intermediate in the East Cascades, Okanogan Highlands, and Blue Mountains, and lowest in the Columbia River basin and the Great Basin (Verschuyl et al. 2008).

Vegetation dynamics—

Vegetation dynamics result from the interplay between succession and disturbance processes. In the simplest case, vegetation development at a patch scale may be characterized by change in composition and structure following stand-replacement disturbance (e.g., from fire, insects, or disease). In such cases, succession can initially be relatively rapid (years to decades) and obvious, but successional change is occurring all the time, even centuries after major disturbances. The rate and nature of succession is advanced or retarded by a myriad of major and minor disturbances (O'Hara et al. 1996, Oliver and Larson 1996). In this sense, at each occurrence, disturbances may reset the vegetation condition of any patch on new trajectories that play out over space and time (Peterson 2002, Peterson et al. 1998). Patch boundaries are non-stationary as well. Patches are blended or bisected by disturbances of various kinds and combinations.

Large or severe disturbances may imprint the landscape for centuries before attenuating.

Large or severe disturbances may imprint the landscape for centuries before attenuating. The effects of small and low-intensity disturbances can be shorter in duration but still shape succession by altering site and microsite conditions that affect plant life histories by their influence on seed-fall, seed dispersal, post-dispersal seed losses, germination and recruitment, growth and mortality of juvenile and adult plants, reproduction, and a range of other factors.

Given the environmental complexity, diversity of species and processes, and variation in frequency and severity of disturbances that affect MMC forest, the number of successional pathways and development stages is quite large (fig. 18). Consequently, the structure and composition of the MMC forest across eastern Oregon and Washington was highly diverse under pre-Euro-American disturbance regimes. It was a mosaic driven by low- to mixed- and occasional high-severity fire, and the patchwork of size and age class structure and species composition differed considerably. With wildfire exclusion, domestic livestock grazing, and timber harvesting, the fire regime has shifted toward increasingly larger and more severe fires, which tend to simplify the landscape into fewer, larger, and less-diverse patches and ultimately more homogenous landscapes.

Ecological researchers are just beginning to understand ecoregional variation in the dynamics, structure, and composition of MMC forests. Given the generally sparse network of dendroecological and landscape reconstruction studies (especially in Oregon) our understanding of ecoregional and PVT-scale variation is provisional. In the eastern Cascades of Washington where high-severity fire was a component of the disturbance mix, the vegetation in MMC was a mosaic of small to very large patches comprised of early-, mid-, and late-successional forest conditions, and a broad variety of pure and mixed-cover types. In the eastern Cascades

Illustration by Tom Spies

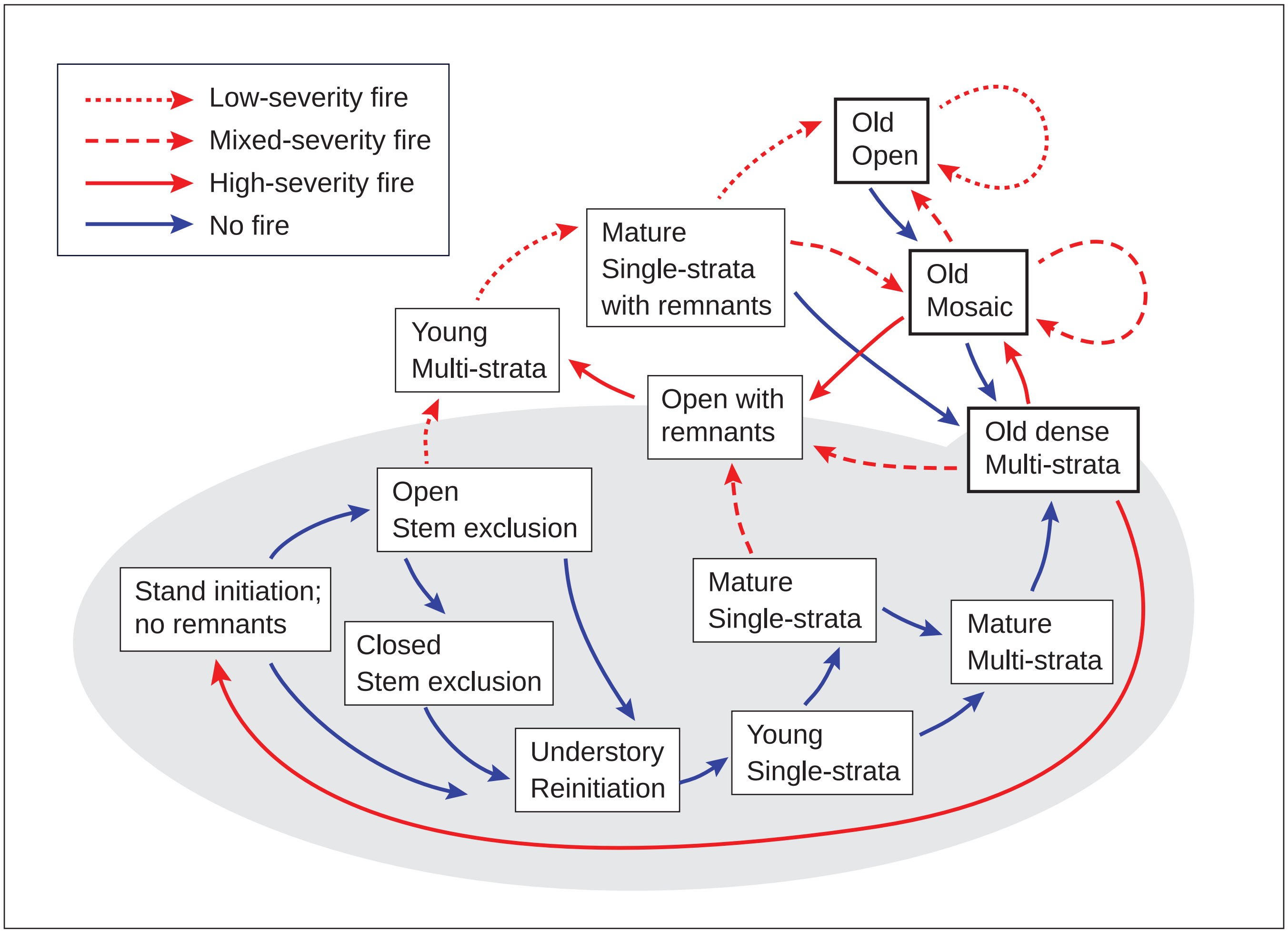


Figure 18—Conceptual model of vegetation states and transitions associated with fire severity. Under low- to mixed-severity fire regimes, moist mixed-conifer forest landscapes were dominated by patches characterized by old trees and relatively open understories or old trees in a mosaic of open and dense understory patches (bold boxes in upper right). Forest structure and composition shifted among various structurally diverse stages (boxes in upper non-gray area) depending on ecoregion, fine-scale environmental variation, and amount of high-severity fire. Under fire exclusion and even-age management, forest structure and disturbance regimes have shifted potential forest stages and pathways into the lower region of the diagram (gray area) where the amount of dense old forest, high-severity fire, and uniform young forest is much greater than it would have been under the pre-Euro-American fire regime. Some old, dense multi-strata forest would have existed under the pre-Euro-American period but it probably was a small portion of the moist mixed-conifer type across the region.

of Oregon, where the disturbance mix may have had less high-severity fire, MMC forests might have been a mosaic of large shade-intolerant forest species that can resist fire, and sparse or patchy understories of species that regenerated in areas that were disturbed by surface fires, insects, or disease, or in patches where disturbance had not occurred for several decades (Merschel 2012). On drier sites where fires were less severe and more common (<25 years), the environment may have favored growth and establishment of ponderosa pine or Douglas-fir, and understories would have been relatively open or patchy as surface fires from low- and mixed-severity fires periodically cleared them out. More work is underway and in time our understanding of this complex issue will be more complete.

Some sites within the mosaic of MMC forest would have had long intervals (>75 to 100+ years) between fires. It is on these sites that dense, multilayered forest conditions would have developed. In areas of high topographic complexity (e.g., eastern Washington Cascades and Blue Mountains) such sites are found on north-facing slopes at mid to upper elevations or at lower elevations near streams where moisture conditions would have made fuels too wet to burn except under infrequent hot dry weather conditions (Camp 1999, Camp et al. 1997). In areas of simpler topography defined by broad elevation gradients and isolated cinder cones (e.g., eastern Oregon Cascades), elevation and climate may explain the distribution of shade-tolerant species and fire regimes (Merschel 2012). Thus, many successional pathways occur in these landscapes but only a few of them would have reached old forest conditions where forests are relatively dense and dominated by shade-tolerant species, which are maladapted to fire-prone environments and droughty climate.

Despite the variation in disturbance regimes and environments, historical stand densities of large fire-tolerant trees in both dry and MMC were typically low and fell within a relatively narrow range (table 3). Of 15 estimates of historical small, medium, and large tree stand densities from different dry and MMC types and environments (based on seven total studies), 12 estimates fell within the range of 40 to 170 trees per hectare (16 to 70 trees per acre) and most of these estimates probably represent the minimum because most trees from cohorts that were regenerated in the 18^{th} and 19^{th} centuries were removed by disturbances. Regardless of their size, those that remain are simply the subset of survivors. Historical densities of large trees, most of them ponderosa pines and Douglas-firs, appear to fall within a relatively narrow range of low values from around 20 to 40 trees per hectare (8 to 16 trees per acre) (table 3). Figure 19 illustrates the strong contrast in historical versus current tree density in five locations in eastern Oregon and Washington.

Table 3 —Historical and current density (trees per hectare) of large diameter (typically >50 cm) trees in dry and moist mixed-conifer forest

Region	Forest description	Historical density	Current density	Source	Notes
		Mean ± standard deviation (range) and time period			
Eastern slope of the Cascade Range, southern Oregon	Mixed-dry	35 ± 15 (3–105) 1914–1922	21 ± 12 (0–45) 1997–2006	Hagmann et al. 2013	Historical density is reported for trees >53 cm diameter at breast height (dbh) based on timber inventory transects that covered 10 to 20 percent (6646 ha) of a 38 641-ha study area of ponderosa pine and dry and moist mixed-conifer forest. Current density is based on 24 current vegetation survey (CVS) plots within the study area, and density is reported for all trees >53 cm dbh. Inventory plots were assigned to habitat types (dry or moist) using mapped potential vegetation types (ILAP).
	Mixed-moist 1914–1922	35 ± 17 (3–147) 1997–2006	16 ± 15 (0–114)		
Warm Springs Indian Reservation, eastern slope of the Cascade Range, Oregon	Mixed-dry	24 ± 10 (1–62) 1922–1925	NA	Hagmann, unpublished data, 2013[a]	Historical density is reported for trees >53 cm dbh based on timber inventory transects. Plot size varied by species. Transects covered >20 percent (6952–11 327 ha) of a 32 293–ha study area of dry and MMC forest. Inventory plots were assigned to habitat types using mapped potential vegetation types (ILAP).
	Mixed-moist	26 ± 12 (0–102) 1922–1925	NA		
Eastern slope of the Cascade Range, Wenatchee National Forest, Washington	Mixed-dry/moist (estimates for several Douglas-fir potential vegetation types)	HDSG[b]—32 WDSH—2 WDTS—41 WMSH—27 1899	HDSG—77 WDSH—82 WDTS—99 WMSH—94 CDG—28 1999	Ohlson and Schellhaas 1999[c] CDG—71	For trees >40 cm (16 in) dbh. See table 5 for more information.
Blue Mountains, central Oregon	Mixed-conifer	27–32 1916–1932	NA	Forest Service inventories	Historical density reported for trees > 50 cm.
Eastern slope of the Cascade Range, central Oregon	Mixed-dry (Douglas-fir)	28 ± 19 (0–71) Early 20th century	17± 17 (0–50) 14 sample sites 2009–2010	Merschel et al. [N.d.]	Sample sites (~ 1.0 ha) were widely and randomly distributed in mature forest mapped in any of the mixed-conifer dry or mixed-conifer dry plant association groups. The number of sample sites used in each estimate is noted following the reported density. Density is reported only for shade-intolerant trees >50 cm. Historical density is based on diameter reconstruction of cut stumps, and current density is based on live trees only. Estimates refer to areas that were selectively logged, but 66 of 177 sites sampled had no evidence of logging. Period of historical density is estimated to be early to mid-20th century.
	Mixed-dry (grand fir)	30 ± 19 (17–85) Early 20th century	20 ± 15 (0–45) 16 sample sites 2009–2010		
	Mixed-moist	21 ± 17 (0–67.5) Early 20th century	12.9 ± 11 (0–45) 24 sample sites 2009–2010		
Ochoco Mountains, central Oregon	Mixed-dry (Douglas-fir)	32 ± 13.8 (5–64) Early 20th century	25.1 ± 15 (0-64) 26 sample sites 2009–2010		
	Mixed-dry (grand fir)	38 ± 19 (7.1–67) Early 20th century	19 ± 20 (0-65) 14 sample sites 2009–2010		
	Mixed-moist	18 ± 16 (0–48) Early 20th century	13 ± 13 (0-40) 14 sample sites 2009–2010		

Table 3 —Historical and current density (trees per hectare) of large diameter (typically >50 cm) trees in dry and moist mixed-conifer forest (continued)

Region	Forest description	Historical density	Current density	Source	Notes
		Mean ± standard deviation (range) and time period			
Eastern slope of the Blue Mountains, eastern Oregon	Mixed-conifer	NA	38 TPH 2012	Johnston 2013, unpublished data[d]	Density is reported for trees >50 cm in unlogged areas and provides an estimate of historical large tree density in logged areas in dry and MMC forests.
Eastern slope of the Cascade Range and foothills, Oregon and Washington	Mixed-dry	NA	21 ± 23 (n = 512) 1994–2004	Reilly 2012, unpublished data[e]	Density based on forest CVS plots distributed on a 2.5 by 2.5 km grid (5 km in wilderness areas) throughout national forests of Oregon and Washington. Sample sites include developmental stages ranging from early seral in recently disturbed areas (logging, wildfire, etc.) to mature old-growth forest. Density estimates are reported for trees >50 cm dbh. The number of CVS plots used to calculate density is reported in parentheses. CVS plots were assigned to PVTs based on ILAP.
	Mixed-moist	NA	27 ± 27 (n = 280) 1994–2004		
	Mixed-dry	NA	19 ± 16 (n = 471) 1994–2004		
Blue Mountains ecoregion, Oregon and Washington	Mixed-moist	NA	23 ± 21 (n = 1041) 1994–2004		

[a] On file with: Keala Hagmann, School of Environmental and Forest Sciences, University of Washington, Box 352100, Seattle, WA 98195.

[b] Plant association groups with respective plant associations:

HDSG (Hot/Dry/Shrub/Grass):
Pinus ponderosa/Agropyron spicatum
Pseudotsuga menziesii/Agropyron spicatum
Pinus ponderosa/Purshia tridentata/Agropyron spicatum
Pseudotsuga menziesii/Purshia tridentata/Agropyron spicatum
Pseudotsuga menziesii/Symphoricarpos albus/Agropyron spicatum

WDSH (Warm/Dry/Shrub/Herb):
Pseudotsuga menziesii/Arctostaphylos uva-ursi
Pseudotsuga menziesii/Arctostaphylos uva-ursi/Purshia tridentata
Pseudotsuga menziesii/Spirea betulifolia var. *lucida*

WDTS (Warm/Dry/Tall Shrub):
Pseudotsuga menziesii/Physocarpus malvaceus
Pseudotsuga menziesii/Physocarpus malvaceus/Linaea borealis var. *longiflora*

WMSH (Warm/Mesic/Shrub/Herb):
Pseudotsuga menziesii/Symphoricarpos albus
Pseudotsuga menziesii/Symphoricarpos albus/Calamagrostis rubescens

CDG (Cool/Dry/Grass):
Pseudotsuga menziesii/Arctostaphylos uva-ursi/Calamagrostis rubescens
Pseudotsuga menziesii/Purshia tridentata/Calamagrostis rubescens
Pseudotsuga menziesii/Spirea betulifolia var. lucida/Calamagrostis rubescens

[c] Unpublished study of stand development in mixed-conifer forest on the Malheur National Forest. On file with: James G. Johnston, Oregon State University, 321 Richardson Hall, Corvallis, OR 97331.

[d] Ohlson, P.; Schellhaas, R. 1999. Historical and current stand structure in Douglas-fir and ponderosa pine forests. Unpublished report. Portland, OR: U.S. Department of Agriculture, Forest Service, Okanogan and Wenatchee National Forests.

[e] On file with: Matthew Reilly, Department of Forest Ecosystems and Society, Oregon State University, 321 Richardson Hall, Corvallis, OR 97331.

Note: see table 5 for more information. Diameter thresholds, methods of estimation, and spatial coverage used to develop estimates differs among authors.

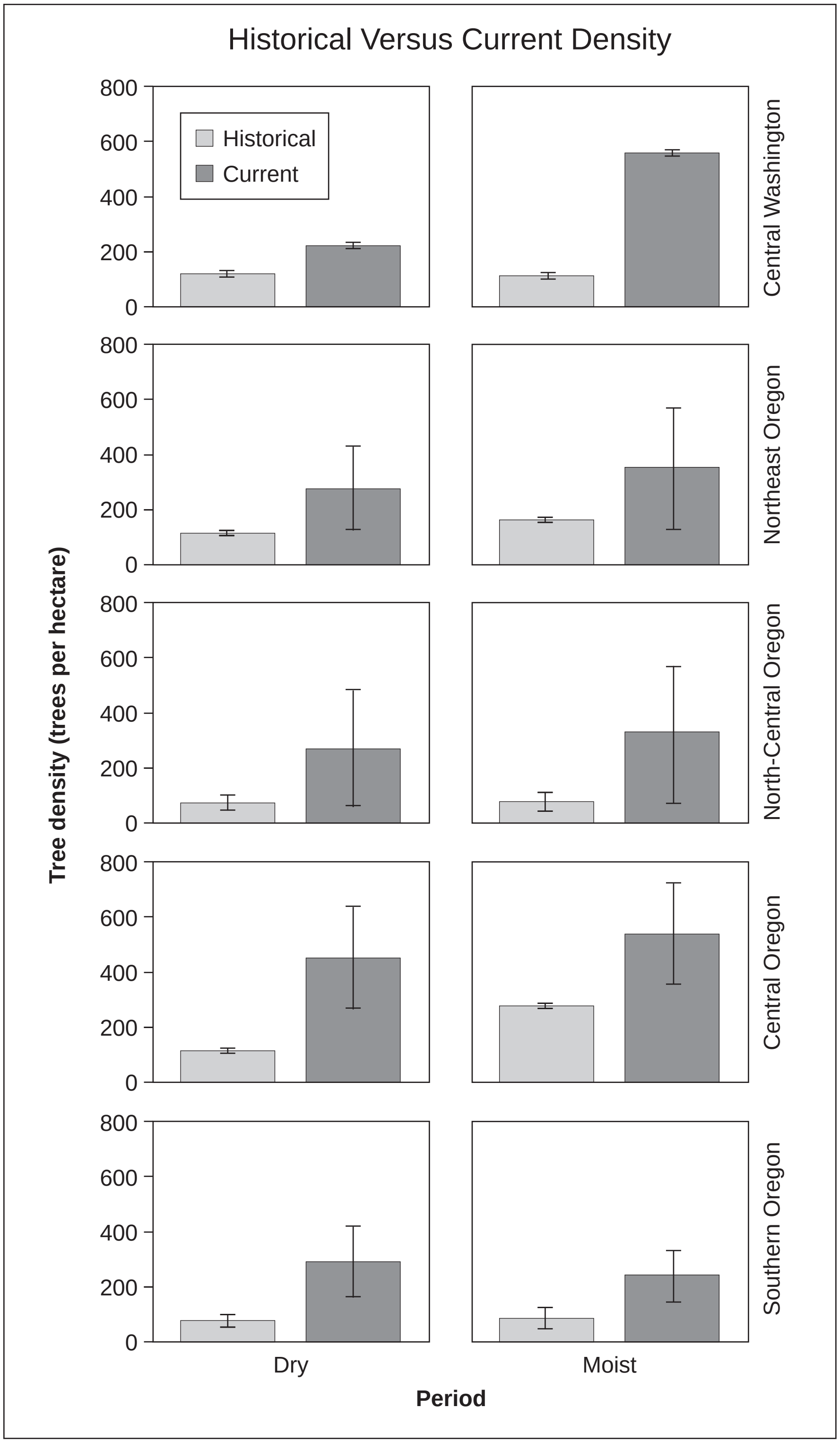


Figure 19—Historical versus current tree density in dry and moist mixed-conifer forest summarized from five study areas. Methods used to calculate density vary by reference period and study region, and diameter thresholds for tree density vary slightly among regions (10 to 15 cm). See the notes section in table 5 for citations and detailed information on how density was estimated in each region and time period.

Landscape Concepts

Provincial or regional landscapes consist of nested local landscapes, which are themselves comprised of yet smaller land units often referred to as stands or patches (Wu and David 2002, Wu and Loucks 1995). At each nested scale, species and ecosystems are controlled by spatial and temporal heterogeneity of patterns and processes. What occurs within a patch is affected by its surroundings and vice-versa. Figure 20 shows four east-west transects across montane regions of eastern Oregon and Washington and illustrates the continuous gradients of forest types that are found on these landscapes. MMC and other mixed-conifer types are thus part of a complex and intermingled landscape mosaic. The outcome of silviculture at the stand or patch level is thus inextricably linked to the interacting parts of the local and regional landscapes.

What occurs within a patch is affected by its surroundings and vice-versa.

Regional or broad-scale patterns of biota (e.g., life form zones and broad land cover types), geologic substrates (surface lithologies), geomorphic processes (land surface forms), and climatic patterns (e.g., spatial patterns of seasonal temperature, precipitation, solar radiation, and wind) constrain ecological patterns and processes occurring at a meso-scale. We call these constraints top-down spatial and temporal controls. For example, Bunnell (1995) found that the species composition of vertebrates in forest types is controlled by the disturbance regimes and mix of forest development stages found in biogeoclimatic zones in British Columbia, Canada. Changes to regional climate over relatively long time frames (100 to 1,000 years) affects relatively large spatial domains (1 million to 10 million ha). In other words, multi-century to millennial-scale changes in climate and geology can have a significant influence on species ranges. New plant and animal communities are potentially organized, new landform features emerge, and new patterns of environments and dominant disturbance regimes arise. A well-known example of landform effects in eastern Oregon is the eruption of Mount Mazama (current Crater Lake) about 7,700 years ago, which created a landscape of deep pumice deposits that control the composition and productivity of forest vegetation across central Oregon (Franklin and Dyrness 1988).

Likewise, fine-scale patterns of endemic disturbances (e.g., native insects and pathogens), topography, environments, vegetation, and other ecological processes provide critical context for patterns and processes over narrower scales or extent. These are termed bottom-up spatial and temporal controls. Bottom-up controls occur over relatively small spatial domains (1 to 100 ha), and drive processes that can vary temporally from hourly to annual time scales. For example, cool, moist, north-facing slopes can create fire "refugia" in which disturbance regimes and environment favor the development of forest containing old, fire-intolerant tree

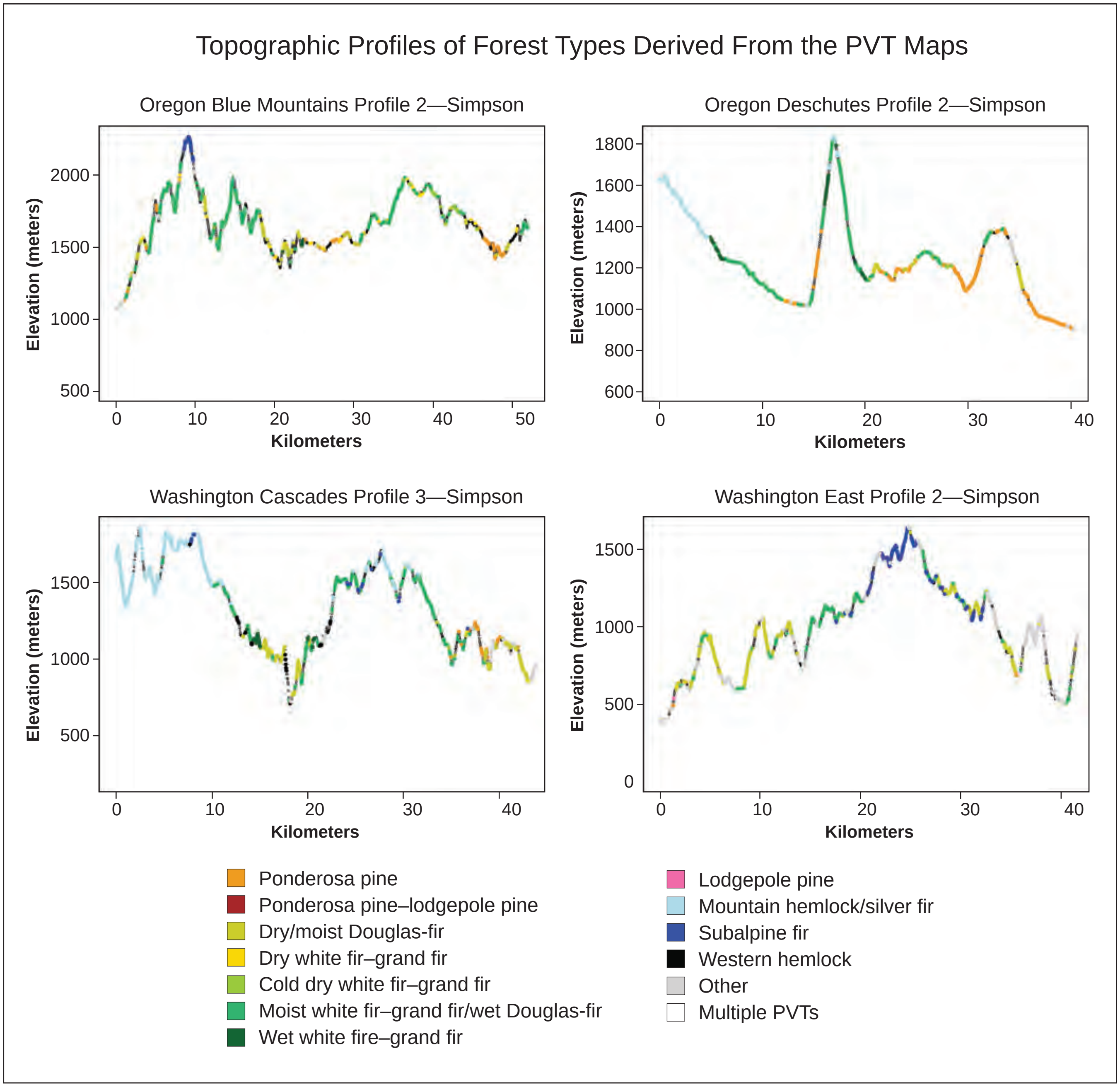


Figure 20—Profiles of predominant forest types, derived from potential vegetation types (PVTs) of Simpson (2007), across an east-west transect for four locations in eastern Oregon and Washington. Topography is exaggerated to show topographic effects.

species (Camp et al. 1997) (fig. 21). These small patches of old forest tree species can serve as seed sources that can affect rates and patterns of succession across the larger landscape (Wimberly and Spies 2001).

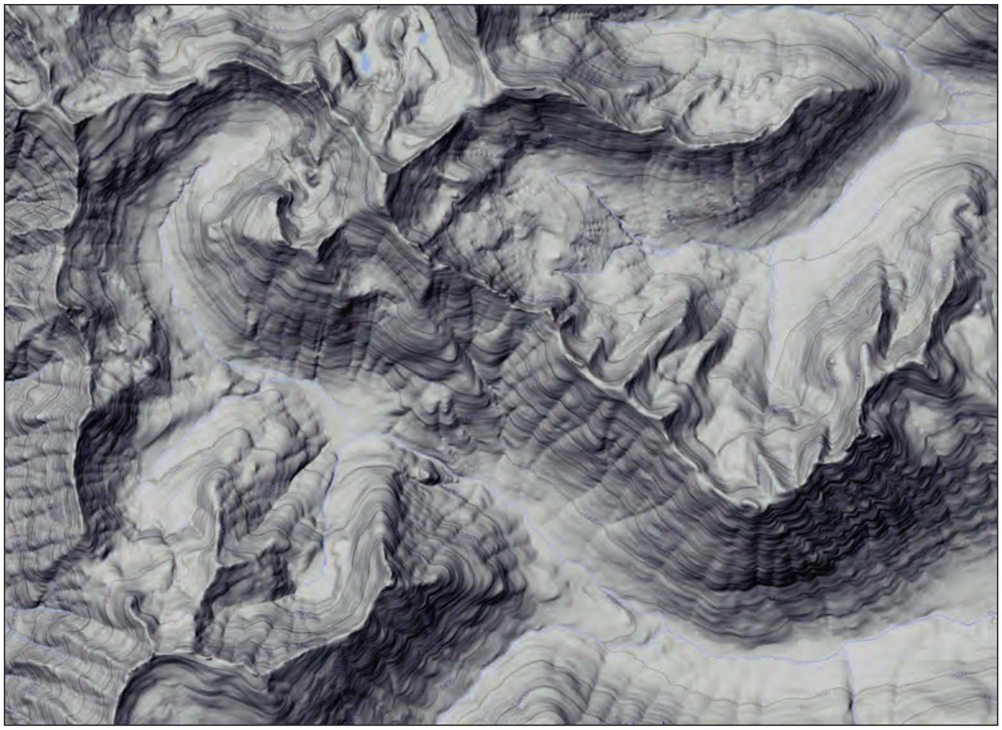


Figure 21—At the localized scale, mountainous terrain creates localized gradients in soil moisture, driven principally by differences in exposure to sunlight and wind (e.g., dry windy south-facing aspects versus cool moist north-facing hillslopes).

At all spatial and temporal scales of the hierarchy, landscapes exhibit transient patch dynamics and nonequilibrium behavior, resulting in ecosystem patterns and processes that may or may not change in linear or predictable ways. This is due to both random and deterministic properties of the supporting land and climate systems, and of ecosystem processes occurring at each level. Lower-level processes are incorporated into the next higher level of structures and processes, and this happens at all levels (figs. 11 and 21) (Wu and Loucks 1995).

Landscape patterns drive processes at local and regional landscape scales; at either scale, no two landscapes exhibit the same patterns across space or time. However, landscape patterns historically exhibited predictable spatial pattern characteristics. For example, a large sample of local landscapes (e.g., subwatersheds) comprised of dry and MMC forest in the lower and mid-montane settings, with subalpine forests in upper montane environments, reflected a predictable frequency-size distribution of cover type and structural class patch sizes (figs. 22 and 23).

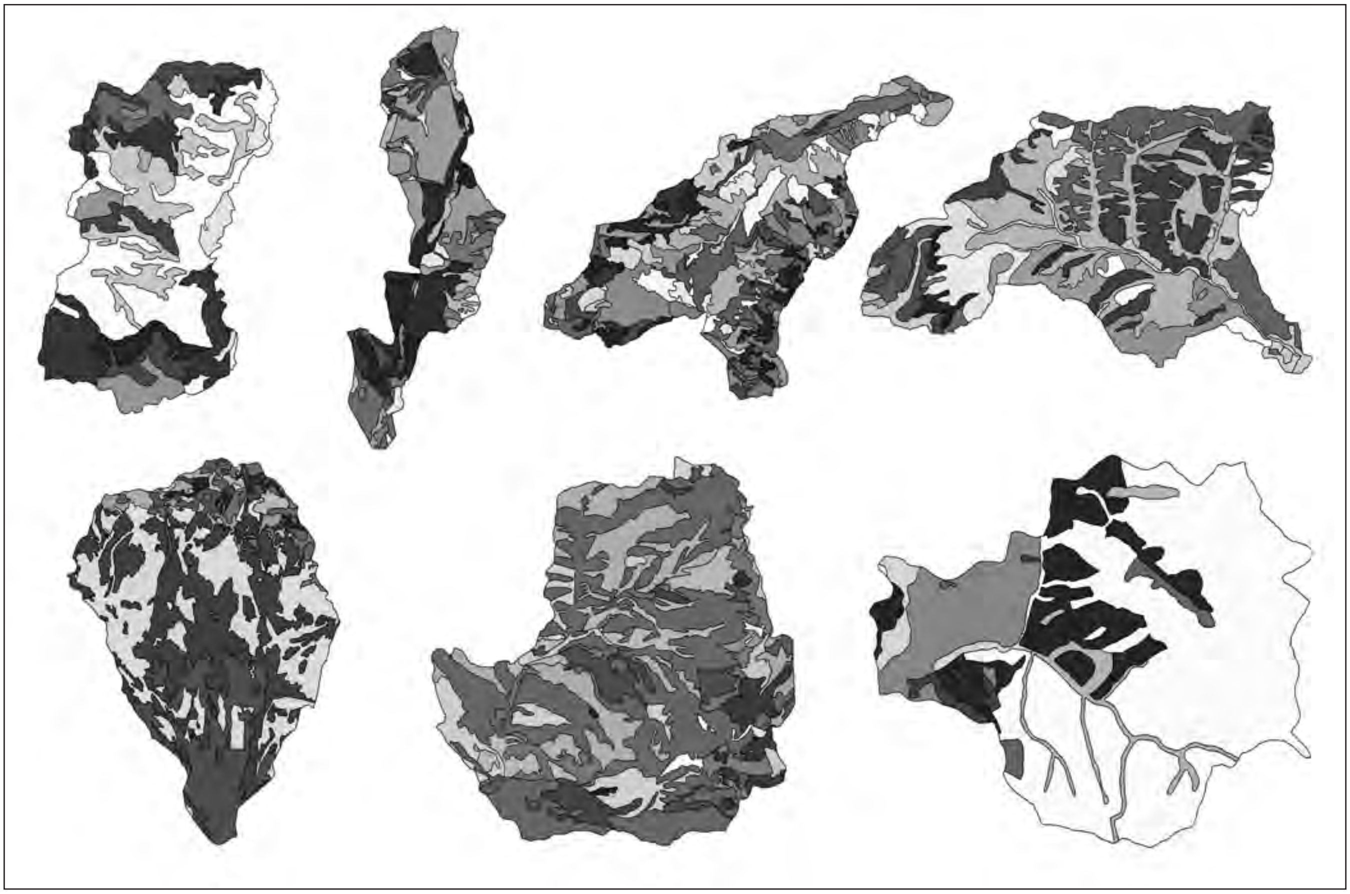


Figure 22—A sample of seven historical (ca. 1900) maps of combined cover type and structural class conditions from subwatersheds of the eastern Washington Cascades. Gray tones indicate unique cover type-structural class combinations. Note the highly variable patch sizes, with many small and fewer large patches. Despite their fewer numbers, the large patches actually make up the majority area of each subwatershed (adapted from Perry et al. 2011).

Here, think of histograms showing patch sizes in an inverse-J shaped distribution, with many small to medium-sized patches, and fewer large patches (fig. 23). Patterns and spatial arrangements of combined cover type and structural stage patch sizes (synonymous with O'Hara et al. [1996] structural classes) and their associated fuel beds provided constraint to the patch size and severity of disturbances. These patterns worked in concert with patterns of topography and weather influences to limit disturbance (Malamud et al. 1998, 2005; Moritz et al. 2010; Perry et al. 2011). This constraint probably appeared to be relatively stationary over multidecadal timeframes, but varied over multicentury periods.

As top-down controls such as regional climate or land surface forms significantly changed over large spaces and long time frames, these ranges were constantly being nudged and redefined. Moreover, because context and constraint

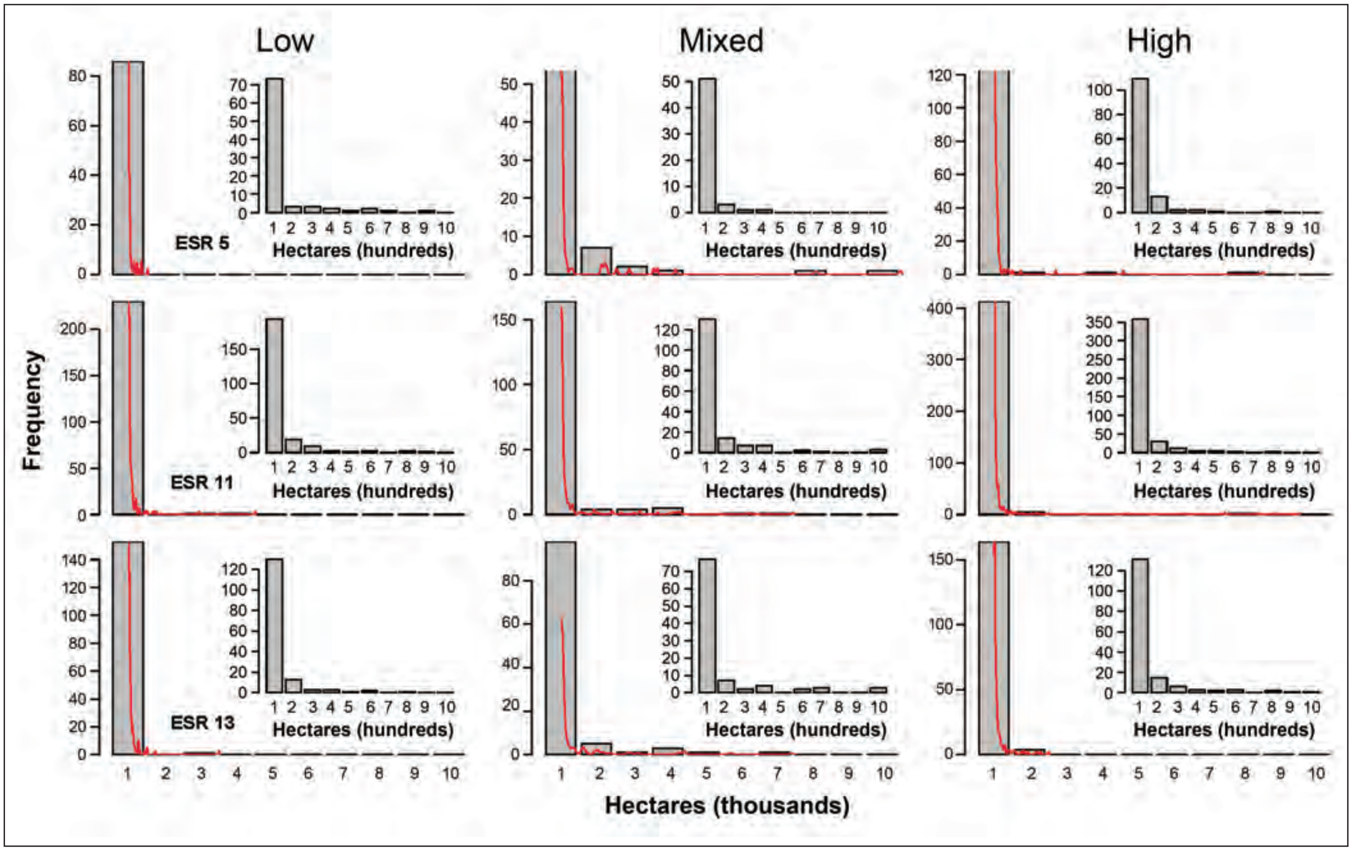


Figure 23—Frequency-size distributions of reconstructed historical (ca. 1900) fire severity patches in three ecoregions of the eastern Washington Cascades. Low, mixed, and high denote severity corresponding with <20 percent, 20 to 70 percent, and >70 percent of the overstory crown cover or basal area killed by fires, respectively. Data are from Hessburg et al. (2007). See Hessburg et al. 2000a for ecoregion details.

varied in the long term, the processes and patterns they reflected also varied over time. In a warming climate, e.g., the envelope of ecological patterns and processes at each level in the spatial hierarchy (literally the range of variation, RV) is reshaped by the strength and duration of warming (fig. 24). Reshaping of landscape patterns within a level can be figuratively represented as an envelope of conditions that drifts directionally in time (e.g., Nonaka and Spies 2005). Hessburg et al. (2000b) developed quantitative ecoregions for the interior Columbia River basin. They showed that there are strongly overlapping spatial patterns of regional climate, geology, and geomorphic processes (fig. 25). Later, they showed that these ecoregions explained some of the variance between various physiographic settings in the amount of low-, mixed-, and high-severity fires (Hessburg et al. 2004). This was first-order evidence of broad-scale spatial controls on meso-scale fire regimes.

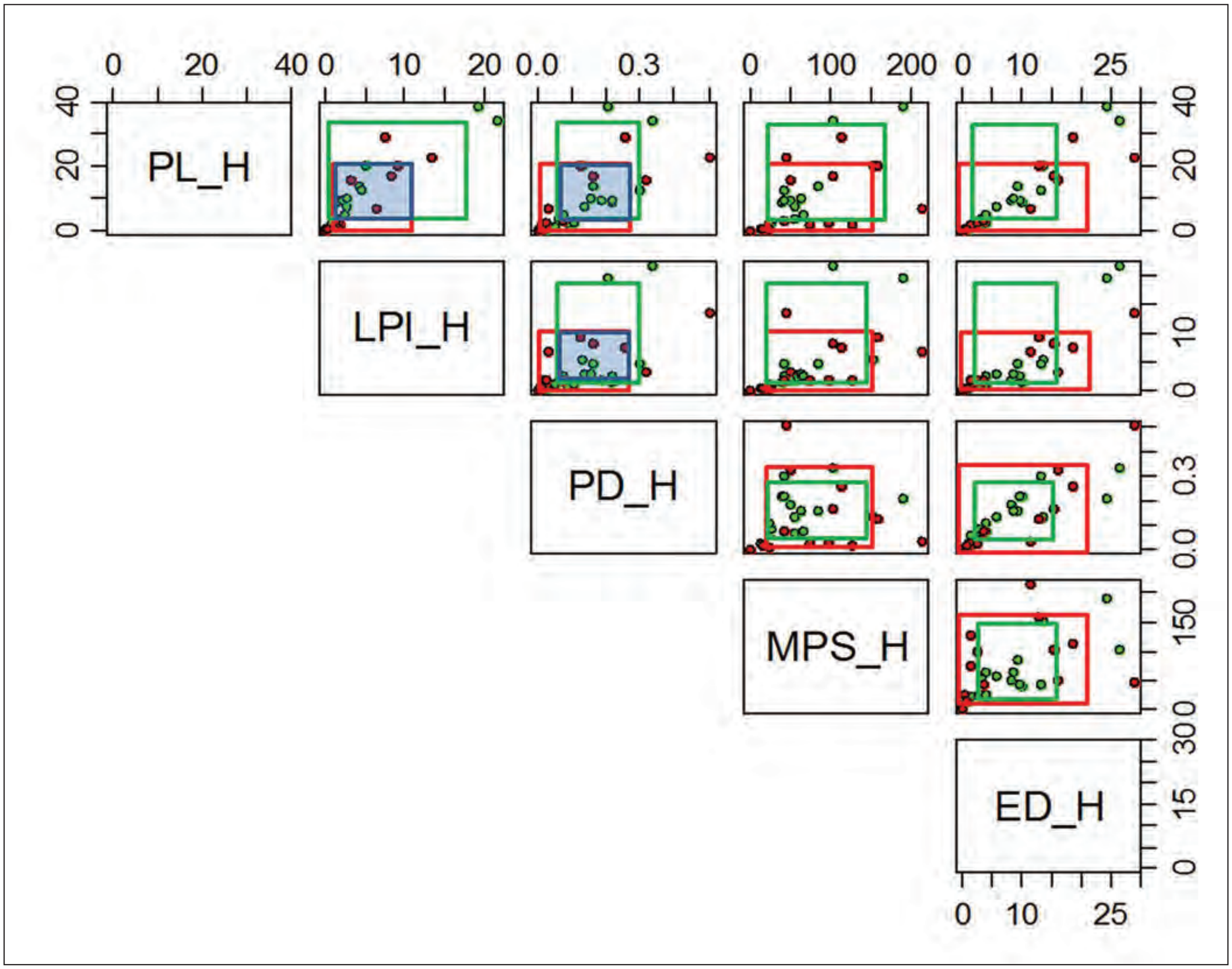


Figure 24—Comparing the historical range and variation (HRV, green dots and squares) and future range and variation (FRV, red dots and squares) with climate warming of five class metrics (paired in all possible combinations) of combined cover type–structural class patch types from a 10-percent random sample of subwatersheds of an eastern Washington Cascades ecoregion. The class metrics are the percentage of landscape area (PL_H, percent), the largest patch index (LPI_H, percent of the landscape), the patch density (PD_H, #/10,000 ha), the mean patch size (MPS_H, ha), and the edge density (ED_H, m/ha). The green square outlines the median 80 percent range of the subwatershed HRV sample. The red square outlines the median 80 percent range of the subwatershed FRV sample. Note the strong overlap between the HRV and FRV ranges in the pairwise combinations. In this example, we show paired combinations in a 2-space representation. Although impossible to graph here, one can imagine these metrics combined in a 5-space representation. In landscape analysis, any number of measures may be combined in an n-space representation to portray the HRV, the FRV, or both. In this way, departures or significant changes in spatial patterns of any contemporary landscape can be compared with the HRV and FRV to decipher the main characteristics of the changes.

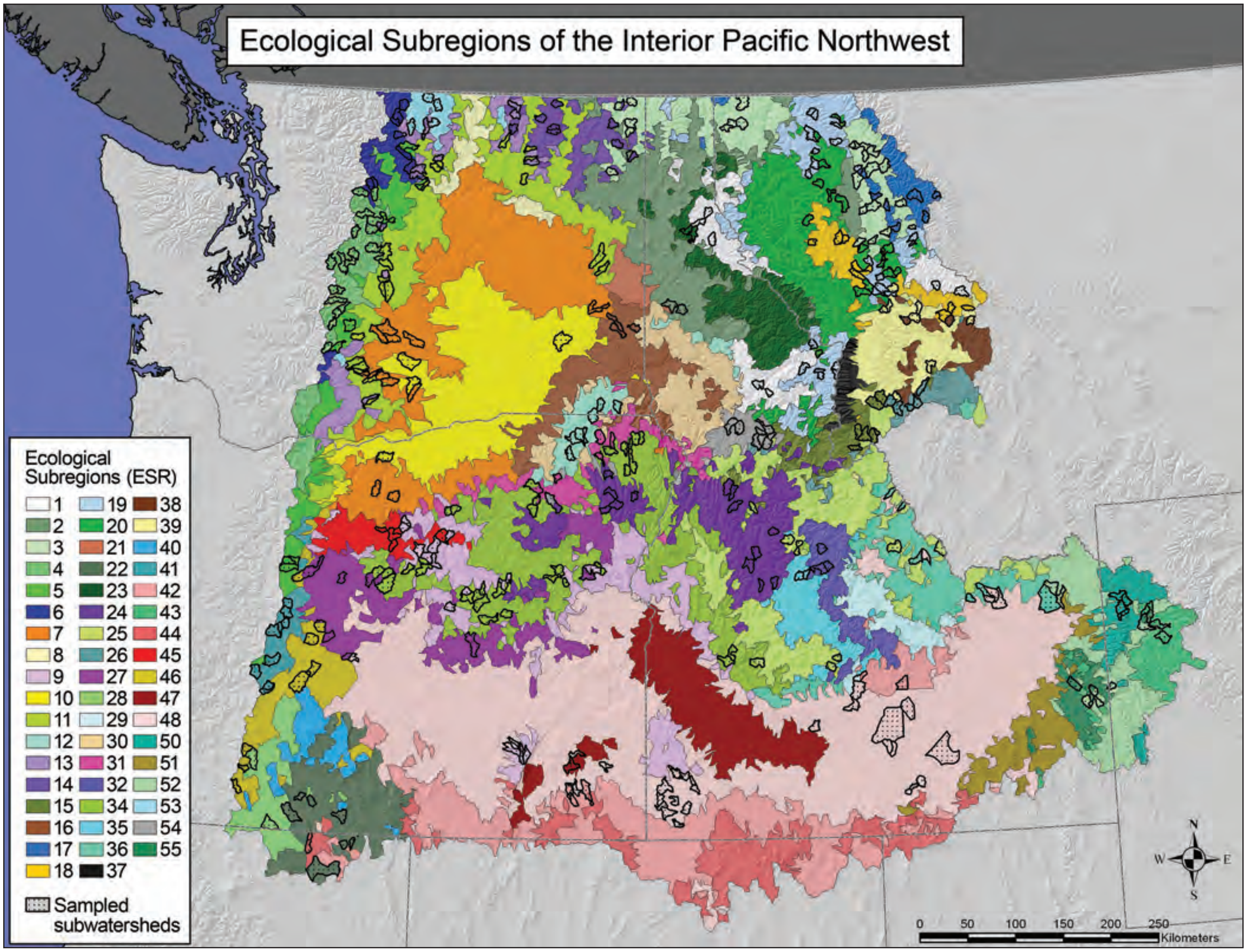


Figure 25—Ecological subregions of the interior Columbia River basin (Hessburg et al. 2000a). Names corresponding with ESR numbers are of the form: [ESR number = Bailey's (Bailey et al. 1994) ecoregion—Dominant Precipitation class(es)—Dominant Temperature class(es)—Dominant Potential Vegetation Group(s)—Dominant Solar Radiation class(es)]: 1 = M330–Wet–Warm–MF–Mod solar; 2 = M333–Moist–Warm–MF/DF–Low solar; 3 = M333–Moist–Cold–MF–Low solar; 4 = M242C–Wet–Warm–MFCF–Low solar; 5 = M242C–Moist–Warm–MF/CF–Mod solar; 6 = M242C–Wet–Cold–CF–Mod/Low solar; 7 = 342I–Dry–Warm–DS–Mod solar; 8 = M333A–Dry–Warm/Hot–DG–Mod solar; 9 = 300–Moist/Dry–Warm–DS/CS–High solar; 10 = 342I–Dry–Hot–DS/DG–Mod solar; 11 = 200/300–Dry/Moist–Warm–MF/DF–Mod solar; 12 = M330–Moist–Warm–DF–Mod solar; 13 = 200/300–Moist–Warm/Cold–MF–Mod solar; 14 = M333A–Moist–Warm/Cold–MF–Mod solar; 15 = M332–Moist/Wet–Cold–CF/DF–Mod/High solar; 16 = 330–Moist–Hot–DG–Mod solar; 17 = M333C–Wet–Cold–MF/CF–Low solar; 18 = M330–Moist–Warm–DF/DG–Mod solar; 19 = M330–Wet–Cold–MF/CF–Mod solar; 20 = M330–Moist–Cold/Warm – MF–Mod solar; 21 = 331A–Moist–Warm–DG–Low solar; 22 = 200–Moist/Dry–Warm–DF–Very high/High solar; 23 = M333D–Wet–Warm/Cold–MF–Low solar; 24 = M332–Moist/Wet–Cold–CF/DF/MF–High solar; 25 = M332–Moist–Cold–DF/CF–High solar; 26 = M332–Moist/Dry–Cold–CF–High/Mod solar; 27 = 300–Dry–Warm–CS/DF–High solar; 28 = 200 – Moist–Warm–DF/DG–High solar; 29 = M332–Dry–Cold–DS/CS–High solar; 30 = 300–Moist–Warm–DG/DF–Mod solar; 31 = 300–Moist–Warm–DF/DG–Mod solar; 32 = M332F–Moist–Cold/Frigid–CF–High/Very high solar; 34 = M332–Moist–Warm/Cold–DF/CS–High solar; 35 = M332–Moist–Cold–CF/DG–High/Very high solar; 36 = M332–Dry–Cold/Warm–DS/CS – High solar; 37 = M332B–Wet–Cold–RK/CF–Mod solar; 38 = M332–Moist–Cold–CF/DF/DG–Mod solar; 39 = M332B–Moist–Warm/Cold–DF/CF/DG–Mod solar; 40 = 200–Moist–Warm/Cold–DF–High/Very high solar; 41 = M242C–Moist–Warm–MF/DF/CF–High solar; 42 = 300–Dry–Warm–DS/CS–Very high solar; 43 = M331D–Moist–Cold–CF/DF/WD–Very high solar; 44 = 342–Dry/Moist–Warm–DS/CS–Very high solar; 45 = 342H–Dry–Warm/Hot–DS/CS–High solar; 46 = M242C – Moist–Warm–DF–High solar; 47 = 342C–Dry–Hot–DS/RS–High solar; 48 = 342–Dry–Warm–DS–High solar; 50 = M331–Moist/Wet–Cold/Frigid–CF–High solar; 51 = 300–Moist/Dry–Warm–DS/WD–High solar; 52 = M331–Moist–Cold–CF/WD – High solar; 53 = M242C–Moist–Cold–CF–Mod solar; 54 = M332–Moist–Warm–MF/DF–Mod solar; 55 = M331D–Moist – Cold–WD–High solar. Potential Vegetation Groups are: AL = Alpine; CF = Cold Forest; CS = Cool Shrub; DF = Dry Forest; DG = Dry Grass; DS = Dry Shrub; MF = Moist Forest; RS = Riparian Shrub; RW = Riparian Woodland; RK = Rock; WA = Water; WD = Woodland. Precipitation classes are: Very Dry = 0–150 mm/yr; Dry = 150–400 mm/yr; Moist = 400–1100 mm/yr; Wet = 1100–3000 mm/yr and Very Wet = 3000–8100 mm/yr. Temperature classes are: Frigid = –10––1 °C; Cold = 0–4 °C; Warm = 5–9 °C; Hot = 10–14 °C. Solar radiation classes are: very low = 150–200 W/m^2; low = 200–250 W/m^2; moderate = 250–300 W/m^2; high = 300–350 W/m^2; very high = 350–400 W/m^2. Mixed composition of attributes is indicated with a " / " and mixed attributes are listed in order of decreasing abundance. Sampled subwatersheds shown are those sampled by Hessburg et al. 2000a.

Large-amplitude and long-term changes relative to organism tolerance (often over several decades to centuries) reshape pattern envelopes by imprinting those landscapes with new disturbance and recovery patterns over large areas.

Changes that are small in amplitude and short term relative to the tolerances of dominant organisms (often multiannual to multidecadal) will do little to reshape the envelope because of the strength of system memory (inertia and elasticity; existing patterns of disturbance and recovery), expressed in successional stages and composition of vegetation. But large-amplitude and long-term changes relative to organism tolerance (often over several decades to centuries) reshape pattern envelopes by imprinting those landscapes with new disturbance and recovery patterns over large areas. For example, changes to the climate of the Southwestern United States desert biome since the last glacial maximum have been so substantial that plant and animal species ranges have fundamentally shifted and reorganized, along with their disturbance regimes (Betancourt et al. 1990).

It is informative for managers to understand the ranges of structural and compositional conditions that occur within landscapes of a region or subregion. Forest ecosystem response will be consistent with the current biophysical conditions and their disturbance regimes, including current climate forcing. For example, it is not clear how much the existing distribution of MMC forest will change with projected future climates and altered disturbance regimes. Fine-scale and gradual processes of mortality, dispersal, and range expansion are probably operating now to “adjust” patterns of species and ecosystems to the prevailing climate and disturbance trends of recent decades. Knowledge of factors controlling movement of species across landscapes provides insights to managers about how current landscape conditions will likely change, and thus how MMC forests may be able to persist under a changing climate. Multiscale research and monitoring are needed to elucidate these trends. Such information would enable a clearer vision to anticipate trends and develop tools for maintaining management options.

Natural Disturbance Factors

In this section, we highlight key disturbance factors in the MMC forest. Disturbance regimes vary across climate and environmental gradients in a relatively predictable fashion. These disturbances strongly influence vegetation composition, structure and habitat conditions, which themselves influence the variability of disturbances. These are brief synopses of complex topics, and we encourage the reader to delve deeper into the literature cited for more detail.

Wildfire—

Fire histories are not adequately documented in many moist forests in eastern Oregon and Washington. Our knowledge of past fire regimes and forest structure comes from proxy records, all of which are rich sources of data, but none of which is complete. Tree rings are one source of empirical data that can lead to multicentury reconstructions of fire and forest history, but only over small areas. Fire and forest vegetation histories reconstructed from aerial photos from the 1930s and 1940s are another source of empirical data, and although many such reconstructions predate most logging, most represent conditions that occurred 30 to 60 years after fires were already excluded. Thus our inferences are constrained by data limitations, especially when we need to apply them to particular potential vegetation types or ecoregions.

Historical fire regimes across the region—In this region, several spatially extensive studies have reconstructed multicentury histories of low-severity fire regimes from fire scars in dry and MMC forests and inferred their climate, topographic, and land-use drivers at landscape and regional scales (Everett et al. 2000, Hessl et al. 2004, Heyerdahl et al. 2001, Kellogg et al. 2008, Wright and Agee 2004). However, few studies have focused specifically on MMC forests as we define them (Hessburg et al. 1999a, 1999c, 1999d, 2007; Heyerdahl et al. 2001; Perry et al. 2011; Tiedemann and Woodard 2002; Woodard 1977; Wright and Agee 2004). These studies suggest that mixed-severity fire was likely a dominant influence with smaller amounts of low- and high-severity fire. Current forest structure and life history strategies of trees typically found in mesic forests support our assumptions that historical fires were less frequent than in dry ponderosa pine and mixed-conifer forests and were of mixed severity (Agee 1993).

Although we have little quantitative information on the size (but see Perry et al. 2011) and frequency of past fires in MMC forests, mean fire return intervals may have much less importance than the range of fire intervals in this forest type (Halofsky et al. 2011). From only a few reconstructed fires (e.g., eight fires in Heyerdahl et al. 2001), there was a broad range of return intervals (several decades to more than 100 years, often within a single stand), but information on historical fire size was unobtainable with these methods. No studies have been designed to identify fine-scale variation in fire severity (i.e., patch size) within individual fires and only one within PVTs (see Hessburg et al. 2007). Landscape-scale reconstructions of forest structure from early to mid-20th century aerial photographs across the region also indicate that fires were of mixed severity (Hessburg et al. 2007). Perry et al. (2011) provided fire severity patch size distributions relating to dry and MMC forests in eastern Washington using the Hessburg et al. (2007) data set.

Historical fire regimes in MMC forests of this region likely varied across a broad range of spatial and temporal scales in response to regional variation in climate and physical geology, and local variation in topography and environments (Gedalof et al. 2005, Heyerdahl et al. 2001, Littell et al. 2009). These factors also control vegetation distribution, but PVT as mentioned above is only one predictor of historical fire frequency or severity (Hessburg et al. 2007, Heyerdahl et al. 2001). Historical fire regimes have been reconstructed in MMC forests elsewhere in the Interior West, but presently we do not have enough information about spatial and temporal variation in the drivers of fire in these forests to know how well these reconstructions capture historical fire regimes throughout eastern Oregon and Washington. Low-severity fires were strongly synchronized in dry forests across the region by climate in the past (Heyerdahl et al. 2008) and the same may be true for moist forests, but the case has not been clearly made. Historical fire severity patterns have been reconstructed using dendroecological methods from about 10 percent of the total area of dry and MMC forests of the eastern Cascades in Washington. Extensive data sets are available from the Hessburg et al. (1999a, 1999b, 1999c, 1999d) Interior Columbia River Basin data archive to continue this work in the Blue Mountains and eastern Oregon Cascades.

Fire dynamics: resulting spatial patterns and tree mortality rates—Fire is a significant driver of mixed-conifer forest and aquatic ecosystems. The periodic influences of fire largely control the rates and patterns of succession and determine forest structure and composition, and aquatic and riparian condition. Despite recent contributions to the literature, there is still much scientific uncertainty and lack of consensus on the characteristics of fire regimes of mixed-conifer forests (amounts of low, mixed, and high severity), and how these varied by ecoregion. This lack of understanding (and investment in understanding) of the fire ecology of MMC forests limits the ability of managers to effectively identify appropriate steps toward ecosystem restoration. We do know that natural fire regimes in MMC forests varied somewhat predictably within areas that shared a similar climate and vegetation type, as well as topographic, edaphic (soil related), and environmental settings (Perry et al. 2011) (app. 3).

Low- to mixed-severity fires produced a highly variable mosaic of living and dead trees at multiple spatial scales, resulting in patchy regeneration in stands and landscapes, and perpetuating the low- and mixed-severity fire cycle. Low-severity fires (killing less than 25 percent of the overstory) typically occurred where fires were most frequent (e.g., every 10 to 30 years). MMC patches that experienced low-severity fire occurred in areas and slope positions that were adjacent to dry mixed-conifer and ponderosa pine woodlands. Low- and mixed-severity fire appear to be dominant in the eastern Cascades and parts of the Blue Mountains based on recent and ongoing work (Merschel 2012). Elevation and aspect influences on plant available soil moisture played important roles. Figure 26 illustrates a forest subject to a typical low-severity fire regime.

Figure 26—An example of a low-severity prescribed fire. These fires are characterized by minimal short-term ecosystem effects. The result of a low-severity fire is fuel reduction and topkill of understory vegetation.

Mixed-severity fires (killing 25 to 75 percent of the overstory) were common in the MMC forest because much of the area in these productive systems was visited by wildfires on a relatively infrequent basis (i.e., about every 25 to 75 years). This allowed for the development of a patchy vegetation mosaic, with patchy patterns of surface and ladder fuels, and both open and closed canopy conditions (app. 3). One may contrast fire severity in this way: low-, mixed-, and high-severity fires produced fine-, meso-, and broad-scale patchiness of tree structure and species composition within local and regional landscapes. Figure 27 illustrates a forest subject to a typical mixed-severity fire regime.

High-severity fires (resulting in over 75 percent overstory mortality) occurred occasionally (Agee 1990, 1993, 1994, 1998; Baker 2012; Hessburg, et al. 2007; Heyerdahl et al. 2001, 2008; Williams and Baker 2012), especially where fire return intervals exceeded 100 to 150 years. These conditions tended to occur in slope and aspect conditions adjacent to wet and cool to cold forests (e.g., western hemlock, mountain hemlock [*Tsuga mertensiana* (Bong.) Carrière], Shasta red fir, and Pacific silver fir [*Abies amabilis* (Douglas ex Loudon) Douglas ex Forbes)]) and did not appear to dominate the fire regime of MMC forests. Figure 28 illustrates a forest subject to a high-severity fire regime.

John Marshall

Figure 27—An example of the impacts of a mixed-severity fire, exhibiting a wide range of effects on the dominant vegetation. Some areas exhibit low fire severity, with little damage to overstory trees, while others exhibit moderate fire severity. Some areas exhibit high severity as indicated by complete mortality of the overstory.

John Marshall

Figure 28—An example of a high-severity fire, characterized by high levels of tree mortality, significant fuel consumption, and often extensive soil heating.

Landscape patterns: patch size and shape resulting from fire—There is still much to learn, but we know from fire history reconstructions using fire scars that small (1 to 50 ha [~2 to 123 ac]) to medium-sized (100 to 5,000 ha [~247 to 12,355 ac]) fires were most numerous, and pock-marked the landscape. They represented 85 to 95 percent of fire events, but only accounted for 5 to 15 percent of all of the landscape area burned by historical wildfires (Malamud et al. 1998, 2005) (fig. 29). The primary role of these smaller fires was to spatially isolate patterns of combustible surface and canopy fuels across the landscape (Moritz et al. 2010). In fact, the common small fires (that are currently suppressed) played a key role in shaping the larger landscape. Burned and newly recovering patches to some extent spatially interrupted the flow of larger fires, often limiting their spread. This was the primary mechanism maintaining the frequency-size distributions of wildfires. Large fires (>5000 ha [12,355 ac]) were much rarer, but they typically burned most of the landscape area that was subject to fire (75 to 95 percent).

We know from fire history reconstructions using fire scars that small to medium-sized fires were most numerous, and pock-marked the landscape.

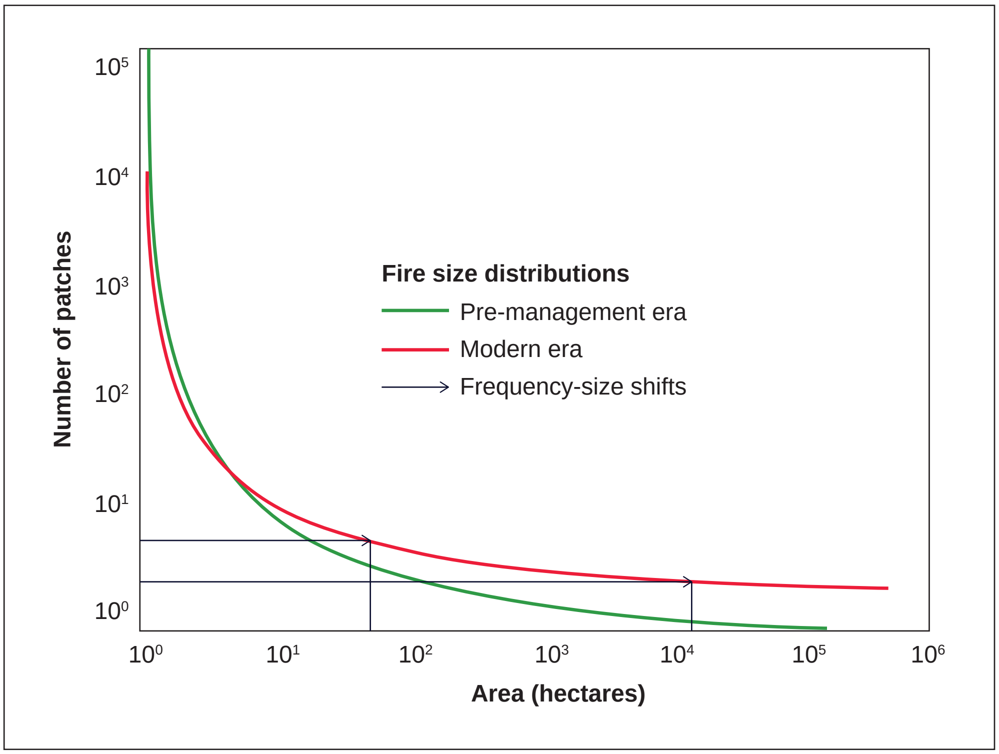


Figure 29—A comparison of fire size distributions in the pre-management and modern era. This illustrates the frequency-size shift toward larger burn patches in the modern era.

The distribution of fire event sizes approximated the negative exponential (an inverse-J distribution in the native form), and this appeared to be true regardless of fire severity (Perry et al. 2011). Overall fire frequency controlled the size of the largest fires. Numerous small- and medium-sized fires created patches of burned areas, in some stage of recovery, which reduced the likelihood of future fire spread. This mechanism likely provided a self-reinforcing type of resilience to the regional landscape. With relatively high fire frequency (as in surface fire dominated fire regimes), the largest fires were generally smaller than with relatively low fire frequency (characteristic of high-severity fire regimes).

Both landscapes and patches provided important feedbacks to wildfire frequency, severity, and size. At a landscape scale, low-, mixed-, and high-severity fires likely maintained patchworks of burned and recovering vegetation. Patches varied by size, age, density, layering, species composition, and surface fuelbed, and these landscape patterns spatially interrupted conditions that supported large disturbances, except under extreme conditions. Low- and mixed-severity fires also provided important patch level feedbacks, which encouraged subsequent low- or mixed-severity fires (Agee 2003, Agee and Skinner 2005, Hessburg and Agee 2003). Such fires frequently reduced surface fuels, which favored shorter flame lengths, and reduced fireline intensities (rate of heat release per unit length of fire front). They also increased the height to live crowns, which favored less torching, and reduced the likelihood of crown fire initiation. They decreased crown density, which reduced crown fire initiation and spread potential, favored young forest tree species, which increased tree survival during wildfires and droughts, and favored medium- and large-sized trees, which increased tree survival during wildfires. Finally, low- and mixed-severity fires produced patchy tree and surface fuel cover, which favored fire-tolerant species, and repeated low- and mixed-severity fires.

During periods of extreme weather or during rare climate events, wildfires could become very large, and fire behavior was often unrelated to the initial forest conditions, but was instead correlated with the conditioning climate or weather influences. Fire size distributions and the vegetation patchwork that supported them have been altered by a host of management and settlement influences, which we have summarized.

Insects—

In eastern Oregon and Washington, several insect species are significant disturbance factors in MMC forests, including four native bark beetle species and two native defoliating species (Hayes and Daterman 2001, Hayes and Ragenovich 2001, Torgersen 2001). In addition to these, there is one nonnative defoliating species and one nonnative sap-sucking species of importance (table 4). The impact of these insects is influenced by other disturbances (e.g., root disease, extreme weather events, wind storms, or wildfire) while outbreaks influence the frequency, severity and extent of other disturbances.

Unlike defoliating insects, bark beetles feed on the phloem tissue beneath the bark, often directly killing the host quickly via girdling. Beetles that initiate host selection are often killed by drowning or immobilization in resin, so successful colonization requires a minimum number of beetles to "mass attack" the tree and overcome its defenses (Franceschi et al. 2005). This number varies, as more vigorous hosts require higher densities of beetles (Fettig et al. 2007). Therefore, the legacy of past disturbances influences the susceptibility of forests to future bark beetle outbreaks by affecting their structure and composition. Many bark beetle species exhibit a preference for larger diameter trees growing in high-density stands with a high percentage of host type (Fettig et al. 2007). Larger diameter trees generally provide for a higher reproductive potential and probability of survival (e.g., Graf et al. 2012, Reid and Purcell 2011) because of the greater quantity of food (phloem) available on which larvae feed. The effects of density are mediated through factors that affect host finding and colonization success, primarily microclimate, tree spacing, and changes in host vigor as mediated through the influence of growing space. Over the last decade, the majority of insect-caused tree mortality detected during aerial surveys in the region has been attributed to outbreaks of the mountain pine beetle (*Dendroctonus ponderosae*). The impact of this insect seems to be increasing in eastern Oregon and Washington, likely owing to changes in tree density and the abundance of preferred hosts (table 4), and is expected to intensify in the future (see "Climate Change" on page 102 and "Changes in Disturbance Factors: Insects and Disease" on page 115).

Defoliators consume, mine, or skeletonize the foliage of trees and may cause tree mortality depending on the species and host, and the timing, frequency and severity of feeding. Without question, the two most important defoliators in MMC forests of eastern Oregon and Washington are the Douglas-fir tussock moth (*Orgyia pseudotsugata*) and western spruce budworm (*Choristoneura occidentalis*) (table 4, figs. 30 and 31), which are notable for their infrequent, but occasional large-scale outbreaks (Torgersen 2001). Natural enemy populations (parasites, predators, and parasitoids) have a strong regulatory effect on their populations resulting in long time lags between outbreaks. In particular, western spruce budworm populations are well-coordinated with climate (Kemp et al. 1985, Volney and Fleming 2007). From the 1950s to 1980s, large-scale efforts were implemented to control Douglas-fir tussock moth and western spruce budworm outbreaks using aerially applied

Table 4—Insects considered major disturbances in moist mixed-conifer forests of eastern Oregon and Washington

Common name	Scientific name	Primary host(s) in region	Scale of outbreaks	Impacts
Bark beetles:				
Douglas-fir beetle	*Dendroctonus pseudotsugae*	Douglas-fir	Local to landscape	Mortality of individual trees or small groups of trees is typically associated with defoliation, drought, windthrow, wildfire, or root disease. Once established, outbreaks may cause large amounts of tree mortality over extensive areas.
Fir engraver	*Scolytus ventralis*	Grand fir, white fir	Local to landscape	Mortality of individual trees or small groups of trees is typically associated with drought and defoliation. Once established, outbreaks may cause large amounts of tree mortality.
Mountain pine beetle	*Dendroctonus ponderosae*	Lodgepole pine, ponderosa pine, western white pine	Local to regional	Episodic outbreaks are a common occurrence, but the magnitude and extent of recent outbreaks have exceeded the range of historic variability in much of the Western United States.
Western balsam bark beetle	*Dryocoetes confusus*	Grand fir, subalpine fir	Local to landscape	Mortality of individual trees or small groups of trees is common and associated with windthrow.
Defoliators and other:				
Balsam woolly adelgid	*Adelges piceae*	Grand fir, silver fir, subalpine fir	Local	Accidentally introduced into the United States from Europe and first detected in the Pacific Northwest in 1929. Adelgids are sap-sucking insects. Feeding causes abnormal growth responses in affected trees termed “rotholtz.” Severe infestations cause growth loss, top-kill, and tree mortality.
Douglas-fir tussock moth	*Orgyia pseudotsugata*	Douglas-fir, grand fir, white fir, subalpine fir	Local to regional	Larval feeding causes growth loss, top-kill, and tree mortality. Outbreaks are often short-lived.
Larch casebearer	*Coleophora laricella*	Western larch	Local	Accidentally introduced into the United States from Europe and first detected in the Pacific Northwest in the 1950s. Larvae mine needles. Severe defoliation occurs, but larch (a deciduous conifer) withstands repeated defoliation. Growth loss is common, but tree mortality is rare.
Western spruce budworm	*Choristoneura occidentalis*	Douglas-fir, grand fir, white fir, subalpine fir, western larch	Local to regional	Early instar larvae mine needles and buds. Late instar larval feeding causes growth loss, top-kill, and tree mortality. Feeding on staminate flowers and conelets affects regeneration.

William M. Ciesla, Forest Health Management International, Bugwood.org

Figure 30—Douglas-fir tussock moth larvae.

Lia Spiegel

Figure 31—Spruce budworm larvae.

Although insect infestations affect timber and fiber production, and indirectly affect a range of ecosystem goods and services, numerous organisms depend on insect-related disturbances for their existence.

chemical and biological (bacteria- and virus-based) insecticides, but were thought to have little effect on population dynamics or levels of tree mortality (Torgersen 2011, Torgersen et al. 2005). A successful example of classical biological control has significantly limited the impact of larch casebearer (*Coleophora laricella*) in the region (Ryan 1997).

Although insect infestations affect timber and fiber production, and indirectly affect a range of ecosystem goods and services, numerous organisms depend on insect-related disturbances for their existence. Trees weakened or killed by insects and other disturbances (e.g., pathogens) can result in the transience of old forest structural conditions, and also create structure and food sources that have significant value to wildlife communities. The section "Animals of the Moist Mixed-Conifer Ecosystem" on page 75 provides some detail on the role of snags and downed logs to wildlife. Furthermore, mortality of individual or small groups of overstory trees has a significant influence on the fine-scale spatial heterogeneity of mixed-conifer forests (Fettig 2012). For example, some insects, specifically bark beetles, inflict density-dependent tree mortality (Fettig et al. 2007), and consequently maintain a mix of tree species, ages, sizes and spatial heterogeneity that influence other disturbances (e.g., wildfire).

There is some evidence that large-scale tree mortality events associated with insect outbreaks may increase fire risk and severity in affected forests. In recent years, these relationships have been studied most extensively in lodgepole pine forests affected by mountain pine beetle (fig. 32). Although there is some disagreement over predicted changes in fire behavior during and after mountain pine beetle outbreaks (reviewed most recently by Jenkins et al. 2014), most studies predict increases in surface fire rates of spread and fireline intensities resulting from increases in fine fuel loadings and reductions in sheltering (e.g., Hicke et al. 2012, Klutsch et al. 2011, Page and Jenkins 2007). During outbreaks, changes in foliar chemistry and moisture content may also increase probabilities of torching and crowning, and the likelihood of spotting (Jolly et al. 2012, Page et al. 2012). Similarly, feeding by defoliators reduces crown cover, increasing the amount of light reaching the forest floor and influencing understory and mid-story vegetative dynamics. For example, western spruce budworm outbreaks have been shown to increase surface fuel loads, yet changes in fire behavior were not significant (Hummel and Agee 2003).

The current structure and composition of dry and MMC forests is thought to be more susceptible to large-scale defoliation by Douglas-fir tussock moth and western spruce budworm in areas where fire suppression and selective harvesting of ponderosa pine and larch have favored Douglas-fir, grand fir, and white fir.

Figure 32—Mortality of lodgepole pine attributed to mountain pine beetle (*Dendroctonus ponderosae*) in northern Utah. In recent years, the effects of mountain pine beetle on fuels and fire behavior have been studied most extensively in the lodgepole pine forests of the Rocky Mountains. This species, however, now represents a significant threat to the mixed-conifer forests of eastern Oregon and Washington.

Defoliation, largely the result of infestation of these two insects, often predisposes trees to subsequent mass attack by other insects, specifically bark beetles such as the Douglas-fir beetle (*Dendroctonus pseudotsugata*).

Diseases—

In eastern Oregon and Washington, several notable forest pathogens produce significant disturbance in MMC forests, including five host-specialized dwarf mistletoes and four native root pathogens. Disturbance comes in the form of tree mortality and reduced tree growth, both of which are influential to forest succession and stand dynamics processes. Numerous stem decay organisms are also common in large and older trees of most species. These are excluded for brevity here, but we urge interested readers to delve into this literature. For a concise survey of the most common diseases and insects affecting inland Pacific Northwest forests, we refer the reader to Goheen and Willhite (2006), and recommended references therein.

Four tree-killing root diseases naturally occur in MMC forests: laminated root rot, caused by *Phellinus weirii*; Armillaria root disease, caused by *Armillaria ostoyae*; and both the P- and S-type annosum root diseases (formerly *Heterobasidion annosum*, now *H. irregular* and *H. occidentale*, respectively) (Filip 1990; Filip and Goheen 1982, 1984; Goheen and Filip 1980; Hadfield et al. 1986). Root diseases were common (<5 to 10 percent of patches affected) but not dominant in most presettlement-era MMC forests, where they provided structural diversity within patches and enhanced heterogeneity in size of openings, amount and shape of edge, and size of patches. They were likely most visible in areas with relatively infrequent fire, and less visible in areas influenced by frequent low- and mixed-severity fires (e.g., south-facing slopes and ridges), and where the dominant tree cover was ponderosa pine or western larch.

Laminated root rot infects and kills susceptible Douglas-fir, grand fir, and white fir that grow in patches missed by fire. Transmission of the fungus that causes this disease occurs via mycelial growth, when roots of susceptible host trees come in contact with those of infected trees. Because the root systems of host trees are often well rotted after they are infected, these trees usually fall over in a jackstraw arrangement (Hadfield et al. 1986). Owing to the dominance of historical wildfires and the relative rarity of root rot centers (in comparison with current conditions), historical root disease centers likely provided small to large gaps that contained root disease resistant hardwood shrubs and trees and other resistant conifer species, which enhanced plant species richness, and provided mast and a variety of habitats suitable for small mammals (Maser et al. 1979; Thomas et al. 1979a, 1979b).

Armillaria root disease also infects and kills susceptible grand and white fir, and occasionally Douglas-fir that grow in patches missed by fire. Transmission of the fungus that causes this disease occurs when roots of susceptible host trees come in contact with those of infected trees. Inter-tree transmission is facilitated by fungal mycelia and by specialized root-like fungal structures called rhizomorphs. Armillaria root disease ecology at the end of the 19th century was probably very similar to that of laminated root rot. Armillaria root disease probably played a role in forest succession and stand dynamics of many refugia dominated by grand or white fir. Refugia were found in shaded draws, on cool north slopes, in riparian areas and stream confluence zones (Camp et al. 1997), and adjacent to rock outcroppings and talus slopes, where fires burned with difficulty. This pathogen also overwhelmed low-vigor, mature, weakened, and injured trees, and those stressed by drought or lightning strike, scorched by fire, or attacked by other root pathogens (Filip and Goheen 1982, Goheen and Filip 1980). Thus, it is fairly common to find more than a single root pathogen colonizing trees.

P- and S-type annosum root disease centers were relatively uncommon in presettlement-era forests. These diseases require freshly cut stumps or wounds for windborne spores to infect and initiate new root-disease centers. Before tree harvesting, annosum root disease existed as a butt rot of trees with root collar and stem wounds. In central, southern, and northeastern Oregon, stands that have had multiple entries to harvest trees have been shown to have the highest annosum root disease and associated bark beetle-caused tree mortality (Filip et al. 1992; Schmitt et al. 1984, 1991). Most surprising in the MMC forest is the rate of increase in S-type annosum root disease (*H. occidentale*) in grand and white fir. These forests contain large increases in S-type annosum because stumps were infected by spores when stands were logged (Filip et al. 1992, Hadfield et al. 1986, Otrosina and Cobb 1989). Because S-type isolates are primarily pathogenic on true firs and spruces, the roles these stumps will play in the future incidence of disease is uncertain. Infection centers will continue to expand until fire or silvicultural activities create conditions for the reintroduction of young forest species.

Pine stump infection by P-type annosum (*H. irregular*) is often high in Douglas-fir and grand fir forests, but mortality in ponderosa pine is uncommon. With prolonged warming, however, P-type annosum may become more serious on what are now mesic white and grand fir sites.

In the current condition, all major tree-killing root diseases except P-type annosum are widespread, following landscape colonization by grand fir, white fir, and Douglas-fir (Hessburg et al. 1994). Collectively, the effects of root diseases on tree growth and mortality, and their contributions to flammable fuels are ecologically significant. At a watershed or subwatershed scale, as much as 7 to 10 percent of the area can be influenced at any one time by active infection centers.

Dwarf mistletoes have occurred for many millennia in Douglas-fir, ponderosa pine, western larch, lodgepole pine, and true firs but none was particularly threatening to the long survival of its host species (Alexander and Hawksworth 1975, Parmeter 1978, Tinnin 1981). Douglas-fir dwarf mistletoe (*Arceuthobium douglasii*) and western larch dwarf mistletoe (*A. laricis*) were probably common in mid-seral and old forests before the 20th century. Areas infested with these mistletoes tended to be the more mesic plant associations, in which fires appeared with moderate frequency. Mistletoes were probably most common on south-facing slopes, where fires were low- or mixed-severity, maintaining multiple cohorts, sizes, ages and layers of host trees. High-intensity fires would typically eliminate most (but not all) mistletoe-infested trees over large areas, and mistletoes would slowly re-invade from the perimeter at the rate of 3 to 4.5 m (10 to 15 ft) a decade (Hawksworth 1958, 1960; Parmeter 1978; Wagener 1965), or from islands of infested trees that escaped burning. Especially in ponderosa pine and Douglas-fir, severe mistletoe infections provide an abundance of mistletoe brooms, fine fuels, resinous stems, branches, and cankers. Even low- and moderate-intensity fires would often torch these trees, destroying severely infected trees and infection centers (Koonce and Roth 1980, Parmeter 1978, Weaver 1974). No doubt active and passive crown fires were initiated in areas of severe mistletoe infestation.

Dwarf mistletoe in Douglas-fir was probably more common on northerly aspects and in riparian areas, where the interval between fires was longer. Under historical fire regimes, Douglas-fir dwarf mistletoe was probably widely distributed but at low to moderate severity (Arno 1988, Fischer and Bradley 1987, Harrington 1991). Mature Douglas-fir, with thick outer bark and crown bases elevated well above the forest floor, were quite resistant to surface fires. Douglas-fir dwarf mistletoe was well distributed in scattered, thick-barked overstory trees that had developed on young forest dominated landscapes under the influence of low-intensity fire, but further influence was minimal because understory Douglas-fir stocking was minimal.

Conversely, young Douglas-fir had thin, resinous outer bark and crowns close to the forest floor, two characteristics that increased vulnerability to surface fires. When mistletoe brooms occurred on young trees, the likelihood of tree torching was increased (Harrington 1991, Tinnin 1984, Tinnin and Knutson 1980). Under the right wind and weather conditions, fires crowned from mistletoe-infected understories. In addition, mistletoe brooms in Douglas-fir nullified benefits of inter-tree competition and natural branch pruning by maintaining a flammable link with the forest floor. In patches where Douglas-fir was abundant in the understory, e.g., in northerly aspects, Douglas-fir dwarf mistletoe was probably quite abundant.

Given the range of fire frequency and severity in historical MMC forests, the western larch mistletoe was likely the most prevalent and influential in terms of tree growth and mortality. Western larch dwarf mistletoe was perhaps the most widespread of mistletoes in old forest stands. Of all the dwarf mistletoes, larch mistletoe survived fire in overstory western larch with the greatest constancy (Bolsinger 1978), perhaps because of larch's exceptional resistance to damage by fire (Lotan et al. 1981), its resistance or tolerance to both tree-killing and opportunistic root pathogens (Filip and Schmitt 1979, Hadfield et al. 1986), and the lack of primary bark beetle associates. Larch mistletoe brooms are weak and brittle and frequently break off when still relatively small. Under historical fire regimes, branch litter accumulating under infected hosts caused lethal fire scorching of some infected trees (Alexander and Hawksworth 1975). According to Tinnin et al. (1982), the increased burn potential accentuated the advantage of fire-adapted species such as western larch.

Dwarf mistletoes create brooms in trees, sometime quite large, and this produces critically important habitat structure for many species of wildlife. Brooms are used for nesting, roosting, and hiding cover (e.g., see Bull and Henjum 1990, Bull et al. 1989, Forsman 1983, Sovern et al. 2011). Birds and squirrels also contributed to reintroduction of mistletoes to large host patches (Hawksworth et al. 1987). Mistletoes would persist in residual ponderosa pine and western larch overstory trees by virtue of their resistance to fire, or from irregularities in fuel continuity or arrangement, or fire behavior, and the spread of these mistletoes to newly regenerating patches would be much quicker (Parmeter 1978). The presence of mistletoe brooms is a prime example of a long-standing struggle between objectives to manage for wood production (i.e., foresters would be inclined to remove diseased trees) and objectives to maintain or enhance wildlife habitat (wildlife specialists would advocate keeping these trees).

At least 40 percent of all of the Douglas-fir, western larch, ponderosa pine, and lodgepole pine forests east of the Cascade crest are infected with dwarf mistletoe (Bolsinger 1978). Infections are more widely distributed and have had a greater impact on tree health than ever before, except where large wildfires have recently occurred. Because of fire exclusion and selective timber harvesting, many remaining forests are densely stocked and multilayered, conditions that are conducive to spread of mistletoes. Conifers such as Douglas-fir or ponderosa pine with severe mistletoe infections exhibit declining crown vigor and reduced resistance, and are eventually attacked and killed by bark beetles and opportunistic root pathogens like Armillaria root disease (Hadfield et al. 1986, Morrison et al. 1991).

Animals of the Moist Mixed-Conifer Ecosystem

The landscape ecology concepts of resilience and natural range of variation described in the previous sections of this report provide important tools for understanding animal habitat and population function within MMC forests of the Pacific Northwest. Native animals evolved in the context of historical disturbance regimes and resultant landscape patterns. Managing for the natural range of variability (NRV) provides a conceptual starting point for capturing the habitat needs of native wildlife. Landscape resilience is also important to animal populations because it represents the tendency of a landscape to return to conditions that the native animal community is adapted to. Landscapes that are not resilient may shift into novel environmental conditions that can contribute to substantial changes in the animal community and loss of biodiversity. However, it is important to note that simply managing for resilient landscape conditions that are within the NRV may not be sufficient for the conservation of all animals (Wiens et al. 2008).

Native animals evolved in the context of historical disturbance regimes and resultant landscape patterns.

Some native species have been affected by threats not related to habitat amount and distribution, including interactions with invasive species (e.g., northern spotted owls [NSO] [*Strix occidentalis caurina*] and barred owls [*Strix varia*] [Gutierrez et al. 2007]), exposure to exotic diseases (e.g., bighorn sheep [*Ovis canadensis*] [Schommer and Woolever 2008]), or sensitivity to effects of specific human activities (e.g., disturbance associated with roads or recreation [Gaines et al. 2003]). Conservation of these species often requires more specific strategies that address habitat in combination with the specific risks to which they are sensitive. Several recent assessments have used this conceptual approach to compare the current landscape to estimates of habitat patterns under historical disturbance regimes to provide a "coarse filter" for evaluating the amount and distribution of habitat conditions, in conjunction with "fine filter" assessments of conditions for selected individual species that are sensitive to specific risks (Gaines et al., in press; Suring et al. 2011; Wales, et al. 2011, Wisdom et al. 2000). Results of these assessments are summarized in "Effects of Altered Disturbance Regimes, Land Use, and Climate Change on Animal Populations and Habitats" on page 115. Our focus in this section is to provide an overview of important considerations for managing fish and wildlife habitat and animal populations in MMC forests.

As highlighted in previous sections, the MMC forest type is characterized by environmental conditions that are intermediate between warm/dry and cold/wet forest types that are often adjacent to or intermixed with the MMC. This pattern results in an animal community that is composed of an overlapping mix of species that are also found in the adjacent forest types. The composition of the animal community and abundance of individual species within MMC forests depends on many factors including landscape patterns, forest structure, and how wet or dry conditions are within an individual stand and the surrounding landscape.

Using the wildlife habitat relationships database of Johnson and O'Neal (2001), Lehmkuhl (2005) concluded that bird and mammal communities in eastside interior MMC forests were a mix of species typical of low-severity low-elevation ponderosa pine forest (84 percent species similarity) and high-elevation high-severity mixed-conifer forest (71 percent species similarity). About 40 percent of all species were shared in common among the three types. Mixed-conifer forest was more similar to ponderosa pine forest in supporting relatively more generalist or young forest species than higher elevation forests. Within eastside habitat types and disturbance regimes, the fire-prone ponderosa pine cover type supports the most species of amphibians, reptiles, and birds; whereas mammals are most species-rich in mixed-conifer types (Bunnell 1995; Kotliar et al. 2002; Sallabanks et al. 2001, 2002).

A key concept for understanding animal distribution and abundance in MMC forests is that animals need to acquire a variety of resources to meet their life-history needs and most animals use a variety of different habitats and structural features to acquire those resources. Morrison et al. (1998: 10) described wildlife habitat as "an area with a combination of resources (like food, cover, water) and environmental conditions (temperature, precipitation, presence or absence of predators and competitors) that promotes occupancy by individuals of a given species (or population) and allows those individuals to survive and reproduce." In MMC forests, these wildlife habitat components are emergent properties of the forest communities and their growth, disturbance, and stand dynamics (sensu Oliver and Larsen 1996) processes. How these processes work at a variety of scales is important in determining where habitat components occur and whether they are arranged in a manner that allows animals to survive and reproduce. Like landscapes, areas used by animals can be thought of as "habitats within habitats" (fig. 33).

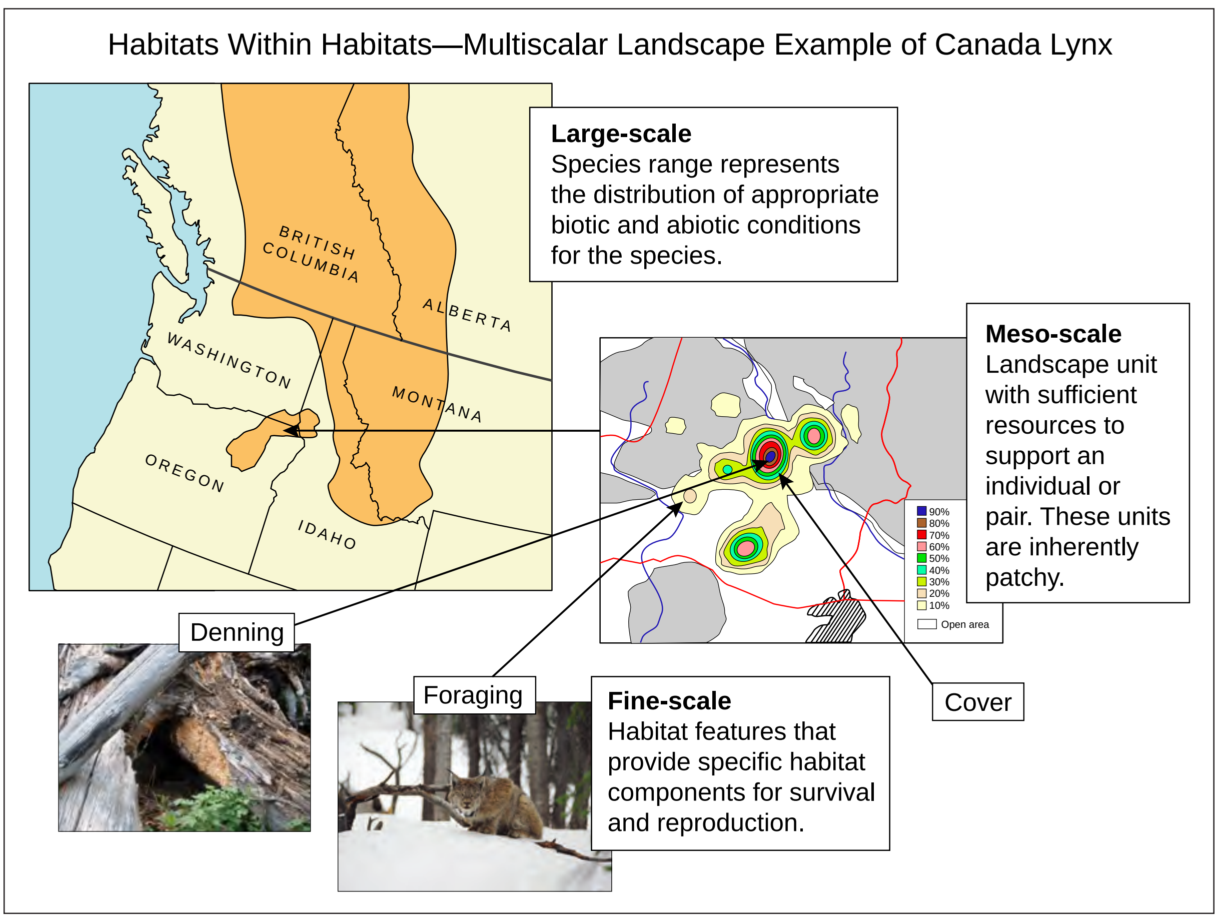


Figure 33—Wildlife habitat is represented at different scales. Individual animals are connected to key features of their environment over a confined spatial domain, small groups of a species interact regularly over a larger area, and populations of a species are distributed across large geographic areas. This multiscalar view illustrates the importance of considering habitat requirements of a species at different spatial scales.

Landscape heterogeneity at a variety of scales plays an important role in determining the assortment of habitat conditions and resources available to the animal community in that landscape.

At the finest scale, animals use habitat features associated with specific forest structure attributes (e.g., snags for foraging and nesting); at the meso-scale (sub-basin or watershed scale), they must find the appropriate configuration of those resources to meet their life-history requirements (e.g., the right combination of food availability and security from predators); and at the broadest scale, animals need to be able to move to find mates, disperse to new areas, prevent genetic isolation, and maintain broad-scale population function (meta-population dynamics). Selection of a finer-scale habitat feature is therefore constrained by habitat selection decisions made at broader scales (e.g., the forest stand an animal chooses to use is constrained by the variety of stands within the watershed where the animal is located [Johnson 1980]).

Landscape heterogeneity at a variety of scales plays an important role in determining the assortment of habitat conditions and resources available to the animal community in that landscape. At the broadest scale, the distribution of wildlife and fish species is determined by regional to subcontinental gradients in climate, topography, soils, and vegetation (Hansen et al. 2011). The tolerances of species to these biophysical gradients result in predictable patterns of community diversity. Across Oregon and Washington, species richness of birds, trees, and shrubs is highest in the Okanogan Highland and Siskiyou ecoregions (Hansen et al. 2006, Swenson and Waring 2006). The lower forest ecotones across the East Cascades, Okanogan Highlands, and Blue Mountains are also high in species diversity (Olson et al. 2001). These locations have intermediate precipitation, warm growing season temperatures, and intermediate primary productivity. These conditions provide diverse food resources, variable vegetation structure, and high levels of habitat diversity, with grassland/shrubland, dry forest, and moist forest habitats all in proximity. Amphibian diversity is low in the eastern Cascades compared to western Oregon and Washington, primarily because the dry climate in the eastern Cascades is less conducive to occupancy by amphibians (Olson et al. 2001).

The spatial patterning of habitats across Oregon and Washington is also important in influencing connectivity for wildlife and fish populations. Genetic diversity, metapopulation dynamics, and population viability for many species is dependent upon the ability of individuals to move across landscapes (Bennett 2003, Crooks and Sanjayan 2006, Hilty et al. 2006). Regional-scale landscape permeability patterns for forested landscapes in eastern Washington and Oregon (fig. 34) are important because these forests serve as important potential source areas for a variety of species in their own right, and they provide critical linkages between ecosystems in the Rocky Mountains, the Cascade Range, and into Canada (Singleton et al. 2002, Theobald et al. 2012, WHCWG 2010).

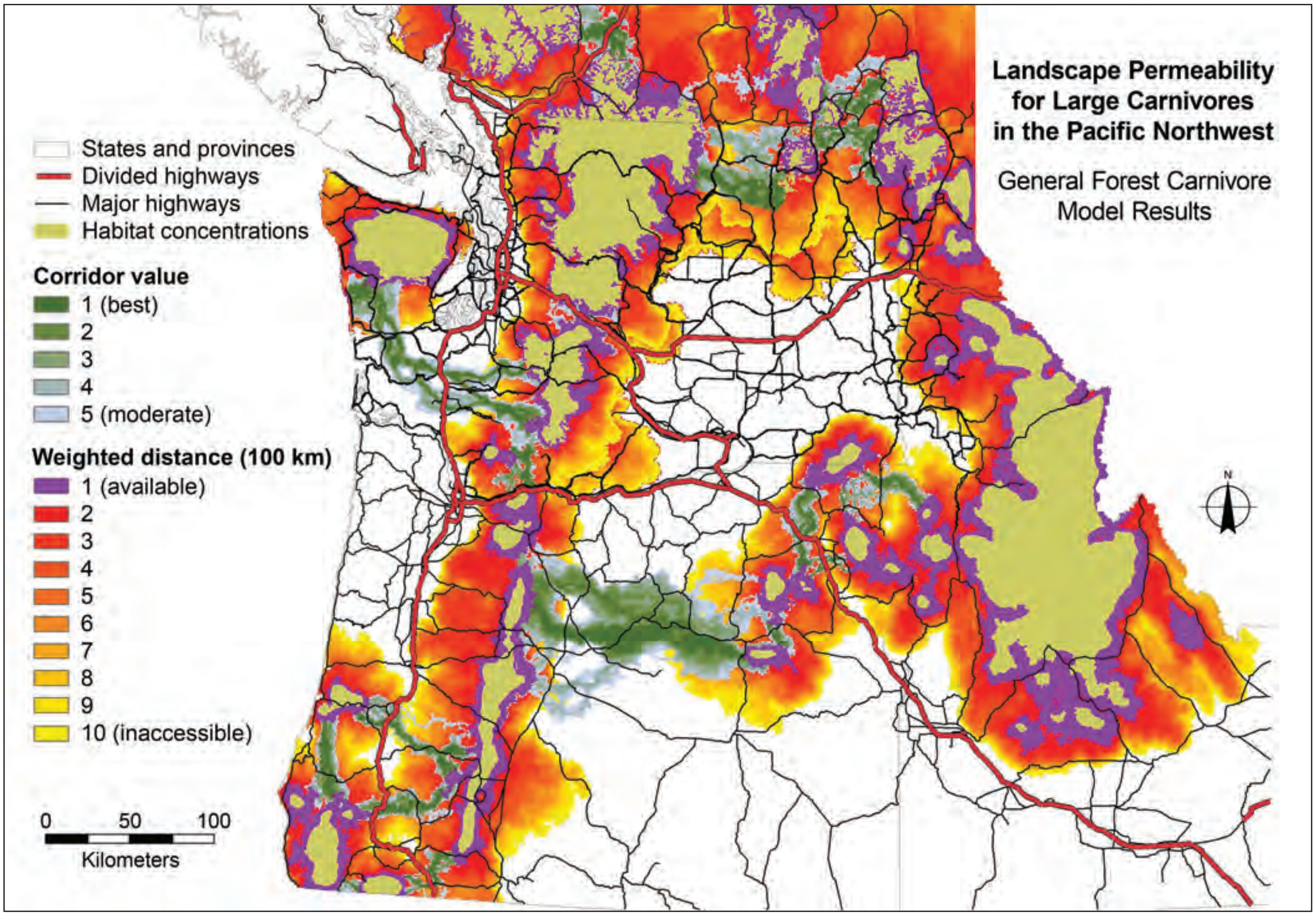


Figure 34—At a regional scale, habitat for large carnivores is more or less permeable to movement, depending on various vegetation and topographic conditions. This information is an important consideration when planning treatments for forests at a landscape scale.

Meso-scale landscape patchiness determines the arrangement of habitat resources for animals. A critical feature of wildlife habitat in mixed-conifer landscapes in eastern Washington and Oregon is the multiscale (landscape and stand) diversity and juxtaposition of patch types of differing composition and structure (Perry et al. 2011). Habitat heterogeneity on presettlement landscapes was largely a result of mixed-severity fire. As reviewed in previous sections of this report, the combination of extensive topographic variability, other intrinsic factors of the landscape (e.g., soils, accumulation of fuels, etc.), and the vagaries of weather interacted in each fire event to create complex mosaics of habitat with soft boundaries that distinguish the effects of varying fire severity. Forest patches in different stages of development typically have different habitat characteristics and provide different sorts of habitat features (Johnson and O'Neal 2001, Thomas 1979). These processes result in a variety of forest structure and composition patterns that can support different terrestrial wildlife communities (Smith 2000). Again, NRV provides a conceptual guide for providing the mosaic of habitat conditions that native species evolved with, but species with specific habitat needs or sensitive to specific risks often require special consideration (e.g., Suring et al. 2011).

Fine-scale within-stand diversity of vegetation composition and structure provides specific wildlife habitat features required for denning, nesting, foraging, and security for individuals of many wildlife species. Different stand development stages provide different habitat structures and resources.

Recently disturbed patches tend to support high plant productivity, and often exhibit a particularly high diversity of plant and animal species, including many specialist species (Hagar 2007, Swanson et al. 2011) (fig. 35). Focal species for assessing early-seral and postfire habitats in interior MMC communities in the Pacific Northwest include fox sparrow (*Passerella iliaca*), white-headed woodpecker (*Picoides albolarvatus*), western bluebird (*Sialia mexicana*), and Lewis's woodpecker (*Melanerpes lewis*) (Gaines et al., in press; Wales et al. 2011).

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Figure 35—Postfire and other recently disturbed forest sites provide productive habitat for many wildlife species. This recently burned area on the Okanogan-Wenatchee National Forest has shrub and herbaceous vegetation that provides habitat features generally not found in more mature forests.

Early successional habitats are also important for ungulate summer forage. Recent research has highlighted the importance of high-quality late-summer forage for elk survivorship and reproduction (Cook et al 2013). Early-seral openings within MMC forests can play a particularly important role by providing late-summer forage resources for ungulates (Lehmkuhl et al. 2013). Several species of migratory songbirds are also associated with deciduous shrub communities found in early-seral forest patches (Betts et al. 2010). Structural legacies (including live trees, snags, and logs) from predisturbance stands contribute important habitat components in recently disturbed areas by providing cavities, perches, escape cover, and protected microclimates (Bull et al. 1997, Franklin et al. 2000, Van Pelt 2008). The abundance of large snags and logs is a particularly important characteristic of post-fire habitats (Smith 2000).

Vertebrate communities within even-age young forest patches are generally less diverse and abundant than those found in other stand development stages, particularly for sites dominated by dense, closed-canopy conditions with little structural or understory diversity. Nevertheless, mid-seral stands can provide important ecological functions, depending on specific habitat conditions and landscape context. Dense even-age stands can provide security cover for prey species where conditions support the prey but prevent effective access for foraging predators. For example, avian predators like NSO and northern goshawks (*Accipiter gentilis*) select areas with structurally diverse forest canopies to facilitate foraging, even while their prey can often be relatively abundant in nearby dense stands (e.g., Greenwald et al. 2005). The presence of biological legacies (big old trees, snags, and logs) from predisturbance stands can provide important structural diversity in mid-seral stands by breaking up an otherwise continuous canopy, providing cavities for nesting and denning, and adding structure to the forest floor (Bull et al. 1997, Franklin et al. 2000). Mid-seral stands also serve an important role as "old growth in waiting." Late-successional forest conditions can be relatively transient in interior mixed-conifer forests compared to forests west of the Cascade Mountains owing to the variety of disturbance processes associated with this forest type (reviewed in this section and in "Natural Disturbance Factors" and "Aquatic Habitats"). Mid-seral forest stands can provide an important "conveyor belt" of forest stands that are developing structural characteristics of older forests that can serve as replacements when other older stands are affected by disturbance. Active management to facilitate the development of old forest characteristics including canopy structural diversity and small openings has become an important principle for management of mid-seral stands and is particularly important for the recruitment of old forest habitat conditions for wildlife (Franklin and Johnson 2012).

Older forests with a diversity of tree ages and sizes (including larger, older trees) provide many unique ecological functions. Across the variety of forest structural conditions found in MMC forests, animal abundance and species diversity are favored by structural complexity in the form of varied tree sizes, abundant snags, and coarse woody debris (McComb 2001). Big old trees, living and dead, standing and down, provide unique and important habitat structures within old forest stands (reviewed by Bull et al. 1997, Thomas et al. 1979, and Van Pelt 2008). For example, snags and down logs provide denning and resting sites for American marten (*Martes americana*) (Bull et al. 2005). Long-legged myotis bats (*Myotis volans*) roost in snags and bark crevices of large old trees (Ormsbee and McComb 1998). Stout lateral epicormic branches can provide nest sites for northern goshawks and other species (Daw and DeStefano 2001). Cavities (in snags or living trees) are particularly important keystone structures for wildlife communities. Such cavities are created by the interactions of pathogens, fire, strong winds, and primary cavity excavators, and are subsequently used by a variety of species (Bednarz et al. 2004, Parks et al. 1999).

The security, thermal, and moisture characteristics found beneath and within large down logs can be especially important for small mammals, amphibians, reptiles, and invertebrates.

The security, thermal, and moisture characteristics found beneath and within large down logs can be especially important for small mammals, amphibians, reptiles, and invertebrates (Bull et al. 1997). Large old trees, especially in dense mixed-conifer patches, have higher loads of epiphytic forage lichens, which are critical winter food for some species (Lehmkuhl 2005).

The multilayer canopies found in older stands contribute to unique thermal characteristics within stands and provide structural complexity important for arboreal mammals and their avian predators. Goshawks and NSO use multilayered canopies (often found in MMC forests subject to periodic fire) because such layering provides room to fly within forest stands. Canopy gaps allow for understory tree establishment and allow for sunlight to penetrate the canopy to the forest floor, contributing to understory productivity and diversity. Understory productivity is the foundation of food availability for many species. For example, northern flying squirrels (*Glacomys sabrinus*) are more abundant in stands with diverse understories with plants that provide critical fruit and mast foods (Lehmkuhl et al. 2006b); diverse understory vegetation can provide important forage resources for ungulates (Lehmkuhl et al. 2013); and fruit-bearing shrubs provide important food resources for black bears (*Ursus americanus*) and other species (Lyons et al. 2003).

Typical focal species for assessment of MMC old forest habitat conditions in Washington and Oregon are northern goshawk and pileated woodpecker (*Dryocopus pileatus*) (Gaines et al., in press; Wales et al. 2011; Wisdom et al. 2000). Northern goshawks are broadly distributed throughout the Interior West, and are

Impacts of Wildlife Herbivory

Herbivory (i.e., grazing and browsing) by wild ungulates, primarily elk, can significantly affect forested and grassland ecosystems and is worthy of consideration in any management plan (see Lehmkuhl et al. 2013 for a recent review). Elk and deer can account for a significant portion of the animal unit months (AUMs) on a range: e.g., elk and deer made up about 90 percent of the estimated 78,000 AUMs on the Yakima elk herd range in the eastern Washington Cascades. Utilization rates in grassland and forest are typically 40 to 60 percent of annual production, with herbivory concentrated in productive riparian areas and meadows but occurring in all forest types. Herbivory effects may intensify in some areas with a spring and summer forage base that has declined since the early 1900s with meadow contraction, development of dense forests due to fire suppression, regrowth of old burns and pre-1990 clearcuts, and infrequent forest clearcut harvesting as a result of regulatory changes during the last 20 years. Sustaining livestock and elk on a diminishing habitat base may have degraded habitat and enhanced establishment of exotic species in sensitive meadow and riparian areas.

The impact of wild ungulate herbivory has been well documented in the Blue Mountains from long-term exclosures and other research. In Douglas-fir and grand fir forests, Riggs et al. (2000) found that ungulate herbivory seriously depressed biomass of shrub genera *Acer, Linnaea, Amelanchier, Salix, Sorbus, Ceanothus, Physocarpus,* and *Rubus* outside of 25-year-old exclosures, within which understory biomass was two times greater than that outside exclosures. Johnson (2007) found that the shrubs *Rosa, Symphoricarpos,* and *Purshia* increased dramatically in the absence of long-term browsing in shrubland communities in the Blue Mountains. Tiedemann and Berndt (1972) reported five times more shrub cover, primarily snowbrush ceanothus (*Ceanothus velutinus*) with Saskatoon serviceberry (*Amelanchier alnifolia*), chokecherry (*Prunus virginiana*), and common snowberry (*Symphoricarpos albus*), inside versus outside a 30-year-old exclosure in a Douglas-fir/pinegrass (*Calamagrostis rubescens*) habitat type dominated by ponderosa pine with Douglas-fir. Likewise in the eastern Washington Cascades, Lehmkuhl et al. (2013) documented high long-term grazing impacts on some species of typical and atypical forage shrubs, such as common snowberry, oceanspray (*Holodiscus discolor*), and pinemat manzanita (*Arctostaphylos nevadensis*) in mid-elevation Douglas-fir and grand fir cover types, sticky currant (*Ribes viscosissimum*) in high-elevation subalpine fir cover types, and little sagebrush (*Artemisia arbuscula*), scabland sagebrush (*A. rigida*), and desert bitterbrush (*Purshia tridentate*) in low-elevation nonforest cover types. Impacts on ungulate herbivory on shrubs are likely underestimated because historically high levels of herbivory by both wild ungulates and livestock in the study area already may have suppressed or eliminated highly palatable shrubs.

Proposed forest restoration management in the Blue Mountains (mainly the Douglas-fir and dry grand fir types) to reduce uncharacteristic fuel loads and restore resilient stand structures and processes likely will increase the forage base for ungulates by reducing canopy cover to <40 percent and stimulating understory development (Hedrick et al. 1968, Long et al. 2008, Lyon and Christensen 2002, Skovlin et al. 1989, Vavra et al. 2005; but see Sutherland and Nelson 2010). Opening the canopy of long-closed stands with diminished seed banks of herbaceous species might increase invasive species cover (McGlone et al. 2009, Sabo et al. 2009; but see Wienk et al. 2004); however, responses of invasive species can vary considerably because of site conditions (Sutherland and Nelson 2010) and can be mitigated by management (e.g., weed control along roads). See Lehmkuhl et al. (2013) for a complete discussion of restoration impacts on ungulate habitat.

associated with structurally diverse old forest conditions that provide nest sites and foraging opportunities (reviewed by Greenwald et al. 2005, and Squires and Kennedy 2006). Pileated woodpeckers are also broadly distributed in wet to dry mixed-conifer forests throughout the Interior West. They are strongly associated with large snags and down logs for nesting and foraging (Bull and Holthausen 1993, Bull et al 1992). American marten are also included as a focal species for assessment of old forest conditions, but are more closely associated with forest types that are colder and wetter than the moist mixed-conifer forest (Gaines et al., in press; Munzing and Gaines 2005).

Conservation concerns about the NSO, a federally listed threatened species, have had a substantial impact on management of MMC forests in the eastern Cascade Range (USFWS 2011). We do not have the space for an exhaustive review of the substantial literature on NSO in the eastern Cascade Range, however we will highlight a few key aspects of spotted owl biology directly related to MMC forests. These highlights are important for understanding spotted owl management in MMC forests, but they also provide some useful insights for conservation of other old forest species associated with MMC forest outside of the range of the NSO (management considerations for old forest species are discussed in "Effects of Altered Disturbance Regimes, Land Use, and Climate Change on Animal Populations and Habitats" on page 115).

Spotted owls in the eastern Cascade Range have similar requirements for spatial and structural heterogeneity as in other parts of their range (reviewed by Courtney et al. 2004), but differences in disturbance processes (i.e., mixed-severity fire regimes, past selection harvest practices, and presence of mistletoe clumps) contribute to some differences in apparent habitat associations. The classic description of NSO nesting, roosting, and foraging habitat is "a multilayered, multispecies canopy dominated by large (>76.2 cm [30 in] diameter at breast height [d.b.h.]) conifer overstory trees, and an understory of shade-tolerant conifers or hardwoods; a moderate to high (60 to 80 percent) canopy closure; substantial decadence in the form of large, live coniferous trees with deformities—such as cavities, broken tops, and dwarf mistletoe infections; numerous large snags; ground-cover characterized by large accumulations of logs and other woody debris; and a canopy that is open enough to allow owls to fly within and beneath it" (Thomas et al. 1990: 164). Although NSOs in the eastern Cascade Range inhabit patches that meet this description (particularly in moist sites), they also use patches with smaller trees, where structural heterogeneity is enhanced by the presence of mistletoe brooms and biological legacies from previous, more open stand conditions (Buchanan et al. 1995, Everett et al. 1997, Sovern et al. 2011). These stand structure conditions are often highly divergent from historical conditions (Everett et al. 1997, Hessburg et al. 2005, Lehmkuhl et al. 1994), and provide fuel characteristics and spatial patterns that are conducive to uncharacteristically high-intensity, widespread wildfire events (Agee 2003).

Spatial and structural heterogeneity is also important for spotted owl prey. Bushy-tailed woodrats (*Neotoma cinerea*) and northern flying squirrels compose more than 50 percent of NSO prey biomass in these areas (Forsman et al. 2001, 2004). Lehmkuhl et al. (2006a, 2006b) found that bushy-tailed woodrats and northern flying squirrels were associated with the presence of large snags, mistletoe brooms, and downed logs, with woodrats being found in both mixed-conifer and ponderosa pine types. Presence of diverse understory vegetation that provided a variety of food items was also important for flying squirrels (Lehmkuhl et al. 2006b).

Competitive interactions with barred owls are an important factor contributing to recent NSO population declines (Forsman et al. 2011). Spotted owl populations in Washington and northern Oregon declined by approximately 40 to 60 percent from 1989 to 2008 (Forsman et al. 2011). Many experts expect that NSO populations will continue to decline through much of their range as barred owl numbers increase (USFWS 2011). Barred owls in the eastern Cascade Range are most abundant in flatter valley bottom moist forest settings (Singleton et al. 2010). Interactions with barred owls may have contributed to the displacement of NSO pairs into drier midslope settings as early as the 1990s when most NSO activity centers were documented, resulting in relatively few historical NSO sites being documented in valley-bottom MMC settings that otherwise appear to be suitable NSO habitat (Singleton 2013). Persistence of the NSO population in mixed-conifer forest landscapes may depend on the ability of the local NSO population to adapt to the presence of barred owls (Gutteriez et al. 2007). Substantial habitat loss from natural or human-caused disturbances (e.g., fire or forest management activities) has the potential to exacerbate the negative impacts of interactions with barred owls by increasing competition for limited habitat resources (Dugger et al. 2011, Forsman et al. 2011).

Several other unique habitats also occur in landscapes that support MMC forests, including deciduous forest patches, meadows, riparian areas, rock piles, cliffs, and caves (reviewed in Johnson and O'Neil 2001 and Thomas et al. 1979). Deciduous tree patches provide important habitat characteristics for a variety of species (Hagar 2007), particularly migratory birds (Betts et al. 2010), and can provide abundant cavities for woodpeckers and other species (Martin et al. 2004). Aspen stands were an important component of the historical landscape in areas that support MMC forest, but the availability of aspen habitats has been reduced substantially by altered disturbance regimes and herbivory impacts from both native and domestic animals (Shirley and Erickson 2001, Strong et al. 2010). Natural moist meadows are another particularly productive community within landscapes that support MMC forest. Meadows provide important ungulate forage resources (Lehmkuhl et al. 2013) and contribute to unique edge conditions that are important for a variety of wildlife including great grey owls (*Strix nebulosa*) (Bull and Henjum 1990) and long-eared owls (*Asio otus*) (Bull et al. 1989).

Area of meadows has been reduced by tree encroachment as a consequence of altered disturbance regimes (Haugo and Halpern 2007). Meadow communities have also been altered by domestic livestock grazing and intense use by native ungulates (Beebe et al. 2002). Large carnivores (particularly wolves) historically played an important role in limiting concentrations of native ungulates in these highly productive areas (Beschta and Ripple 2009). Ecological functions of streams and associated riparian areas are reviewed below, but it is worth emphasizing that in the context of animal habitat in disturbance-prone landscapes, riparian areas can provide unique linear landscape features that contribute to important ecological flows through the landscape, including animal movement. Rocks, cliffs, and caves also provide unique landscape elements that can be important for many species (Thomas 1979). Rocky talus and scree slopes can provide unique habitat features for amphibians including larch mountain salamander (*Plethodon larselli*) (Crisifulli 2005) and secure areas for rodents including bushy-tailed woodrats and pika (Simpson 2009). Cliffs provide nesting opportunities for raptors (including peregrine falcon [*Falco peregrinus*]), and caves can provide hibernacula for bats.

The literature on wildlife habitat associations within MMC forests in western North America is vast and beyond the scope of this document. However, two main themes emerge from our review of the information: (1) natural disturbance processes play a key role in the development of stand structure conditions and landscape patterns that determine habitat values for animals, and (2) biological legacies (big old trees, snags and logs) provide important habitat structures across all stages of stand development. Within-stand structure is influenced by disturbance processes that function at a variety of scales. Endemic levels of disturbance can contribute to stand and landscape heterogeneity, while large-scale, high-intensity disturbances can reduce stand and landscape heterogeneity. Within-stand structural diversity is generally much greater in stands with a variety of tree age and size classes, even if those stands are recently disturbed or dominated by younger trees. Big trees and logs are often a legacy of previous stand conditions that have been retained after some disturbance event, like fire or harvest (Franklin et al. 2000). The variety of extremely valuable habitat structures provided by these large old trees include cavities, large branches, broken tops, brooms, and platforms (Bull et al. 1997). The presence of large old trees within a stand can make a big difference for wildlife habitat values in both old and young forests.

Aquatic Habitats

From the literature, two concepts emerge as foundational for managing linked terrestrial and aquatic ecosystems: (a) watersheds and their aquatic habitats and species are dynamic and adapted to insect, disease, weather, and wildfire disturbances, and (b) the climate will continue to have a profound influence on terrestrial and aquatic ecosystems, disturbance processes, and their interactions (Bisson et al. 2003; Luce et al. 2013, 2014; Tague and Grant 2009).

Disturbances play a vital role in structuring aquatic ecosystems.

Disturbances play a vital role in structuring aquatic ecosystems. Wildfires influence hydrological and physical processes, such as surface erosion, sedimentation, solar radiation, wood recruitment, and nutrient exchange in streams (Benda et al. 2003, Luce et al. 2009, Miller et al. 2003, Wondzell and King 2003). The timing and severity of erosion and sedimentation differ by the physical geography, geology, and geomorphic processes; precipitation; and fire regime. Erosion contributes to sedimentation and can depend on riparian vegetation density and the speed of vegetation recovery after disturbance. Chronic erosion delivers fine sediment for a fairly long time, usually in the absence of revegetation after disturbance, or it comes from road rights-of-way, trails, and bulldozer lines. Postfire riparian erosion results in larger pulsed sediment and wood delivery to streams, and in some circumstances, channels can be reorganized, affecting aquatic habitats (Benda et al. 2003, Miller et al. 2003, Minshall 2003, Reeves et al. 1995). Water retention properties of the soils and reduced evapotranspiration from vegetation loss in the surrounding landscape affect runoff dynamics. Over time, coarse wood and sediment are depleted. Fluvial action and rot tend to break down stream wood, which can then be transported downstream along with sediment and rock. These processes continue until the system is replenished by subsequent postfire erosional events (Benda et al. 2003, Miller et al. 2003).

In general, episodic large-scale disturbances (e.g., fire) to aquatic ecosystems are inevitable and often beneficial when spaced out over long periods, and this knowledge can form an important ecological foundation for fire and forest management. Anadromous and resident species in such landscapes evolved with these disturbances and the attendant shifts in habitat quantity and quality. These species are affected by habitat reorganization associated with wildfires, including addition of wood material and sediment deposition; thus, they are considered to be fire-adapted. The natural frequency, severity, and extent of historical fires governed the pulse of erosional events that carried wood, rocks, and soil to streams and created variation in the level of stream shading and aquatic habitat available to fish provided by riparian vegetation. Fish populations have been shown to recover rapidly to the natural fire regime in burned reaches, depending on connectivity of stream networks (Burton 2005). Thus, restoration of conditions that promote a natural fire regime will benefit fish populations overall. Where fish populations are robust, recolonization of stream segments disturbed by fire is tenable (Burton 2005).

This dynamic view diverges from traditional frameworks that suggest that aquatic ecosystems should be managed as stable systems perpetually maintained for select species. Stable equilibrium and balance of nature views are equally intractable in terrestrial and aquatic ecosystems. The dynamic view of aquatic systems accepts patterns of disturbance and recovery across landscapes as necessary processes to interconnect mosaics of diverse and changing habitats and communities.

Streams draining burned areas of the Entiat Experimental Forest (EEF) on the east side of the Cascades crest in Washington had peak flows 120 percent higher than prefire conditions (Siebert et al. 2010). Additionally, there were deeper snowpacks and more rapid snowmelt as a result of the disturbance. However, only the upper elevations of the EEF consist of mixed-conifer forest with slightly higher moisture (mean annual precipitation at mid-elevation = 580 mm) (Seibert et al. 2010). Similar hydrologic issues could likely prevail following fire disturbance in moist forests. The primary lesson from Siebert et al. (2010) is that prefire hydrologic data are essential to quantifying the postfire response.

Climate change affects forest landscapes, wildfire regimes, aquatic habitats, management options, and their interactions. Climate variability can have important effects on stream hydrology (Jain and Lall 2001, Poff et al. 2002, Tague and Grant 2009) and related physical processes (Bull 1991, Meyer et al. 1992, Pederson et al. 2001, Schumm and Hadley 1957). Changes in hydrology can happen rather abruptly (i.e., 10 to 100 years), and decadal- to multidecadal-scale climate regime shifts can influence stream flows more than the management practices on which we focus most attention (Jain and Lall 2001). April 1 snowpack has declined in mountainous regions across the Western United States (Mote 2003, Mote et al. 2005). Changes are largely attributed to elevated winter and spring temperatures (Hamlet et al. 2007, Stewart et al. 2005). Snowpack decline is expected to continue as temperatures rise throughout the region. Changes in the timing and magnitude of precipitation are expected because of interactions between rising air temperatures, snowfall, and rainfall across complex local terrain. Warming will result in more precipitation falling as rain than snow, and earlier snowmelt timing (e.g., Hamlet et al. 2005).

Climate change profoundly affects processes that create and maintain aquatic habitats. Some effects are direct, particularly those involving stream temperature, water yield, peak flows, and timing of runoff (Luce et al. 2014). Other effects occur indirectly as climate change forces alteration of the structure and distribution of forest communities and the characteristics of wildfire. These processes will indirectly affect fish in relevant watersheds. Altered snowmelt run-off regimes in the mixed-conifer forests will affect downstream fish-bearing reaches. Discharge patterns affect water depth; thus, earlier low flows downstream may reduce habitat availability for fish dependent on deeper, slower flowing habitats. Changing precipitation and fire regimes are expected to compound the effects of warming trends by shifting hydrologic patterns and those of sediment transport and solar radiation (Dunham et al. 2007, Isaak et al. 2010). The structure and composition of riparian vegetation imparts important influences on temperature and soil moisture gradients immediately adjacent to streams (Anderson et al. 2007).

Fish and other aquatic biota are likely to be affected by wildfire and climate change in MMC forests. Aquatic ecosystems consist of interacting species at all trophic levels. Productivity in each level can depend on inputs and transfers across the terrestrial-aquatic ecotone. Movement of terrestrially derived production into streams can affect primary production with implications further up the food web, including aquatic insects and fish (e.g., Baxter et al. 2004, Nakano et al. 1999). Resources transported downstream and organic matter of terrestrial origin can alter downstream fish-bearing habitats according to spatial variation in the headwaters (Baxter et al. 2004, Binckley et al. 2010, Wipfli et al. 2007). Convergence of low-order streams that drain large landscapes at downstream habitats potentially results in productivity "hot-spots" (e.g., Kiffney et al. 2006). Efforts to establish transport distances and specific responses in fish have been limited in mixed-conifer areas (but see Polivka et al. [N.d.][1]). Nevertheless, some successional stages can have measurable influence on the standing crop of aquatic macroinvertebrates (Medhurst et al. 2010).

Streams in dry and MMC forests are not especially diverse in terms of fish species, but they do consist of species that are sensitive to temperature increases in spawning and rearing streams and that respond to disturbances such as wildfire. Less is known about how fire will affect macroinvertebrate communities that serve

[1] Polivka, K.M.; Dwyer, G.; Mehmel, C.J.; Sirianni, K.M.; Novak, J.L. [N.d]. Effects of temperature and virus persistence time on wild and pesticide strains of the nucleopolyhedrovirus (NPV) pathogen of larval Douglas-fir tussock moth (*Orgyia pseudotsugata*). Manuscript in preparation. On file with: Karl Polivka, Pacific Northwest Research Station, Forestry Sciences Laboratory, 1133 N. Western Ave., Wenatchee, WA 98801.

as food resources for fish. Successional patterns following disturbance can affect macroinvertebrate communities, but it is unclear whether changes at one trophic level will be detectable at levels above (i.e., fish) (Medhurst et al. 2010).

In fish-bearing streams of Idaho, the intensity of fire determined the density and age structure of rainbow trout (*Oncorhynchus mykiss*) (Dunham et al. 2007). Relative to controls, streams adjacent to habitats that burned showed relatively more rapid growth of age 1+ rainbow trout, an effect that was augmented in streams that were physically re-organized by erosion and sedimentation following the fire (Rosenberger et al. [N.d.]). This study ranged from lower elevation dry forest to elevations containing MMC and subalpine forest in the Boise River basin. Fish recovery following riparian fire can be fairly rapid in streams where there is a robust local population to recolonize disturbed habitat, and management of fire severity might be most important in areas where local populations are weak and isolated (Burton 2005). Management of roads might also be necessary, given that fragmentation of stream habitat might not only affect the ability of a fish population to recolonize stream segments affected by fire, but might also affect patterns of genetic diversity in resident trout species (Neville et al. 2009).

Cutthroat trout (*Onchorhynchus clarki*) and bull trout (*Salvelinus confluentus*) are the most temperature-sensitive species likely to be present in mixed-conifer zones. Bull trout occupy the coldest freshwater habitats of all salmon and trout species and these habitats are predicted to be severely affected by climate change in coming decades (Isaak et al. 2010, Rieman et al. 2007, Wenger et al. 2011). Bull trout life histories include migratory forms that spawn and rear in cold streams close to headwaters, then migrate downstream to larger rivers where they live as adults prior to spawning ("migratory") or migrate to lakes following rearing ("adfluvial"). Fragmentation of cold water habitats by stream warming can increase physiological stress to bull trout and decrease interconnectivity of adequate spawning and rearing aggregations. In the "resident" life history form, bull trout remain in cold headwater streams for spawning and rearing, and as adults. Thus, they are subject to these physiological stressors at all life stages. Cutthroat trout are also common to upland streams in mixed-conifer forests of the eastern Cascades. Management for the persistence of these and other coldwater fish species in the face of climate change should again focus on maintaining strong, genetically diverse populations in well-connected stream networks. This may mean road management as a means of addressing climate change, by increasing passage and opening stream networks. In small streams within mixed-conifer watersheds of the Wenatchee River sub-basin, cutthroat trout density and total biomass are limited by higher flows (fig. 36).[2] Thus, changing flow regimes as a result of precipitation and snowmelt-timing shifts can potentially have consequences for the persistence of populations of this species.

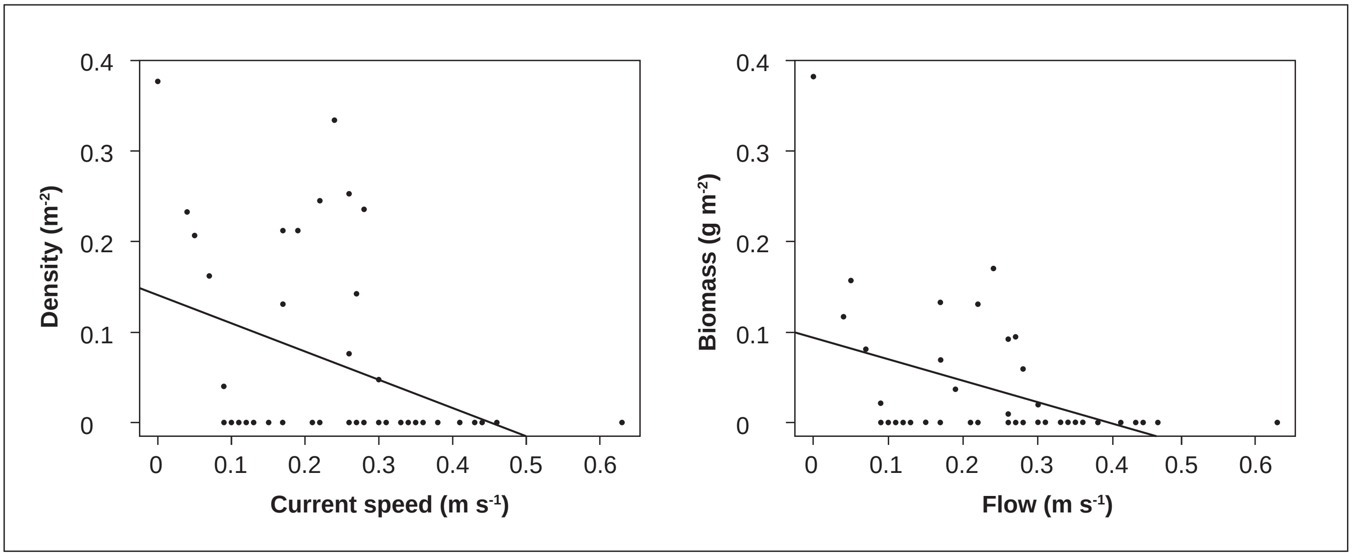

Figure 36—(Left) Cutthroat trout (*Oncorhynchus clarki*) density as a function of current velocity in habitat units (usually pools) distributed among six first-order streams in moist mixed-conifer forests in the Wenatchee River Basin (Washington, USA). Decreasing linear relationship was significant with much unexplained variation ($F_{1,43} = 7.75$, $p = 0.008$, $r^2 = 0.133$). (Right) Cutthroat trout biomass as a function of current velocity in habitat units (usually pools) distributed among six first-order streams in moist mixed-conifer forests in the Wenatchee River Basin (Washington, USA). Decreasing linear relationship was significant with much unexplained variation ($F_{1,43} = 8.34$, $p = 0.006$, $r^2 = 0.143$) (see footnote 2).

Active management can restore a full spectrum of ecological patterns and processes. Long-term restoration and maintenance of the physical, biotic, and ecological processes that are important to maintaining diverse terrestrial and aquatic systems requires strategies that go beyond simply treating fuel accumulations, attempting to prevent high-severity fires, or attempting to maintain existing fish strongholds. The most effective means to minimize negative consequences of expected climate change, and related effects on aquatic systems, is to protect the evolutionary capacity of these systems to respond to disturbance. In this light, management would focus on protecting relatively undisturbed aquatic habitat, and, where necessary, restoring habitat structure, the processes that support it, and the life history complexity of native species, to the best practical extent (Gresswell 1999). Restoring degraded aquatic ecosystems requires a similar perspective. To conserve or promote resiliency in aquatic (as in terrestrial) systems,

[2] Bennett, R.L.; Polivka, K.M. [N.d]. Factors important to monitoring of headwater fish populations and possible effects of landscape features. Manuscript in preparation. On file with: Karl Polivka, Pacific Northwest Research Station, Forestry Sciences Laboratory, 1133 N. Western Ave., Wenatchee, WA 98801.

land managers can turn their focus to conserving and restoring the physical and biological processes and patterns that create and maintain diverse networks of habitats and populations, rather than engineering the condition of the habitats themselves (Benda et al. 2003, Ebersole et al. 1997, Frissell et al. 1997, Gresswell 1999, Minshall 2003, Naiman et al. 2000, Rieman et al. 2003). This means that management is capable of restoring (1) natural patterns in the timing and amount of stream flows (Poff et al. 1997); (2) natural production and delivery of coarse sediment and large wood to streams (Beechie and Bolton 1999, Meyer and Pierce 2003, Reeves et al. 1995); (3) riparian communities that function as sources of organic material, shade, and stream buffering (Gregory et al. 1991); (4) streams, flood plains, and hyporheic zones that are reconnected (Naiman et al. 2000); and (5) habitats that are required for the full expression of native life history strategies, gene flow and variability, and demographic support among populations (Dunham et al. 2003, Gresswell et al. 1994, Healey and Prince 1995, Poole et al. 2001, Rieman and Dunham 2000; Rieman et al. 2003, Roghair et al. 2002). Management maintains forests and streams that can respond to and benefit from disturbances across a broad range of event sizes and intensities, rather than minimizing the threat of the disturbance.

Logical priorities for restoration activities emerge from an evaluation of the changes and constraints imposed by these changes (e.g., Beechie and Bolton 1999, Luce et al. 2001, Pess et al. 2002). Habitat loss and fragmentation, channelization, chronic sediment inputs, accelerated erosion, and changes in hydrologic regime (Lee et al. 1997, NRC 1996) are all problems that merit attention. However, restoring physical connections among aquatic habitats may be one of the most effective first steps to restoring productivity and resilience of many native fish populations (Rieman and Dunham 2000, Roni et al. 2002), given that network connectivity is a key component of the adaptive response by fish to disturbances (Burton 2005). Eliminating the threat of large and severe disturbance may be insufficient to prevent local fish population extinctions in many streams (Dunham et al. 2003, Rieman et al. 2003).

The geographic location and sensitivity of watersheds can be used to guide priority setting for management actions (Rieman et al. 2000, 2010). From an aquatic conservation and restoration perspective, priorities for active vegetation and fuels management occur as follows:

Priority 1. Watersheds in which the threat of extensive, high-severity fire is high and local populations of sensitive aquatic species are at risk because they are isolated, small or vulnerable to invasion of exotic fish species (Dunham et al. 2003, Kruse et al. 2001). In these instances, the first priority for management is to restore connectivity among patches of favorable fish habitat (Dunham et al. 2003, Rieman et al. 2003). Where this is impractical, active forest management to reduce the severity and impact of potential fires could be an important short-term strategy (Brown et al. 2001, Rieman et al. 2003).

Priority 2. Watersheds in which there is little to lose, but much to gain. In some watersheds, habitat degradation is extensive and remaining native fish populations are depressed or locally extinct. Watersheds that are heavily roaded and influenced by past intensive management often contain forests vulnerable to severe fires (Hessburg and Agee 2003, Rieman et al. 2000). Existing road systems can initially facilitate understory vegetation treatments and fuel reduction, and can subsequently be removed, moved, or improved to reestablish hydrologic and biological connectivity (e.g., Roni et al. 2002). Here, short-term risk of ground-disturbing activities may be offset by the potential long-term benefit of reconnecting and expanding habitats and populations. In many of these locations, ongoing treatment with fire will likely be needed.

Priority 3. Watersheds in which sensitive aquatic species are of limited significance. Given that the vulnerability of dry and MMC forests to high-severity fire is associated with lands that have been intensively managed (and absent any subsequent composition or stocking management following treatment), the need for active fire and fuels management now may be greatest in areas where aquatic ecosystems and related physical processes have been most significantly altered (Rieman et al. 2000). In some locations, complete restoration of all native plants and animals may be impractical. These are logical places to experiment with active management, where learning can proceed without taking unacceptable risks (Ludwig et al. 1993; see Rieman et al. 2010 for an example).

Management for climate change requires consideration of dynamic hydrologic simulations that are used to represent climate change scenarios (e.g., see Hamlet et al. 2005, Miller et al. 2003). These must relate sensitivity of models and inferences to the main assumptions about climate change, and the low-frequency climate variability that is assumed (e.g., decadal and longer scale fluctuations). Additionally, dialogue between landscape ecologists and hydrologists will be necessary to integrate the effects of climate change on watersheds in MMC forests and how these will be transmitted across the terrestrial-aquatic ecotone to stream habitats. In other words, climate will affect fire frequency and intensity, and management options for aquatic habitats will require consideration of processes discussed above and how the ecological outcome is related to them (Bisson et al. 2003). Landscape heterogeneity will contribute to the issues faced on the ground by managers in MMC ecosystems throughout the interior northwest (Rieman et al. 2003).

There will be key uncertainties to consider in the management of aquatic habitat in MMC landscapes as well. Variable incidences of fire, variable need for fuels treatment, variable current quality of aquatic habitat, and management tradeoffs between human well-being (property, resources) and conservation all increase the complexity of management decisions (Bisson et al. 2003). Establishment of an effective adaptive management program will incorporate all of these considerations, as well as the spatial and temporal scale. Specific areas might require a more explicit management approach, whereas across the entire MMC forest ecosystem, a more generalized approach based on existing research might be effective (Bisson et al. 2003). Success will depend on continued dialogue between landscape ecologists, aquatic ecologists, and forest managers. Also stakeholders in the public should be asked to consider the scientific recommendations and to participate in further development of management and research objectives.

Riparian habitats—

There are relatively few studies that focus specifically on the historical disturbance regimes and structure and composition of riparian zones of mixed-conifer forests. The little that we do know comes from what can be gleaned from the studies of Agee (1988, 1994), Camp (1999), Camp et al. (1997), Everett et al. (2003), Fetherston et al. (1995), Garza (1995), Gregory et al. (1991), Gresswell (1999), Naiman and Decamps (1997), Naiman et al. (1993, 1998), Olson (2000), Olson and Agee (2005), Skinner (1997), Taylor and Skinner (1998, 2003), and Wright and Agee (2008). With the exception of low-gradient stream reaches (e.g., ≤5 degrees of in-stream slope angle), the fire regimes of riparian zones track fairly well with those of the adjacent upslopes. In contrast, the disturbance regime of low-gradient and often fish-bearing reaches tracks with the hydrologic regime. Low-gradient reaches were often depositional zones during flood events, with intact flood plains, and often supported hardwood tree and shrub vegetation cover. Disturbance in these environments was typically driven by flooding and ice flows, and their spatial and temporal variability, rather than wildfires, although wildfires do burn through these areas.

Riparian areas are key wildlife habitats in interior dry forests because of the presence of free water, cool moist microclimates, and soil moisture that supports diverse vegetation.

Gaining a better understanding of the historical disturbance regimes and vegetation patterns of riparian zones is a fertile area of research. The existing datasets of the Interior Columbia River Basin mid-scale assessment (Hessburg et al. 1999a, 2000a) provide extensive reconstructions of now nearly 400 subwatersheds whose riparian zones could be reevaluated to provide further insights on successional patterns and historical fire severity.

We also know that riparian areas are key wildlife habitats in interior dry forests because of the presence of free water, cool moist microclimates, and soil moisture that supports diverse vegetation (Lehmkuhl et al. 2007b, 2008). Small mammals are more diverse in dry forest riparian areas compared to adjacent uplands. Several species, such as the water shrew (*Sorex palustris*), are associated with free water, whereas species typical of wet or mesic forests become obligate riparian species as the surrounding upland vegetation becomes dryer (Lehmkuhl et al. 2008). Forest birds are no more diverse in riparian areas than in adjacent uplands, but the composition of birds differs with the addition of many riparian obligate species associated with deciduous trees, dense shrubs, and herbaceous vegetation (Lehmkuhl et al. 2007b).

Postfire riparian conditions can have long-term effects on aquatic biota owing to changes in temperature and sedimentation according to local geomorphology and burn severity. In fish-bearing streams of Idaho, the intensity of fire determined the density and age structure of rainbow trout (Dunham et al. 2007). Relative to controls, streams adjacent to habitats that burned showed relatively more rapid growth of age 1+ rainbow trout, an effect that was augmented in streams that were physically reorganized by erosion and sedimentation following the fire (Rosenberger et al. [N.d.]). This study ranged from lower-elevation dry forest, to elevations containing MMC and subalpine forest in the Boise River basin. These observations are thus relevant to management strategies in mixed-conifer forests at higher elevations.

Human Impacts to Moist Mixed-Conifer Systems: Influences of the Last 100 to 150 Years

Humans have inhabited parts of eastern Oregon and Washington for more than 10,000 years. Native American communities developed many land management practices to serve their needs for sustenance and used fire extensively in woodland environments east of the Cascades (Agee 1993; Langston 1995; Robbins 1997, 1999; Robbins and Wolf 1994; White 1983, 1991, 1992, 1999). However, there is limited knowledge of the impacts of their management on MMC forests (Whitlock and Knox 2002).

With the arrival and settlement of Euro-Americans in the early 1800s came another wave of human impacts. Key change agents included initial widespread timber harvests, highly effective fire prevention and suppression (largely since the 1930s), extensive sheep and cattle grazing and livestock fencing, development of extensive road and railroad networks, subdivision of regional landscapes by ownership, widespread and repeated timber harvest entry via selection cutting and clearcut logging, conversion of native grasslands and shrublands to agricultural uses, and urban development. As a result, today these forests neither resemble nor function as they once did 100 or 200 years ago.

Changes to the landscape that naturally flowed from these impacts include:

- A highly fragmented range of forest patch sizes, stemming from 4, 8, and 16 ha (10, 20, and 40 acre) treatment areas and from other land uses near urban areas. The resulting patchiness differed greatly from the original spatial pattern of the landscape. In some places, patches created by repeated harvesting entries created homogenous forest with a skewed species composition and more uniform age and size classes. The original mosaic, more of a gradient of seral stages created by the complex patterns of fire and other tree mortality factors, included an array of different sized patches and a full complement of seral or successional stages.
- After regeneration harvests, a more homogenized forest structure emerged, in which treatments drove stand structure and composition toward a single cohort and commercially desired species. After selection cutting, stand structure moved toward multiple cohorts because of continuous regeneration and release (some stands were entered several times), and species composition moved toward domination by shade-tolerant species in dense multilayered arrangements.
- A simplified vegetation mosaic was created by removal of large, fire-tolerant trees over repeated harvest entries, which typically left many smaller and fewer large trees. The result was ever increasing density and a surplus (in comparison with the native disturbance regimes) of young and mid-aged forests.
- A shift from fire-tolerant to intolerant tree species composition (timber harvest removed large ponderosa pine, Douglas-fir, western larch, western white and sugar pine [*Pinus lambertiana* Dougl.], and regenerated grand fir, white fir, Douglas-fir, and subalpine fir dominated stands).
- An increased vulnerability to large and severe fires, insect outbreaks, and disease pandemics.
- Fewer grasslands and shrublands.

Management activities have also had a profound effect on the spread of nonnative species, especially in the dry rangeland and forest ecosystems. As the climate continues to change, conditions for spread of some nonnative invasive (e.g., cheatgrass [*Bromus tectorum*]) and native invasive (e.g., barred owl) species will improve. Despite substantial efforts to control invasive species in the United States, the threat will remain high in coming years unless significant steps are taken to slow the advance of invasive plants and animals.

Timber Harvest and Associated Activities

Timber harvest and road development over the last 100 years have had a significant impact on the landscape patterns of MMC forests in eastern Oregon and Washington (fig. 37). Harvest activity was quite intense for nearly five decades beginning at the start of World War II. Clearcutting, selective harvesting, and subsequent planting for reforestation created substantial areas of structurally simple early- to mid-aged stands. Present-day stand structure of these previously harvested areas is

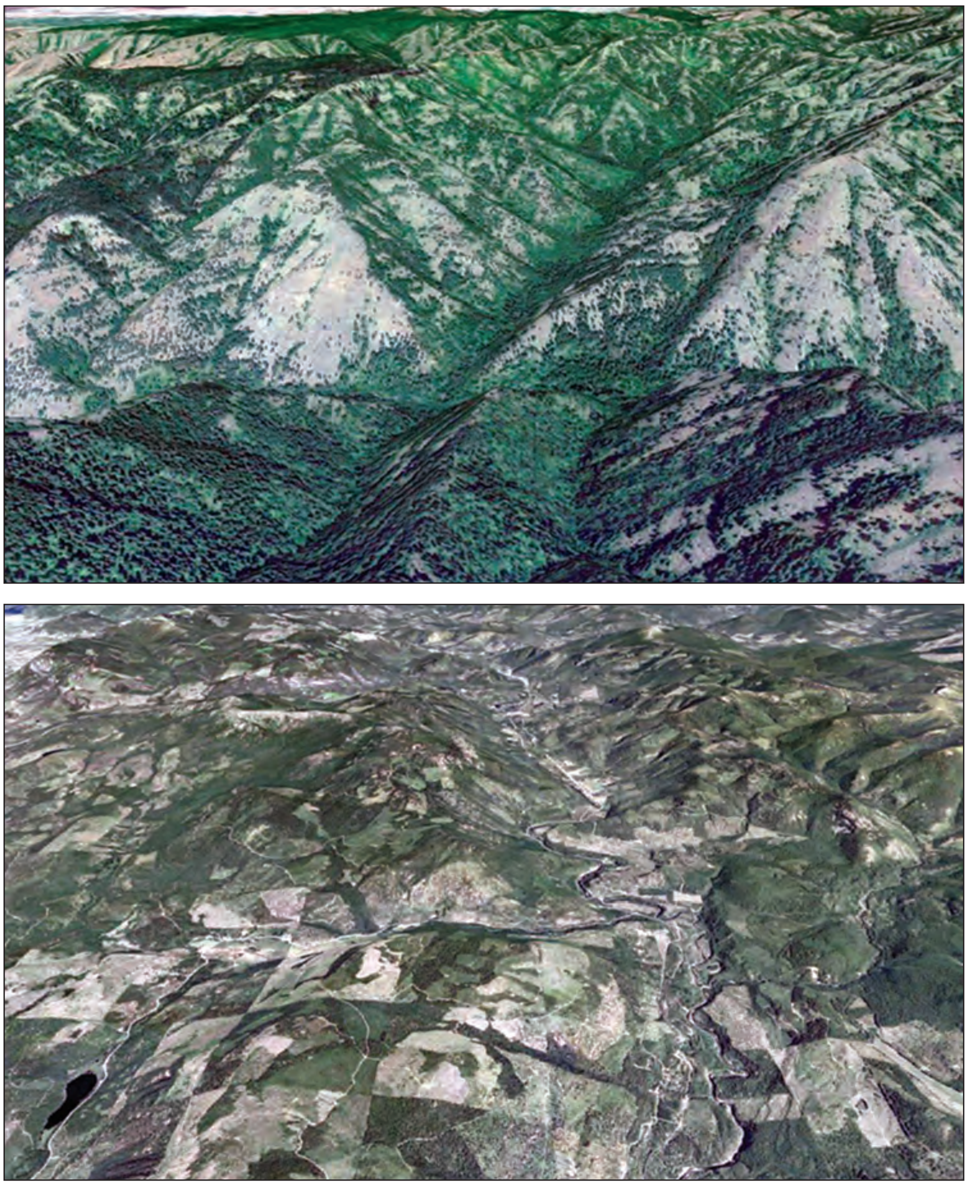

Figure 37—Timber harvest and road development over the last 100 years have had an impact on the spatial configuration of mixed-conifer forests in eastern Oregon and Washington. The upper image shows a portion of the Umatilla National Forest without stand management, fragmentation with land ownership, and roads. The lower image shows the effects of land ownership, roads, and harvest units on forest pattern. Images are from Google Earth™.

predominantly even-aged where regeneration harvesting was practiced and uneven-aged where selection cutting was more common, with tree species and genetic compositions that are often not particularly well-adapted to the environment. Large and old live and dead trees (snags) are conspicuously absent from many forests (Churchill et al. 2013; Franklin and Johnson 2012; Hessburg et al. 2003, 2005; Larsen and Churchill 2008, 2012; Merschel 2012).

Historical replanting at many sites used maladapted seedling stock from distant seed sources, and frequently with species compositions that were not ecologically appropriate for the site conditions (e.g., overly high proportions of economically important ponderosa pine and Douglas-fir). Seed collection zones were not yet established and there was only rudimentary understanding of the relationships linking seed provenance adaptation to particular site conditions. Prior to replanting, broadcast burning treatments were often employed. If omitted, significant surface fuel accumulations were often left behind (Agee 1998, Hann et al. 1997, Hessburg et al. 1999a, Huff et al. 1995). These and others factors have limited the function of some present-day MMC forests and the ecosystem services that can be obtained from them. Some of the post-harvest stands across this region contain large (more than about 53 cm [21 in] d.b.h.) young trees with species compositions and stand structures that may succumb to bark beetles or fire, and may benefit directly from manipulation of tree species composition and density.

Since the early 1990s, when certain management constraints (e.g., the “eastside screens”) were adopted, timber harvest has been scaled back by about 90 percent. Currently, the scale and scope of active management focuses on a limited number of cautiously selected locations, primarily for the purposes of restoring ecological processes and reducing fuel loads.

Fire Exclusion and Natural Fire Dynamics

In the absence of fire, forest density increased, the role of shade-tolerant tree species expanded, and fuel loads have increased.

Fire frequency in the region declined because of land-use changes in limited areas as early as 1880 or 1890. By 1900, fire frequency in dry forest types was in sharp decline (Everett et al. 2000; Haugo et al. 2010; Hessburg et al. 1999a, 1999c, 2000a, 2005, 2007; Hessl et al. 2004; Heyerdahl et al. 2001, 2008; Wright and Agee 2004), resulting in further changes to forest structure and composition. In the absence of fire, forest density increased, the role of shade-tolerant tree species expanded, and fuel loads have increased (Baker 2012; Everett et al. 1994; Hessburg et al. 2000b, 2003, 2005; Huff et al. 1995; Lehmkuhl et al. 1994; Merschel 2012; Perry et al. 2011). Not coincidentally, human population densities have increased rapidly as many people prefer to live at the lower forest ecotone. The amount of area classified as wildland-urban interface (WUI) has increased dramatically in recent decades (Gude et al. 2008, Radeloff et al. 2005).

Departure from historical fire regimes—

Land use activities and fire-suppression efforts from the early 20th to early 21st centuries have been effective at controlling small fires, eliminating the landscape patchiness that was historically important for interrupting fire flow at large spatial scales (Moritz et al. 2011, Perry et al. 2011). Large wildfires today typically result from rare or extreme weather or climatic events (Littell et al. 2009, Moritz et al. 2011). That was also the case historically. The significant difference associated with 20th century vegetation and climate change is the increase in frequency of very large (>10,000 ha) wildfires in many ecoregions of the West as a result of both more flammable landscapes and changing weather conditions. Owing to their increasing frequency and size, some large wildfires may synchronize large areas of the mixed-conifer landscape, setting the stage for more large and severe fires in the future. This happens because large fires, especially those dominated by crown-fire effects, can simplify and homogenize landscape patterns of species composition, tree size, age, density, and layering.

However, this is not always the case. A recent study in California found that some wildfires can achieve ecological and management goals by reducing landscape-scale fire hazard and increasing forest heterogeneity (Miller et al. 2012). This is likely to be true when the pattern and dispersion of fire severity and fire event patches are more or less consistent with the native fire and climatic regimes. We note here that despite referenced changes in fire regimes, many contemporary fires are still substantially dominated by mixed-severity fire, and the building blocks for landscape restoration are readily apparent. One can still observe the interactions of bottom-up, local (e.g., stand and landscape structure, topography), and top-down spatial controls on fire extent and fire severity (Perry et al. 2011).

Grazing

Sheep and cattle entered the region with the first settlers, but the earliest of them focused on agriculture, and their herds were typically small, including just enough livestock to work and support their family farms. Cattlemen initially believed that their stock could survive the harsh winters, and by 1860 there were at least 200,000 head of cattle in the region. A severe winter in 1861–1862 killed many cattle (Galbraith and Anderson 1991). Severe winters occurred again in 1880–1881 and in 1889–1890, after which most cattlemen recognized that shelter and feed were required for a sustainable operation.

By the late 1880s, severe grazing by cattle left the rangelands stressed (overgrazed). Large cattle herds also grazed adjacent dry forests and nearby grassy riparian zones. Because cattle require a lot of water, they preferred to graze the riparian zones, creek bottoms, and wet meadows that supported lush grasses through the dry summers, and provided a ready supply of water. Riparian zones constitute 1 to 4 percent of the land area of eastern Oregon and Washington national forests (Kauffman 1988), yet supply more than 80 percent of the grasses and herbs consumed by livestock (Roath and Krueger 1982).

Introductions of sheep, which require less water and are capable of using varying rangeland conditions, boomed in the mid- to late 1880s coincident with the decline in cattle. Eventually sheep numbers outstripped cattle, and violent conflicts often arose between Basque and Mormon sheepherders and resident cattlemen. The battles were the fiercest in Crook, Lake, Wheeler, and Deschutes Counties in south-central Oregon, where 8,000 to 10,000 sheep were killed per year for several years (Galbraith and Anderson 1991). Extensive grazing by sheep left native bunchgrasses and forbs in worse condition than was caused by cattle grazing. By the late 19th century, numerous exotic plants such as the bull and Canada thistles (*Cirsium vulgare* and *C. arvense*), cheatgrass, Dalmation and yellow toadflax (*Linaria dalmatica* and *L. vulgaris*), diffuse and spotted knapweeds (*Centaurea diffusa* and *C. maculosa*), and leafy spurge (*Euphorbia esula*) had become established in the region (Hann et al. 1997; Langston 1995; Wissmar et al. 1994a, 1994b).

Grazing permits and fees were required on national forest lands beginning in 1906, although grazing intensity increased until the 1920s (Wilkinson 1992). Congress approved the Taylor Grazing Act of 1934, which regulated grazing on the public domain (later, Bureau of Land Management lands) through the use of permits; subsequently, cattle and sheep numbers declined. Recent assessments of grassland condition suggest that grasslands are slowly recovering from some of the impacts of historical grazing, and current conditions are perhaps the best they have been in 100 years (e.g., see Harvey et al. 1994, Johnson et al. 1994, Skovlin and Thomas 1995). However, because of countless nonnative plant species introductions, some changes in native plant community structure and productivity are likely permanent (Lehmkuhl et al. 2013). There is ongoing public debate over whether livestock grazing on public lands should be continued under the Taylor Grazing Act (e.g., see Belsky et al. 1999, Beschta et al. 2012). Although public land managers have made steady progress in reducing cattle impacts to riparian vegetation and sediment load in small streams and creek bottoms, this is likely an area for continued improvement.

Land Development

The national forests of eastern Oregon and Washington are largely surrounded by private lands. Throughout most of the 1900s, these lands were used for natural resource extraction or agriculture. In recent decades, rural home development has expanded. The rapid increase of these exurban homes adjacent to wildlands poses a major threat to ecological functioning, fire management, and native biodiversity. Exurban development can fragment habitats, create barriers to animal movement and other ecological processes, alter natural disturbance, extirpate top predators, increase weeds, increase mesocarnivores (medium-sized predators) and diseases, and ultimately, cause local extirpation of some native species (Hansen et al. 2005, McKinney 2002, Pickett et al. 2001). The presence of homes in the WUI also strongly affects fire management options in adjacent national forests, increasing the need to suppress fire to protect property and lives, requiring identification and mitigation of hazard trees, and increasing firefighting costs.

Theobald et al. (2011) evaluated changes in abundance and connectivity of "high-quality" forest patches resulting from land use changes in the Western United States. They estimated that land uses associated with residential development, roads, and highways have caused roughly a 4.5-percent loss in area (20 000 km^2 [7,722 mi^2]) of large, unmanaged forested patches, and continued expansion of residential land will likely reduce forested area by another 1.2 percent by 2030. Projected losses for 2000–2030 were particularly high in Oregon and Washington (fig. 38). When considering both patch size and overall landscape connectivity for forests across the West in 2000, the most important forested areas were found in the Cascades Ecoregion and the Canadian/Middle Rockies Ecoregion, which includes the Okanogan Highlands and the Blue Mountains (Theobald et al. 2011). This

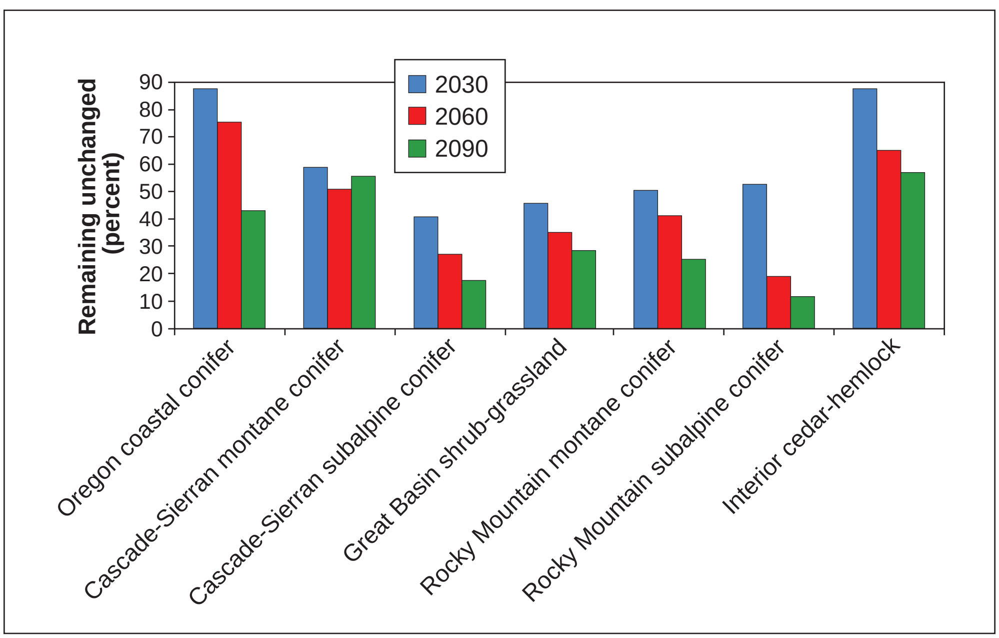

Figure 38—For biomes in Oregon and Washington, the model results among scenarios indicate that Rocky Mountain subalpine conifer in the Blue Mountains and Cascade Range will undergo the greatest loss in area. Data from Rehfeldt et al. (2012).

analysis illustrates the high contribution to subcontinental forest connectivity made by forests in Oregon and Washington, and the vulnerability of this connectivity to future land use changes.

Climate Change

Climate change is affecting MMC forests in eastern Oregon and Washington through changes that include warmer temperatures (Mote and Salathé 2010) and snowpack declines (fig. 39; Mote 2003, Mote et al. 2005). Over the next 100 years, because CO_2 concentrations are projected to rise, these impacts will likely increase in magnitude and extent over the region (Littell et al. 2010). Impacts to terrestrial ecosystems include increased fire frequency and severity, increased susceptibility to some insects and diseases (Fettig et al. 2013, Preisler et al. 2012), and increases in the presence of some invasive species (CIG 2009). Regionally, increased summer temperature and decreased summer precipitation and snowpack (Cayan et al. 2001) are projected to result in a doubling of the area burned by fire by the 2040s and a tripling by the 2080s (CIG 2009). Changes in the length and timing of seasons, especially the growing season, the timing of bud break (phenology), and the seasonal availability of soil moisture are expected to produce large positive and negative shifts in forest growth, with a net effect of increased forest mortality in the eastern Cascades (Choat et al. 2012, Grant et al. 2013, Williams et al. 2012).

In addition to general regional changes in temperature and precipitation predicted by models is the likelihood that change at the local scale could be quite variable owing to high variability in physiographic environments where moist mixed-conifer forests are found.

In addition to the general regional changes in temperature and precipitation predicted by various regional models there is the likelihood that change at the local scale could be quite variable owing to high variability in physiographic environments where MMC forests are found (Daly et al. 2007). While most areas may become warmer, some may become moister and others drier, particularly during certain seasons of the year. Canyon bottoms may remain cool due to accentuated cold air drainage.

For wildlife, changes in climate and vegetation will impact habitat characteristics, reproductive success, and food and water availability. These changes will alter species assemblages and distributions, migration routes, interactions with competitors and predators, and may impact population viability for many species (Staudinger et al. 2013). Impacts may be particularly severe for currently rare or endemic species with restricted distributions (Parmesan 2006). Anticipated changes to wildlife include (1) the susceptibility of high-elevation habitats and species dependent on snowpack (e.g., wolverine [*Gulo gulo*] for the Blue Mountains), (2) impacts on wetlands and associated species, especially those sensitive to water temperature (e.g., tailed frog [*Ascaphus montanus*]), and (3) phenological mismatches for migratory birds and other species (Parmesan 2006). Changes are expected to happen more quickly than species' potential to adapt (Staudinger et al. 2013).

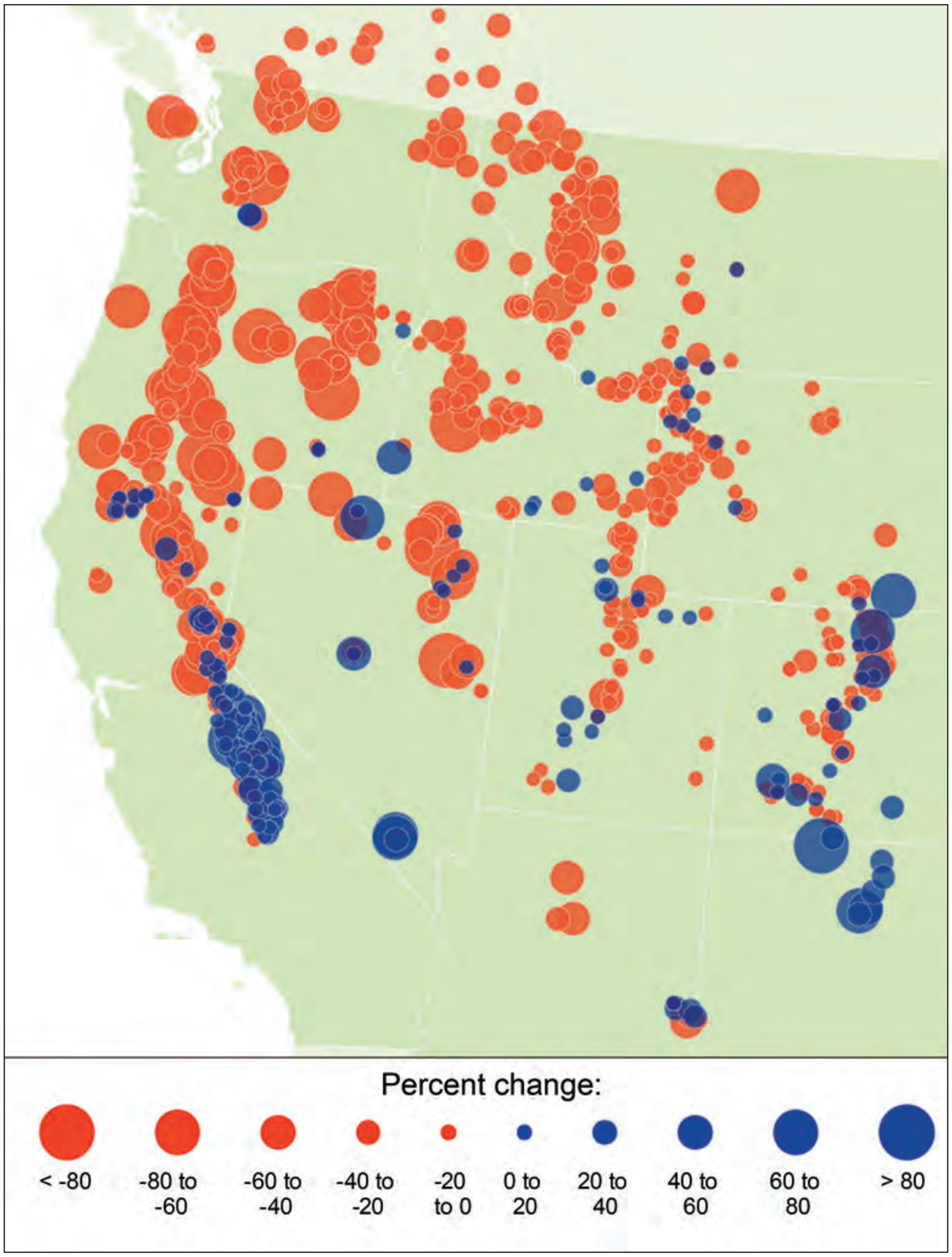

Figure 39—Historical snowpack decline in the Pacific Northwest region 1950–2000 (Mote 2009).

Maintaining landscape patterns that allow species to move in response to changes in climatic and habitat conditions may be particularly important for conservation of sensitive species (Heller and Zavaleta 2009, Nuñez et al. 2013).

Change in climate and disturbance: 1950 to present—

El Niño/Southern Oscillation (ENSO) remains the most important coupled ocean-atmosphere phenomenon to cause climate variability on seasonal to interannual time scales in the Pacific Northwest. Across decadal timescales, longer-term oscillations such as the Pacific Decadal Oscillation (PDO) come into play. The PDO was in an extended cool phase from about 1945 to 1976 and in a warm phase from 1977 to 1998 (Mantua and Hare 2002, Mantua et al. 1997, Mote et al. 2003, Waring et al. 2011). Minimum temperatures have increased most rapidly in the semi-arid Columbia River basin and the Great Basin, and in semi-arid portions of the Okanogan Highlands, but have changed relatively little west of the Cascade Crest and in the Blue Mountains (fig. 40). Mean average temperature has been observed to have increased by 0.8 °C (1.50 °F) since 1900 (Mote and Salathé 2010). Climate

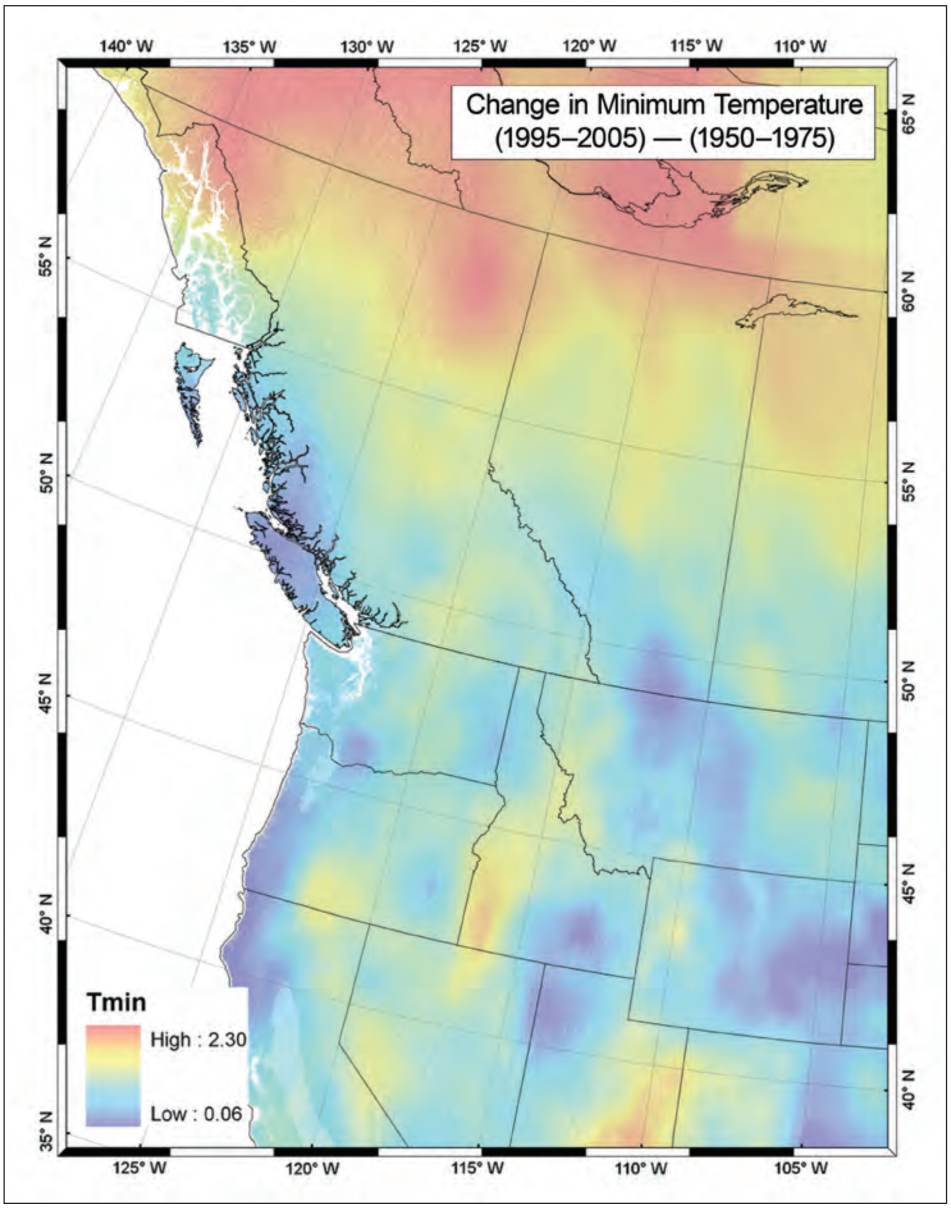

Figure 40—Changes in minimum temperature throughout the Pacific Northwest. Source: Waring et al. 2011.

forecasting models, when averaged, project increases in annual temperature of 1.1 °C (2.0 °F) by the 2020s, 1.8 °C (3.2 °F) by the 2040s, and 3.0 °C (5.3 °F) by the 2080s, compared with the average temperature from 1970 to 1999.

Trends in historical and projected future changes in precipitation in the Pacific Northwest are less clear than for temperature (Stephens et al. 2010). For example, precipitation in the Pacific Northwest has increased by 13 to 38 percent since 1900, but has shown substantial interannual and interdecadal variability during the 20th century (Mote et al. 2003), which current climate models are unable to simulate under future warming scenarios (Ault et al. 2012). Some, but not all models predict slight future increases in annual precipitation (1 to 2 percent in 2030 to 2059, and 2 to 4 percent in 2070 to 2099; Littell et al. 2009).

The ecoregions of eastern Oregon and Washington (notably the eastern Cascades and the Blue Mountains) have a Mediterranean type of climate characterized by warm dry summers and cool moist winters. The lack of summer precipitation coupled with earlier springs and later winters suggest increased vulnerability of forests and forest associates to climate change. Anticipated changes in fire regimes in the Northwest United States is a direct result of these changes in climate (Westerling et al. 2006). The following summarizes the key climate change projections and impacts to moist mixed conifer forests in the Pacific Northwest region (CIG 2009, IPCC 2013, Mote et al. 2005).

Projected outcomes:

- Reduced snowpack amounts (up to 50 percent projected decline)
- Increased rain on snow
- Warmer air temperatures (up to 3 °C by 2080)
- Change in the timing of snowmelt (earlier)
- More climate extremes

Projected impacts to moist mixed-conifer forests:

- Reduced soil moisture
- Increased drought stress
    - Increased fire intensity, severity, and frequency
    - Increased susceptibility to insect attack
- Phenologic shifts (earlier onset of budbreak, longer growing season)

Climate change impacts on forest ecosystems—
Climate change influences on forest tree species are a function of the ecophysiological tolerances unique to each species. Waring et al. (2011) used a process-based forest model to estimate changes in tree species competitiveness (influenced by a variety of ecological factors) between 1950–1975 and 1995–2005 based on climate

effects on simulated leaf area index (LAI). Although LAI may not be the prime indicator of a conifer tree's capacity to respond to changing climates, it provides an index of potential response. Across their western North America study area, they found a significant decrease in the competitiveness of over 50 percent of the evergreen species in ecoregions in the northern and southern portions of the study area. Within Oregon and Washington, competitiveness changed little in the West Cascades and was moderately reduced in the East Cascades, the Okanogan Highlands, and the Blue Mountains (fig. 41). Furthermore, regions exhibiting reduced competitiveness under climate change scenarios were predicted to experience higher rates of disturbance by insects, fire, and other factors (Waring et al. 2011).

These changes in climate are also projected to influence the area of suitable climate for vegetation types differently and shift vegetation assemblages over the landscape. Rehfeldt et al. (2012) developed climate envelope models for biomes of North America and projected change in area and distribution under six climate change scenarios. However, there is still debate about whether this approach realistically captures physiological tolerances of forest species (Loehle 2011).

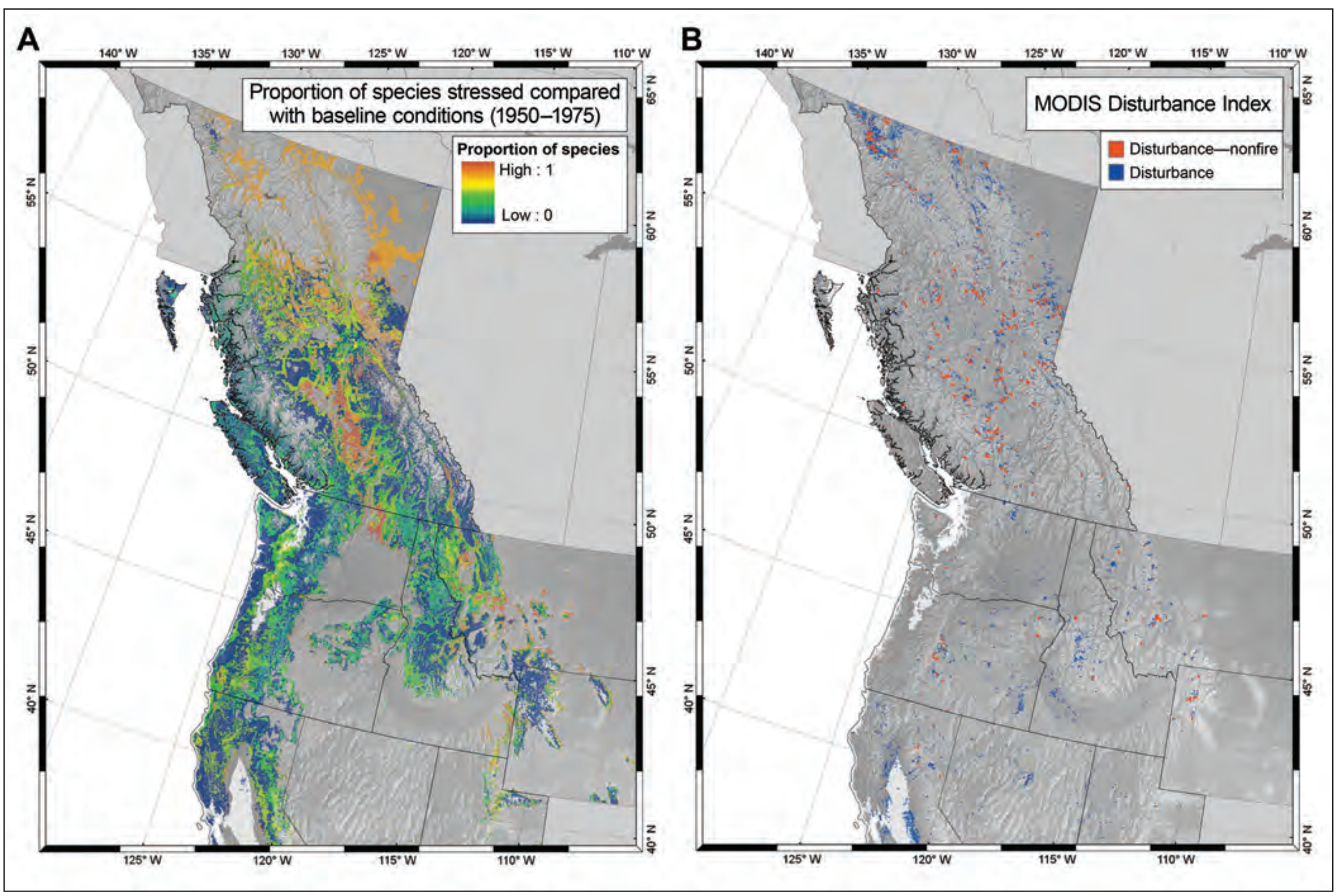

Figure 41—Map presenting the proportion of 15 coniferous species predicted as no longer well-adapted in the years 1995–2005 compared with baseline conditions (1950–1975). From Waring et al. (2011).

2

We acknowledge that limitations to the use of climate envelope approaches for modeling potential species distribution changes are well known (Fettig et al. 2013). Nonetheless, modeling approaches provide some insight into the possible magnitude and geography of projected changes in species distribution in the coming decades.

For biomes in Oregon and Washington, model results among scenarios estimate that all Oregon Coastal Conifer and Interior Cedar-Hemlock types will undergo the least reduction in the areas they currently occupy. Subalpine biomes in the Cascades and the Rockies are projected to eventually undergo substantial reduction in currently occupied areas, with only 27 percent and 19 percent of these biome types remaining. Drier conifer forest area is also forecasted to decline substantially: 41 percent of Rocky Mountain Montane Conifer, which currently occurs in the Okanogan Highlands and the Blue Mountains, is forecasted to remain in 2060. Importantly, by 2090, only 12 to 57 percent of the current area is forecasted to have climates suitable to the biomes occurring there today, although the rate of decline is uncertain. Of course we do not know how extensive the geographic shift in biomes will eventually be, because there are many complex ecological interactions that will play out over time, and ecophysiological modeling is in its infancy. Current models give us an indication of how dramatic the changes could be. Nonetheless, some areas currently occupied by subalpine forest in the Cascades, for example, will likely become suitable as mixed-conifer forests now found at mid elevations. Climate in the Northeast Cascades of Washington becomes suitable for the Coastal Hemlock Biome type. Forest areas in the Okanogan Highlands and Blue Mountains are largely replaced by climates suitable for the Great Basin Shrub-Grassland type.

Forest productivity is also projected to change under future climates. Productivity is forecasted to increase in what are currently subalpine and alpine zones across the region and to decline in the drier forests, which are largely at the lower forest ecotone in the East Cascades, Okanogan Highlands, and Blue Mountains (Latta et al. 2010). Concomitantly with warmer temperatures, forests may respond to increased atmospheric CO_2 concentrations through increased water use efficiency. However, forest growth response to rising CO_2 is less clear than it appears it will be for temperature, perhaps because soil moisture availability is expected to decline (Peñuelas et al. 2011). Warming may allow some insect species (e.g., some bark beetles) to complete extra generations per year, and adult emergence and flight activity could occur earlier and last longer. Cold-induced mortality of insects during winter may also decrease. For example, model simulations indicate that the climatic suitability for mountain pine beetle will increase during the next several decades in eastern Oregon and Washington (Bentz et al. 2010, Preisler et al. 2012), and when combined with current forest conditions (Fettig et al. 2007, Hicke and Jenkins 2008), higher levels of tree mortality are expected.

Conversely, there could be some reductions in insect-induced tree mortality when warming increases susceptibility to predators, parasitoids, or pathogens. Preliminary evidence suggests that warmer conditions might shorten the incubation period (time between infection and mortality) of the baculovirus that limits Douglas-fir tussock moth populations (see footnote 2). Given projected warming in winter and decreases in precipitation in summer, the impact of western spruce budworm will likely decrease in the future relative to the current distributions of susceptible forest types in this region (Williams and Liebhold 1995). The insect hibernates in winter, and elevated temperatures during this period reduce survival. Warming may also change the timing and synchrony of bud break in Douglas-fir, which in turn may influence outbreak frequency and severity. Projected climate change impacts to MMC forests in the region are likely to be manifest through ENSO and other oscillations such as the PDO. Moreover, the multiscale (broad-scale and fine topographic) nature by which climate change is likely to manifest across the landscape necessitates a multiscale hierarchy such as the one we have outlined in previous sections of this synthesis (CIG 2009, IPCC 2013, Mote et al. 2005):

- El Niño/Southern Oscillation remains the most important coupled ocean-atmosphere phenomenon to cause climate variability on seasonal to interannual time scales in the Pacific Northwest.
- Projected changes in climate will be manifest at both highly localized and broad regional scales for moist mixed-conifer forests.
- Necessitating the need for multiscale, hierarchical approach to address future climate change impacts.

Changes in Vegetation

Changes in the structure and composition of forests since Euro-American settlement are well documented (Hessburg and Agee 2003). For example, forests are now typically several times denser in most locations than under native fire regimes (Camp 1999, Hagmann et al. 2013, Merschel 2012, Perry et al. 2004) (table 5). However, one recent study (Baker 2012) has indicated that these forests were “generally dense” (100 trees per acre [275 trees per hectare]) in the late 1800s and suggested that the amount of change in the structure of these forests and the need for density reduction has been overstated. It is unclear how density estimates from the Baker study apply to MMC forests, which are a subset of the ponderosa, dry mixed-conifer, and lodgepole pine stands that he sampled, as less than 1 percent of the trees were identified as white or grand fir. This suggests that either his records

Table 5—Historical and current tree density (trees per hectare, TPH) estimates in dry and moist mixed-conifer forests

Region	Forest description	Historical density	Current density	Source	Notes
		Mean ± standard deviation (range) and time period			
Eastern slope of the Cascade Range, southern Oregon	Mixed-dry	64 ± 22 (14–156) 1914–1922	286 ± 130 (109–572) 1997–2006	Hagmann et al. 2013	Historical density is reported for trees >15.2 cm diameter at breast height (d.b.h.) based on timber inventory transects that covered 10 to 20 percent (6646 ha) of a 38 641 ha study area of ponderosa pine and dry and moist mixed-conifer forest. Current density is reported for trees >15 cm d.b.h. and is based on 24 current vegetation survey plots within the sample area. Inventory plots were assigned to habitat types using mapped potential vegetation types (ILAP).
	Mixed-moist	78 ± 37 (0–296) 1914–1922	233 ± 90 (104–378) 1997–2006		
Warm Springs Indian Reservation, eastern slope of the Cascade Range, Oregon	Mixed-dry	65 ± 25 (6–188) 1922–1925	NA	Hagmann, unpublished data, 2013[a]	Historical density is reported for trees >15.2 cm d.b.h. based on timber inventory transects that covered 20 percent (32 293 ha) of a 158 354 ha study area of dry and moist mixed-conifer forest. Inventory plots were assigned to habitat types using mapped potential vegetation types (ILAP).
	Mixed-moist	70 ± 32 (0–289) 1922–1925	NA		
Austin-Whitney Tract in the Blue Mountains of northeast Oregon	Mixed-dry	120 1910	NA	Munger 1912 (summarized by David Powell)	Density is reported for trees >10 cm d.b.h. Munger specifically inventoried "well stocked" stands.
Crawford Creek Tract in the Blue Mountains near Unity, Oregon	Mixed-dry	126 1912	NA		
Lookinglass Creek Tract in the Wallowa Mountains of northeast Oregon	Mixed-moist	162 1910	NA		
Eastern slope of the Cascade Range, central Oregon	Mixed-dry	136 1912	NA	Munger 1917	Density is reported for trees >10 cm d.b.h. Munger specifically inventoried "well stocked" stands.
Eastern slope of the Cascade Range, central Oregon	Mixed-conifer	275 Late 1800s	NA	Baker 2012	General land office survey records were used to estimate stand density. Estimates for a 398 346 ha area are based on 11 856 trees marking section and quarter-section corners. Four trees were used to estimate density at section corners, and two were used at quarter corners. This inventory area included lodgepole pine forests, which can have high stand densities.

Table 5—Historical and current tree density (trees per hectare, TPH) estimates in dry and moist mixed-conifer forests (continued)

Region	Forest description	Historical density	Current density	Source	Notes
		Mean ± standard deviation (range) and time period			
Eastern slope of the Cascade Range, central Oregon	Mixed-dry (Douglas-fir)	447 ± 186.8 (227–778) 14 sample sites 2009–2010	NA	Merschel et al. [N.d.]	Density is reported for trees >10 cm d.b.h. Sample sites (1.0 ha) were widely distributed in mature forest mapped in any of the mixed-conifer dry or mixed-conifer dry plant association groups. The number of sample sites used in each estimate is reported following the reported density.
	Mixed-dry (grand fir)	NA	536 ± 185 (113–825) 16 sample sites 2009–2010		
	Mixed-moist	NA	429 ± 200 (52–900) 24 sample sites 2009–2010		
Ochoco Mountains, central Oregon	Mixed-dry (Douglas-fir)	NA	274 ± 132 (95–533) 26 sample sites 2009–2010		
	Mixed-dry (grand fir)	NA	203 ± 102 (76–430) 19 sample sites 2009–2010		
	Mixed-moist	NA	312 ± 211 (67–799) 26 sample sites 2009–2010		
Eastern slope of the Cascade Range, central Oregon	Dry and moist mixed-conifer	40–80 in 8 stands,<30 in 5 stands. Refers to trees >150 years old.	868 (335–1415) ~2004	Perry et al. 2004	Age and size were sampled for live trees at thirteen 0.2-ha sites. Historical density refers to the number of trees per hectare >150 years in age. Current density refers to all trees >5 cm d.b.h.
Eastern slope of the Cascade Range, northern Washington	Mixed-conifer-dry	HDSG[b] —119 WDSH—168 WDTS—315 CDG—311 WMSH—113 1899	HDSG—218 WDSH—326 WDTS—293 CDG—376 WMSH—556 1999	Ohlson and Schellhaas 1999	Historical density was estimated using 52 one-tenth-acre plots. Authors suggest that changes in fire regime and associated changes in structure began in the 1860s and that maximum density was reached in the mid-20th century. Historical reconstructed densities and current densities are reported for all trees >12.7 cm d.b.h., and are stratified by different Douglas-fir potential vegetation types ranked from most xeric to most mesic. WMSH would correspond to moist mixed-conifer potential vegetation types. It had a fire return interval of <19 years. The relatively high historical density in WDTS, a type related to a mixed severity fire that occurred in 1886.

Table 5—Historical and current tree density (trees per hectare, TPH) estimates in dry and moist mixed-conifer forests (continued)

Region	Forest description	Historical density	Current density	Source	Notes
		Mean ± standard deviation (range) and time period			
Eastern slope of the Cascades Range and foothills, Oregon and Washington	Mixed-dry	NA	470 ± 267 (n = 512) 1994–2004	M. Reilly, unpublished data[c]	Density is reported for trees >10 cm d.b.h. Density is based on forest current vegetation survey (CVS) plots distributed on a 2.5 by 2.5 km grid (5 km in wilderness areas) throughout national forests of Oregon and Washington. Sample sites include developmental stages ranging from early seral in recently disturbed areas (logging, wildfire, etc.) to mature old-growth forest. The number of CVS plots used to calculate density is reported in parentheses. CVS plots were assigned to PVTs based on ILAP map.

Note: Dry mixed-conifer forests are included because several studies indicate that historical and current disturbance regimes and stand densities of the two sites overlap to a large degree (see table 2). Diameter thresholds, methods of estimation, and spatial coverage used to develop estimates varies among authors.

[a] On file with: Keala Hagmann, School of Environmental and Forest Sciences, University of Washington, Box 352100, Seattle, WA 98195.

[b] Plant association groups with respective plant associations:

HDSG (Hot/Dry/Shrub/Grass):
Pinus ponderosa/*Agropyron spicatum*
Pseudotsuga menziesii/*Agropyron spicatum*
Pinus ponderosa/*Purshia tridentata*/*Agropyron spicatum*
Pseudotsuga menziesii/*Purshia tridentata*/*Agropyron spicatum*
Pseudotsuga menziesii/*Symphoricarpos albus*/*Agropyron spicatum*

WDSH (Warm/Dry/Shrub/Herb):
Pseudotsuga menziesii/*Arctostaphylos uva-ursi*
Pseudotsuga menziesii/*Arctostaphylos uva-ursi*/*Purshia tridentata*
Pseudotsuga menziesii/*Spirea betulifolia* var. *lucida*

WDTS (Warm/Dry/Tall Shrub):
Pseudotsuga menziesii/*Physocarpus malvaceus*
Pseudotsuga menziesii/*Physocarpus malvaceus*/*Linaea borealis* var. *longiflora*

WMSH (Warm/Mesic/Shrub/Herb):
Pseudotsuga menziesii/*Symphoricarpos albus*
Pseudotsuga menziesii/*Symphoricarpos albus*/*Calamagrostis rubescens*

CDG (Cool/Dry/Grass)
Pseudotsuga menziesii/*Arctostaphylos uva-ursi*/*Calamagrostis rubescens*
Pseudotsuga menziesii/*Purshia tridentata*/*Calamagrostis rubescens*
Pseudotsuga menziesii/*Spirea betulifolia* var. *lucida*/*Calamagrostis rubescens*

[c] On file with: Matthew Reilly, Department of Forest Ecosystems and Society, Oregon State University, 321 Richardson Hall, Corvallis, OR 97331.

did not sample much MMC forest or that there were few individuals of these *Abies* species present in the late 1800s. Hagmann et al. (2013) examined stand inventory records from the 1920s in landscapes that overlap areas sampled by Baker (2012) and found much lower tree densities for the same areas. Munger (1917) reported a density of 55 large trees per acre for ponderosa pine forest, and 61 in dry mixed-conifer forests in the eastern Cascades in the early 20th century. That study characterized dry mixed-conifer stands as open with widely spaced trees, but noted that the forests consisted of open stands of mature trees interrupted by treeless areas and denser patches of young trees.

Although total stand densities have increased several fold, the densities of large trees (>50 cm [~20 in]) have declined in many areas within older mixed-conifer forest areas (excluding clearcut areas) by as much as 50 percent based on a few studies (table 3). The decline has been observed in both Oregon and Washington and is likely a result of selective logging of large pines and Douglas-firs during the 20th century. One study from Washington reported increases in the density of large trees.[3]

Heyerdahl showed that fire occurrence in several pine and mixed-conifer forest patches in central Oregon strongly declined after 1880, and grand fir establishment started to increase in the 1880s and 1890s (Heyerdahl et al. 2001 and unpublished data; Merschel 2012). If we assume that the presettlement density of trees in mixed-conifer patches was 50 to 100 trees per hectare (20 to 40 trees per acre), then current densities for mixed-conifer forests may be two to six times the historical densities of some patches. For example, several recent studies have estimated current mixed conifer densities at about 200 to more than 300 trees per hectare (80 to more than 120 trees per acre) (Camp 1999, Merschel 2012, Perry et al. 2004). Under the recent disturbance regime of fire suppression and logging of old trees, MMC forests often consist of scattered remnants of medium- to large-sized shade-intolerant dominant trees, with medium-sized shade-tolerant co-dominant trees, and often several dense layers of intermediate and overtopped or suppressed shade-tolerant trees (Oliver and Larsen 1996). Thus, land use and fire regime changes have shifted successional pathways from dynamic fine scale mosaics driven by low- to mixed-severity fire to coarser grained, more homogenous patch types driven by high-severity fire.

Seed availability, either the absence of seeds of a species or the presence of nearby seed sources (e.g., a kind of "mass effects" in which a high rate of

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[3] Ohlson, P.; Schellhaas, R. 1999. Historical and current stand structure in Douglas-fir and ponderosa pine forests. Unpublished report. Portland, OR: U.S. Department of Agriculture, Forest Service, Okanogan and Wenatchee National Forests.

propagule input maintains species on sites that are not well suited for reproduction or long-term survival) (Shmida and Wilson 1985), can have a strong influence on community development, and is often overlooked as a factor in succession. For example, fire suppression has led to an increase of grand or white fir in many forest environments, which has converted many acres of open ponderosa pine forests to dense mixed-conifer forests (Everett et al. 1994, 1997; Hann et al. 1997; Hessburg et al. 1999a, 2000a). In many cases, the true firs and Douglas-fir may be establishing on sites that are less optimal for their growth only because there are massive amounts of seeds in the local landscape (these species dominate the seed rain) that swamp out some of the environmental and ecological controls over species distributions within a community. The densification of these stands could inhibit regeneration of early-seral species such as ponderosa pine and western larch, which are an important component of the overstory, and require relatively open conditions and frequent disturbance for establishment. In many areas, high-grade logging over the 20th century has reduced the densities of these large young forest dominants (Harrod et al. 1999), and they will not return to these sites without disturbances that open the canopy, reduce the overall seed rain of the shade-tolerant species, and reduce stem densities.

Old-growth forests are one of the most ecologically and socially valuable successional stages in the Pacific Northwest (Spies and Duncan 2009), but they are also quite variable in structure and dynamics, especially between the west and east sides of the Cascade Range. The classic old-growth forests of the coastal portion of the Pacific Northwest—fire-infrequent forests that contain large, emergent, long-lived conifers (e.g., Douglas-fir) and dense multilayered mid- and understories of shade-tolerant trees (e.g., western hemlock)—have analogs on the east side of the Cascades. Fire-infrequent old growth on the east side establishes through different pathways and exists under differing conditions. Much of what developed prior to the onset of the timber utilization era, beginning in the later part of the 1800s, consisted of large old ponderosa pine and Douglas-fir in the upper canopy, with occasional western larch and shade-tolerant white/grand fir in the upper canopy and white/grand fir filling lower canopy layers (Merschel 2012). However, this old-forest type would have been uncommon in the drier, fire-prone landscapes, occupying moist, less fire-prone sites. For example, Agee (2003) estimated that only 10 to 17 percent of MMC forest sites would have supported dense old forests under the native fire regime, although some individual watersheds may have had as much as 20 to 35 percent (Hessburg et al. 1999a, 2000a).

The composition of patches of old-growth trees on the east side also depended very much on topographic position. Solar insolation and soil moisture retention,

The composition of patches of old-growth trees on the east side depends very much on topographic position. Solar insolation and soil moisture retention, key factors in determining growing potential and relative drought stress, are considerably different depending on slope aspect and slope position.

key factors in determining growing potential and relative drought stress, are considerably different depending on slope aspect and slope position. Thus the species composition and structure of patches of old forest are quite different at the bottom of a drainage, where soil moisture and shade enable the growth of mixed species of denser, multilayered patches compared with patches on a south-facing midslope, which tend to be single layers of more sparse groups of predominantly large pines.

Another important difference between older, structurally diverse, interior MMC forest stands is that they do not persist for long periods as westside old growth does. It is important to continually recruit forest stands into this older, structurally diverse condition because forests on the east side do not stay in that condition. The transience of old forest conditions in interior MMC forest is a consequence of the unique assemblage of primary disturbances (e.g., fire, insects, and diseases) found in these drier landscapes. The dense, multilayered old forests that have developed under extended periods of fire exclusion are vulnerable to a host of disturbance factors and are unlikely to persist except where local topography, soils, and microclimate are suitable (e.g., sufficient soil moisture). By contrast, the sparser, single-layered old forests, dominated by large pines, are capable of persisting for as long as 500 years.

Historically, the most common expression of old growth would have been a "fire climax" (Perry et al. 2008) or old single-story, park-like forest (sensu O'Hara 1996), where various combinations of large, old, fire-resistant ponderosa pine, Douglas-fir, and western larch would occur in the canopy over a relatively open but variable and ephemeral understory of pines or shade-tolerant tree species depending on the fire frequency at the site. On sites with longer fire return intervals (>25 years) dominated by mixed-severity fire regimes, the canopies of the fire-dependent old growth would include patches (fractions of an acre to several acres) formed by small mixed-severity disturbances from fire or insects and disease. In many landscapes, streamside environments would have longer fire return intervals (>50 years), and support old growth stands with larger components of fir in the understories and sometimes in the overstories.

Young forest communities may have been altered as well, but we have less information about the amounts and patterns of open plant communities maintained by high-severity fire, which was a variable component within the mixed-severity regime of the mixed-conifer forest. Of special interest, clonal aspen patches, often imbedded within mixed-conifer forests, have been heavily affected by fire exclusion owing to their relatively short life expectancy (stands begin to deteriorate at 55 to 60 years of age and are pathologically old and decadent at 90 to 110 years; Bartos

2000). Wildfires normally revitalized aspen clones, with some patches sprouting 10,000 to 20,000 stems per hectare (24,710 to 49,421 stems per acre) in early life stages after fires. Ungulate browsing, both wild and domestic, and a host of stem cankers, foliar diseases, and insect defoliators naturally thin aspen clones to a few hundred stems per hectare after several decades.

Changes in Disturbance Factors: Insects and Disease

The frequency, severity, and scale of insect outbreaks in MMC forests of eastern Oregon and Washington vary considerably (table 5), even by species. In short, effects on vegetation range from short-term reductions in crown cover (e.g., larch casebearer), to modest increases in background levels of tree mortality (e.g., balsam wooly adelgid [*Adelges piceae*]), to regional-scale outbreaks resulting in extensive amounts of tree mortality under some circumstances (e.g., Douglas-fir tussock moth and western spruce budworm). Of more recent concern is the impact of mountain pine beetle outbreaks in the region (see "Animals of the Moist Mixed-Conifer Ecosystem"). Wickman (1992) noted that historically the impacts of insects on mixed-conifer forests were typically of shorter duration and lower severity than was observed in recent decades. A heterogeneous landscape is thought to be more resilient to insect- and disease-caused disturbances (Fettig et al. 2007, Filip et al. 2010). Although it is difficult to isolate the confounding effects of recent management practices and climatic changes, a more heterogeneous landscape likely ensured that most disturbances in MMC forests were more brief and spatially confined in the past (Hessburg et al. 1994). This has been well documented for dry mixed-conifer forests in the region, and it is likely a similar trend existed for MMC forests as they are interconnected with dry mixed-conifer forests that are now substantially modified by fire exclusion and selective harvesting (Hessburg et al. 1994, Lehmkuhl et al. 1994).

Outbreaks of forest diseases caused by native and introduced pathogens are generally thought to become more frequent and severe as a result of climate change (Sturrock et al. 2011). However, diseases caused by pathogens directly affected by climate (e.g., needle blights) are predicted to have a reduced impact under warmer and drier conditions. For thorough reviews on the expected effects of climatic change on forest pathogens, see Dale et al. (2001) and Sturrock et al. (2011).

Effects of Altered Disturbance Regimes, Land Use, and Climate Change on Animal Populations and Habitats

There have been significant changes to fish and wildlife populations associated with MMC forests during the last 150 years. The effects of human land uses and altered disturbance regimes discussed in previous sections of this report have had a significant impact on fish and wildlife populations across this entire area. Additionally, climate change has already produced observable changes in animal communities in the Pacific Northwest (Hixon et al. 2010), and those changes are likely to intensify in the future (Dalton et al. 2013).

Regional landscape assessments over the past 20 years have documented the profound effects of historical human use and forest management on mixed-conifer and other wildlife habitats of eastern Oregon and Washington (Everett et al. 1994; Hann et al. 1997; Hessburg et al. 1999a, 2000a; Lehmkuhl et al. 1994; USDA FS 1996). The area and condition of grasslands, shrublands, and old forest multistory and late-seral single-story forests, and the forest backbone of large residual trees, have declined markedly (Hessburg and Agee 2003, Hessburg et al. 2005) (fig. 42). At the same time, the area occupied by invasive species, roads, intermediate-aged forest, and insect, disease, and fire susceptibility (as indicated by changes in forest structure, fuel loading, and crown fire potential) have increased. Habitat trends reported in recent wildlife habitat assessments reflect a widespread reduction in habitat values relative to historical conditions for wildlife associated with early-seral, open forest, and postfire habitats (Gaines et al., in press; Suring et al. 2011; Wales et al. 2011; Wisdom et al. 2000). Old forest habitat conditions show less decline at a regional scale, but patterns are variable across sub-basins.

For example, Wisdom et al. (2000) found that habitat for species associated with low-elevation old forest, which includes single-story ponderosa pine forest and single- and multistory mixed-conifer forest, has declined in most of eastern Oregon and Washington (table 6). The habitat trend for species associated with old forest (i.e., forests with multistory stands and large old trees), which spans the elevation gradient from low to high, varies across the region from markedly declining in the northeastern Cascade Range, to neutral in the southeastern Cascades and Blue Mountains, to increasing in the Klamath province. The habitat trend for generalist forest mosaic species, which use a broad range of forest cover types and structural stages, has been neutral or positive. The habitat trend for young forest species varies from highly or moderately negative in the Klamath and Blue Mountain provinces, respectively, to positive to neutral in the Cascade Range province (Wisdom et al. 2000).

Habitat assessments in support of national forest plan revision activities in eastern Washington and Oregon have produced results similar to those of Wisdom et al. (2000). In their assessment of wildlife habitat and population viability in national forests of eastern Washington (the Okanogan-Wenatchee and Colville National Forests), Gaines et al. (in press) found that the species whose viability had declined most relative to historical conditions were those species sensitive to

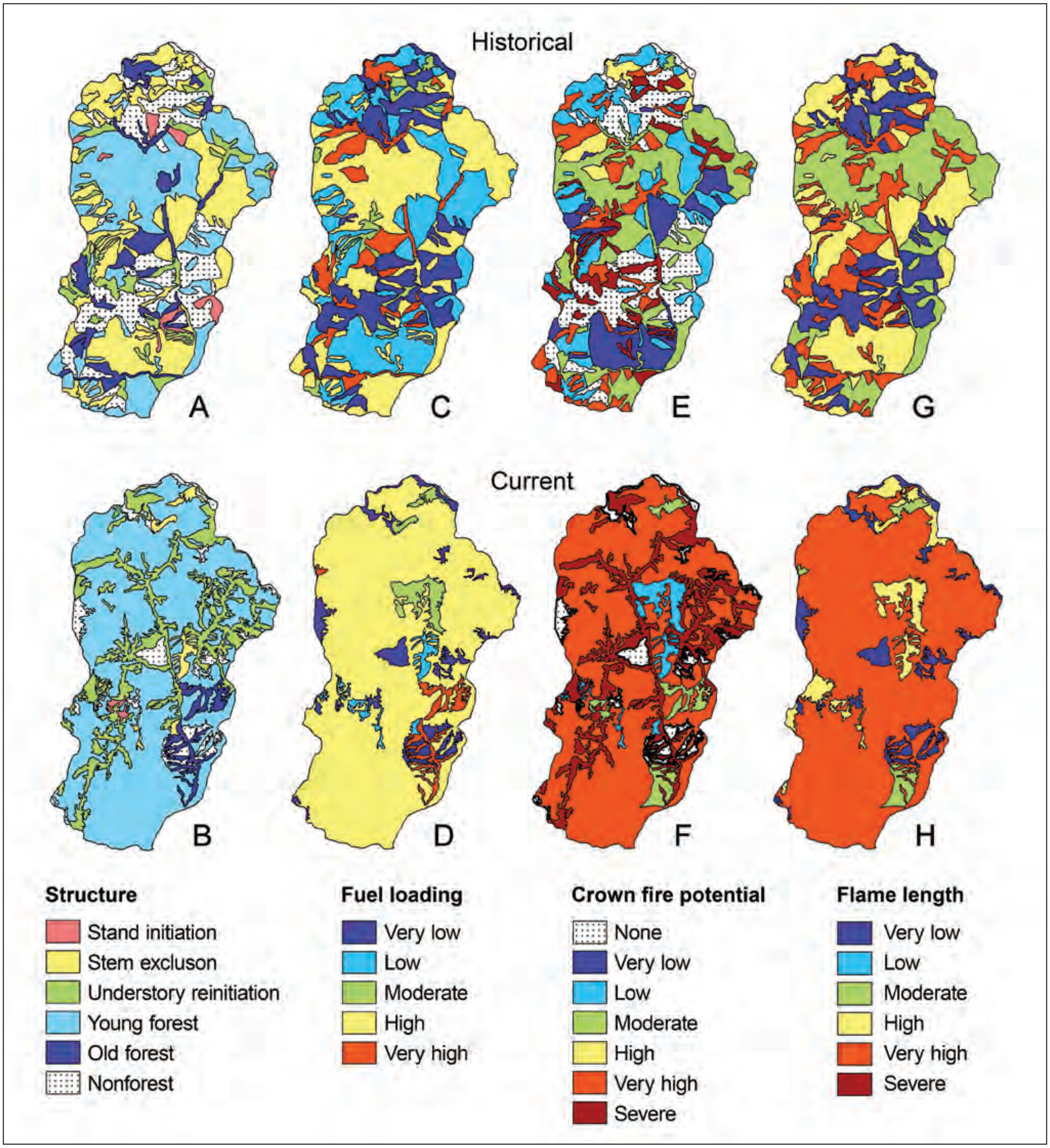

Figure 42—Comparison of an example watershed from the early (ca. 1900, historical) and late (1990s, current) 20th century. Note how spatial patterns of fuel loading, crown fire potential, and flame length closely track forest structural conditions: i.e., process followed pattern. Maps are of the Peavine Creek drainage, a dry forest subwatershed of Lower Grande Ronde subbasin in the Blue Mountains Province displaying historical and current structural classes (A and B), fuel loading (C and D), crown fire potential (E and F), and flame length during wildfire (G and H). Fuel loading classes are very low <22.5 Mg/ha; low = 22.5 to 44.9 Mg/ha; moderate = 45 to 56.1 Mg/ha; high = 56.2 to 67.3 Mg/ha; very high >67.3 Mg/ha. Crown fire potential classes were a relativized index. Flame length classes were very low <0.6 m; low = 0.7 to 1.2 m; moderate = 1.3 to 1.8 m; high = 1.9 to 2.4 m; very high = 2.5 to 3.4 m; severe >3.4 m (Hessburg and Agee 2003, Hessburg et al. 2005).

Table 6—Source forest-habitat trends (1800s to 1995) for families of focal wildlife species in 6th hydrologic unit code (HUC) subwatersheds of the interior Columbia River basin in eastern Oregon and Washington

		Percentage of 6th-code watersheds			
Province	**Family name**	**Decreasing**	**Neutral**	**Increasing**	**Trend**
Blue Mountains	Low-elevation old forest	67	20	13	–
	Broad-elevation old forest	47	17	36	0
	Forest mosaic	7	15	78	+
	Early-seral montane and lower montane forest	53	4	42	–
	Average	43.5	14.0	42.3	0
Northern Cascades	Low-elevation old forest	69	24	7	–
	Broad-elevation old forest	74	13	13	–
	Forest mosaic	17	45	37	0
	Early-seral montane and lower montane forest	30	8	63	+
	Average	47.5	22.5	30.0	0
Southern Cascades	Low-elevation old forest	56	22	22	–
	Broad-elevation old forest	37	15	47	0
	Forest mosaic	0	20	80	+
	Early-seral montane and lower montane forest	45	13	42	0
	Average	34.5	17.5	47.8	0
Upper Klamath	Low-elevation old forest	33	19	48	0
	Broad-elevation old forest	7	5	88	+
	Forest mosaic	5	7	88	+
	Early-seral montane and lower montane forest	98	0	2	–
	Average	35.75	7.75	56.50	0

Note: Trend was determined by >50 percent of subwatersheds decreasing (–) or increasing (+); otherwise, trend was neutral.
Source: Wisdom et al. (2000).

human disturbances or grazing, and associated with shrub-steppe, early-seral, or postfire habitats (e.g., fox sparrow, white-headed woodpecker, western bluebird, and sage thrasher [*Oreoscoptes montanus*]). Species associated with older forest conditions (including northern goshawk, pileated woodpecker, and American marten) also experienced declines in viability relative to historical conditions, but those declines were not as substantial as experienced by the shrub-steppe, early seral, or postfire species.

In their assessment of wildlife habitat values in three national forests in northeastern Oregon (the Umatilla, Wallowa Whitman, and Malheur National Forests), Wales et al. (2011) found that focal species representative of old forest conditions (including goshawk and pileated woodpecker) had current habitat values that were generally consistent with historical conditions at a regional scale, while species associated with open, large tree forests and early-seral or postfire habitats (including Cassin's finch [*Carpodacus cassinii*], white-headed woodpecker, western bluebird, fox sparrow, black-backed woodpecker [*Picoides arcticus*], and Lewis's woodpecker) generally had experienced declines in habitat values relative to historical conditions. Species sensitive to human disturbances (including wolverine and American marten) also experienced declines in habitat values relative to historical conditions. Both Gaines et al. (in press) and Wales et al. (2011) noted that the area of postfire habitat in eastern Washington and Oregon has increased since 1980, but most of it is concentrated in a few watersheds that have experienced large-scale, high-intensity wildfire, producing patterns that did not contribute to population viability for postfire species at a regional scale as effectively as the patterns produced by historical mixed-severity fire regimes.

Two broad factors contributing to reductions in wildlife population viability were highlighted in these assessments: (1) changes in amount and distribution of old forest, deciduous, early-seral, shrub-steppe, and postfire habitats, and (2) impacts of human activities associated with grazing, roads, and recreation. These habitat trends give a general sense of the regional and provincial status of those habitats, but there is much variability in the trends among subwatersheds within provinces. As with stand and landscape management, the variability in conditions is perhaps more important to understand and manage for than the average. Hence, managers need to assess the trends in their local subwatersheds for an accurate assessment of habitat trends and management needs. For example, the trend in broad-elevation old forest habitat was neutral in two of the four provinces; but within those neutral provinces old forest habitat decreased in 47 percent of the subwatersheds in the Blue Mountains and 37 percent of the subwatersheds in the southeastern Cascades (Wisdom et al. 2000).

Human land uses have important effects on animal populations at several scales. At the broadest regional scale, residential, agricultural, and transportation network development has resulted in habitat loss and fragmentation throughout western North America (Theobald et al. 2011). In eastern Oregon and Washington, broad-scale habitat connectivity patterns for species associated with MMC forests are determined by major landform features and associated ecotonal boundaries, in combination with human development patterns (Singleton et al. 2002, WHCWG 2010). Major natural barriers for species associated with MMC forest habitats in the Pacific Northwest include the shrub-steppe landscapes of central Washington and Oregon, and dramatic landforms like the Columbia Gorge, Lake Chelan, the Okanogan Valley (U.S.), and Hells Canyon (Singleton et al. 2002, WHCWG 2010). Major anthropogenic barriers include high-volume highways (e.g., Interstate 90, Interstate 84, portions of U.S. Highway 97, and Canada Highway 3), as well as areas of agricultural, residential, and industrial development. One area of dramatic recent rapid development has been in southern British Columbia, through the Okanogan

Valley and portions of the Highway 3 corridor between Osoyoos and Castelgar (Singleton et al. 2002, WHCWG 2010). These linkage patterns are particularly important for species associated with MMC forests in Washington because populations in Washington are often southern extensions of populations centered in Canada (Singleton et al. 2002, WHCWG 2010).

At a finer scale, forest roads and trails and associated recreational activities can have substantial impacts on wildlife populations associated with MMC habitats (reviewed by Gaines et al. 2003). Common responses of animals to roads and trails include displacement and avoidance, which can contribute to disruption of nesting, breeding, or wintering areas and reduced population density (Benitez-Lopez et al. 2010, Gaines et al. 2003). For example, Naylor et al. (2009) quantified elk behavioral responses to off-road recreational activities including ATVs, mountain biking, hiking, and horseback riding in the Blue Mountains. They found that feeding time decreased and travel time increased in response to these activities. Road access for firewood gatherers can also contribute to loss of large snags and logs in many areas (Hollenbeck et al. 2013). Ecological effects of domestic livestock grazing are reviewed in “Grazing” on page 99.

Altered disturbance regimes and historical land management practices have contributed to substantial changes in the amount and distribution of forest habitats relative to historical conditions, but management activities focused on restoration of historical habitat patterns and stand structure conditions can also have a variety of effects on wildlife populations. These effects can differ substantially across species (reviewed by Kennedy and Fontaine 2009, and Pilliod et al. 2006). At a stand scale, restoration treatments tend to favor those species associated with more open conditions and increased understory vegetation diversity, and have the potential to reduce abundance of species associated with closed canopy, old forest conditions (e.g., Converse et al. 2006, Gaines et al. 2010). However, responses can be quite variable depending on the intensity of the treatment, and although the abundance of individual species can change, restoration treatments generally do not cause substantial changes in species diversity (e.g., Gaines et al. 2010, Lyons et al. 2008, Russell et al. 2009).

Retention of large, old trees, snags, and logs can help maintain the old forest functions of many stands when smaller trees are removed for fuel reduction or forest restoration objectives.

Retention of large, old trees, snags, and logs can help maintain the old forest functions of many stands when smaller trees are removed for fuel reduction or forest restoration objectives. For example, Bull et al. (1995) found that a MMC stand in the Blue Mountains continued to function as old growth for pileated woodpecker and Vaux’s swift (*Chaetura vauxi*) when large trees and logs were retained after treatment to remove small trees killed by western spruce budworm. Most research on the effects of forest restoration or fuel reduction treatments on wildlife have focused on responses soon after treatment (<5 years post-treatment; Kennedy and Fontaine 2009). Longer-term (>10 year post-treatment) effects of forest restoration treatments on animal populations have yet to be well documented. This will be a fertile area for future research as recently treated areas mature and provide opportunities for monitoring the effects of those restoration-oriented treatments.

Old forest habitats are of particular conservation concern because they provide a variety of ecological and social values (reviewed in Spies and Duncan 2009), and old forest characteristics take a long time to regenerate when they are lost as a consequence of high-intensity disturbance or intensive management. In the following section we present information on habitat dynamics and conservation of the NSO. We recognize that much of the area being addressed by this document is outside of the range of the NSO, but we present this information to highlight the lessons from spotted owl conservation that are also important for conservation of other old forest species associated with MMC forests across eastern Washington and Oregon.

The old growth forests of the east Cascades are diverse, dynamic, and shaped by the complex behavior of multiple disturbance factors including fire. The historical structure and function of these forests have been extensively altered by fire exclusion, logging, and other activities (Buchanan et al. 1995, Spies et al. 2006, USFWS 2011). The implications of these changes and the current and future conditions of east Cascades forests for NSO recovery are complex. Habitat conditions for spotted owls in this region are significantly different today than 150 years ago. Although much habitat has been degraded, particularly with the chronic loss of suitable structures for nesting and roosting sites (i.e., selective removal of large trees, removal of snags and damaged live trees) there has also been an increase in dense forest across portions of the landscape (Everett et al. 1997, Hessburg et al. 2005). These changes present multiple challenges for land managers. Where and how much of these dense stands should managers strive to retain? How is retention of dense forest reconciled with meeting restoration goals that necessarily consider the resiliency of a forest in light of impending disturbance factors and climate change?

The recent recovery plan for the NSO (USFWS 2011) stated that the recovery strategy requires action in the face of uncertainty. The plan cites Carey (2007) in advocating “active management for ecological values that will trade short-term negative effects for long-term gains. Collaborative management must be willing to accept short-term impacts and short-term risks to achieve long-term benefits and long-term risk reduction.” The recovery plan described the difficulties in defining conservation objectives for habitat of NSO in these drier forests of the east side:

> Changing climate conditions, dynamic ecological processes, and a variety of past and current management practices render broad management generalizations impractical. Recommendations for spotted owl recovery in this area also need to be considered alongside other land management goals—sometimes competing, sometimes complementary—such as fuels management and invasive species control. In some cases, failure to intervene or restore forest conditions may lead to dense stands heavy with fuels and in danger of stand-replacing fires and insect and disease outbreaks (USFWS 2011: III-20).

In general, dynamic, disturbance-prone forests of the eastern Cascades, California Cascades, and Klamath Provinces could be actively managed in a way that reconciles the overlapping goals of spotted owl conservation, responding to climate change, and restoring dry forest ecological structure, composition and processes, including wildfire and other disturbances

As stewards of NSO habitat, land managers are endeavoring to provide the full complement of habitat requirements. Strong evidence exists suggesting that nesting and roosting habitat, the array of structures in trees that provide a platform, cover, or both, have been largely removed from many areas through decades of selective cutting and sanitation treatments (USFWS 2011). Managers are advised to pay special attention to this habitat management issue when assessing and planning projects, and to make special efforts to retain and restore these habitat elements to the landscape. However, less certainty exists regarding the relationship between owls and habitat used for foraging across the broader landscape. Evidence suggests that owls tend to forage in the moderate- to higher canopy-closure forests and by the same token tend to avoid open stands (reviewed by Courtney et al. 2004).

Northern spotted owl numbers may continue to decline as a consequence of competition with barred owls, but retention and recruitment of habitat continues to be important because reductions in habitat availability are likely to increase competition between the species and will limit opportunities for spotted owls to adapt to the presence of barred owls (Dugger et al. 2011, Forsman et al. 2011, Singleton 2013). Risk and uncertainty are inescapable with management of these forests. Employing a landscape view of assessment and management can help us understand these risks and uncertainties more clearly.

Spotted owls are different from other old forest species associated with MMC forest. For example, northern goshawks and pileated woodpeckers have a broader distribution throughout the Interior West and are associated with a broader range of forest conditions including drier, more open mixed-conifer stands than those used by NSOs, but there are some lessons from spotted owl management that are worth considering for conservation of these other old forest species. In particular, loss of old forest structures to large-scale, high-intensity wildfire is a concern for all old forest species. Landscape conditions that provide for mixed-severity fire patterns and that are resilient to intermittent disturbances will be most likely to provide a shifting mosaic of old forest habitat conditions for all old forest species over time. Managing for patterns that continually bring stands into old forest conditions is also important because we cannot rely on existing old forest stands to remain in that condition over long time periods. This approach of maintaining a shifting mosaic of old forest conditions is likely to be an increasingly important concept for old forest wildlife conservation under future disturbance regimes that are anticipated under many climate change scenarios.

Conservation of unique forest structures (particularly big old trees) is important, but it will be most effective if it is done in the context of sustainable landscape patterns. Big old trees, both vertical and horizontal, living and dead, are important across all stages of stand development (Bull et al. 1997). Where big old trees are absent from a landscape, retaining the largest trees, even of undesirable species, may be appropriate for retaining and developing stand structure. Retaining legacy structures can also provide conditions that facilitate more rapid development of multistory canopy conditions in younger forests.

Another fundamental challenge is that overly aggressive stand sanitization can contribute to a loss of structural diversity. This is a key issue in balancing silvicultural and wildlife habitat objectives at the stand scale. For example, past forest management practices often focused on reducing pathogens like mistletoe and rootrot to enhance fiber production through vigorous tree growth. At moderate levels, these pathogens can enhance spatial and structural complexity within forest stands. Mistletoe can contribute to tree mortality, but it also provides important structures for forest wildlife, particularly in harvested or young stands devoid of snags where mistletoe can create large clumps that function much like cavities to provide nesting and roosting opportunities for a variety of bird and mammal species, including spotted owls and their prey (Parks et al. 1999, Sovern et al. 2011, Watson 2001) (fig. 43). Endemic levels of insects and disease can contribute to stand and spatial heterogeneity important to wildlife, but epidemic levels can be detrimental. We acknowledge that retention of these endemic insect and disease processes involves a delicate balance between tree growth and wildlife habitat objectives, and it entails a certain degree of risk.

Climate change has the potential to strongly influence wildlife populations and habitats (Dalton et al. 2013, Hixon et al. 2013). At the broadest level, changes in the area and location of major habitat types influence wildlife. The habitat types

Peter Singleton

Figure 43—Fine-scale structural diversity associated with dwarf mistletoe clumps and other tree defects provide important habitat components for many wildlife species. This Douglas-fir tree is severely infested with dwarf mistletoe.

that are projected to change in the near future may be of highest concern. Within Oregon and Washington, Rocky Mountain and Cascade subalpine systems are forecast to decline most rapidly (Rehfeldt et al. 2012). Several Forest Service-designated sensitive species are associated with subalpine habitats including wolverine, grizzly bear (*Ursus arctos*), lynx (*Lynx canadensis*), and American marten (Gaines et al. 2000). The wolverine, for example, is thought to be restricted to places with late spring snow pack (Aubry et al. 2010). McKelvey et al. (2011) modeled change in the distribution of such areas in the Western United States under projected future climates. Based on a downscaled ensemble model, they projected that 67 percent of predicted spring snow cover will persist within the study area through 2030–2059, and 37 percent through 2070–2099. Areas with spring snow are currently relatively limited in eastern Oregon and Washington.

Nearly all of these areas except the Eagle Cap Wilderness are projected to entirely lose spring snow by mid-century, which will likely substantially reduce viability of the wolverine populations in those areas (McKelvey et al. 2011). Substantial reductions in the Rocky Mountain montane conifer habitats that cover much of the Okanogan Highland and Blue Mountains are of concern for lynx and the old-forest specialists mentioned above (Aubry et al. 2000). There are many other wildlife species that are likely to be affected by phenological changes resulting from climate change. We are already observing earlier onset of spring and longer, dry summer periods that will increase drought stress (Hixon et al. 2011). These changes will have significant ramifications for the entire biological community (Rosenzweig at al. 2008, Walther et al. 2002).

Maintaining broad-scale landscape patterns that allow species to move in response to changes in climatic and habitat conditions may be particularly important for conservation of sensitive species (Heller and Zavaleta 2009, Nuñez et al. 2013). Climate change also has the potential to alter connectivity for some species. Initial studies have estimated reductions in habitat connectivity under future climate change for wolverine (McKelvey et al. 2011) and American marten (Wasserman et al. 2012). However, we are not aware of studies that have estimated the combined effects of climate and land use change on connectivity in the Northwestern United States.

The effects of climate change are also manifest within the landscape and stand scales most familiar to forest managers. Many wildlife species require access to two or more habitat types in close proximity, such as use of grasslands and forest by elk. Over the past century, ecotones between habitat types have shifted within landscapes owing to changes in climate, livestock grazing, and fire exclusion. In many portions of eastern Oregon and Washington, conifer forests have encroached into grasslands and shrublands (e.g., Hessburg and Agee 2003; Hessburg et al. 2000b, 2005). In the last decade, the frequency of severe fire in the lower forest ecotone has increased (Perry et al. 2011), sometimes leading to a retraction of the forest ecotone. Under future climate and land use, habitat mosaics across landscapes are likely to be even more dynamic. Within forest stands, structural complexity also varies with climate, land use, and disturbance. Fire exclusion can lead to increased density of stems and higher canopy closure; insect outbreaks can result in losses of live large trees and major increases in snags and coarse woody debris. In sum, interactions among climate, land use, and disturbance are expected to bring wide swings in habitat composition within landscape, developmental stage distribution, and within-stand structure. These conditions will challenge forest managers to maintain the stand, landscape, and regional habitat components required by many native species.

Altered snowmelt run-off regimes in the mixed-conifer forests will affect downstream fish-bearing reaches. Discharge patterns affect water depth; thus, earlier low flows downstream may reduce habitat availability for fish dependent on deeper, slower flowing habitats (Luce et al. 2013, Tague and Grant 2009). Cutthroat trout and bull trout are the most sensitive species likely to be present in mixed-conifer zones. Bull trout occupy the coldest freshwater habitats of all salmon and trout species and these habitats are predicted to be severely affected by climate change in coming decades (Isaak et al. 2010, Rieman et al. 2007, Wenger et al.

Sensitive Species of Moist Mixed-Conifer Forests

A variety of terrestrial wildlife species is associated with the moist mixed-conifer (MMC) forests and surrounding landscapes in the Pacific Northwest, including one endangered species (gray wolf [*Canis lupus*], recently delisted in part of its range), three federally listed threatened species (NSO, grizzly bear, and lynx), and three federal candidate species (Oregon spotted frog [*Rana pretiosa*], fisher [*Martes pennanti*], and wolverine). In addition, several other wildlife species of concern recognized by state or federal agencies inhabit these forests (table 7). Many of the species that are at risk in the region are associated with particular stand structure conditions, vegetation community types, and landscape configurations. For example, NSOs are found in MMC forests on the east slope of the Cascades from the Canadian border into northern California (USFWS 2011). They are associated with structurally diverse, large tree, moderate- to closed-canopy conditions generally found in older forests that do not have a recent history of fire, much of which has been altered by both logging and more recently large fires in the last 50 to 100 years. A small population of grizzly bears inhabits MMC forest in the Pacific Northwest. Distribution of this population is limited to a few areas along the Canadian border in the North Cascades and Kettle Mountain/Wedge areas of northeast Washington (Gaines et al. 2010, USFWS 1997). Grizzly bear habitat associations in the Cascades and northeast Washington include moist forests, particularly areas of those forests that are interspersed with moist meadows or avalanche chutes where food plants are abundant (Gaines et al. 2000). Oregon spotted frogs were once found near moving water through much of the Cascades in Oregon and Washington (Cushman and Pearl 2007, McAllister and Leonard 1997). More recently, populations have been documented in Washington's southeast Cascades (near Trout Lake and Conboy in Klickitat County) and in Oregon's south-central Cascades. The historical range of fishers encompasses mesic interior forests in the Oregon and Washington Cascades, northeast Washington, and Blue Mountains (Lewis and Stinson 1998). Fishers were extirpated from most of that range as a result of fur trapping. A small population is present in Oregon's southern Cascades but at present they are largely absent from interior forests in Oregon and Washington.

Table 7—Sensitive wildlife species found in interior mesic forests, including species broadly associated with interior moist forest and ecotones; many of these species are associated with specific forest structure or seral stages found within the interior mesic forest zone

Common name	Scientific name	Class	Status[a]	Comments
Western toad	*Anaxyrus boreas*	Amphibian	FCo	
Rocky Mountain tailed frog	*Ascaphus montanus*	Amphibian	FCo	
Oregon slender salamander	*Batrachoseps wrighti*	Amphibian	FCo	Eastern Oregon Cascade Range
Larch mountain salamander	*Plethodon larselli*	Amphibian	FCo	Southeast Washington Cascade Range
Columbia spotted frog	*Rana luteiventris*	Amphibian	SC	
Northern leopard frog	*Rana pipiens*	Amphibian	FCo	
Oregon spotted frog	*Rana pretiosa*	Amphibian	FC	
Northern goshawk	*Accipiter gentilis*	Bird	FCo	
Upland sandpiper	*Bartramia longicauda*	Bird	SE	
Vaux's swift	*Chaetura vauxi*	Bird	SC	
Olive-sided flycatcher	*Contopus cooperi*	Bird	FCo	
Black swift	*Cypseloides niger*	Bird	FCo	Associated with waterfalls and wet cliffs
Pileated woodpecker	*Dryocopus pileatus*	Bird	SC	
Peregrine falcon	*Falco peregrinus*	Bird	FCo	
Harlequin duck	*Histrionicus histrionicus*	Bird	FCo	
Lewis' woodpecker	*Melanerpes lewis*	Bird	SC	
Flammulated owl	*Otus flammeolus*	Bird	SC	Associated more with dry forest
White-headed woodpecker	*Picoides albolarvatus*	Bird	FCo	Associated more with dry forest
Black-backed woodpecker	*Picoides arcticus*	Bird	SC	
Northern spotted owl	*Strix occidentalis*	Bird	FT	
Silver-bordered fritillary	*Boloria selene atrocostalis*	Insect	SC	
Manns mollusk-eating ground beetle	*Scaphinotus mannii*	Insect	SC	
Gray wolf	*Canis lupus*	Mammal	FE[b]	
Townsend's big-eared bat	*Corynorhinus townsendii*	Mammal	FCo	
Spotted bat	*Euderma maculatum*	Mammal	FCo	
Wolverine	*Gulo gulo*	Mammal	FC	
Silver-haired bat	*Lasionycteris noctivagans*	Mammal	FCo	
Lynx	*Lynx canadensis*	Mammal	FT	
Fisher	*Martes pennanti*	Mammal	FC	Northeast Washington, east of Okanogan River
Long-eared myotis bat	*Myotis evotis*	Mammal	FCo	
Long-legged myotis bat	*Myotis volans*	Mammal	FCo	
Woodland caribou	*Rangifer tarandus caribou*	Mammal	FE	Selkirk Mountain population in northeast Washington
Preble's shrew	*Sorex preblei*	Mammal	FCo	
Grizzly bear	*Ursus arctos*	Mammal	FT	
Poplar Oregonian	*Cryptomastix populi*	Mollusk	SC	
Dalle's sideband	*Monadenia fidelis minor*	Mollusk	SC	
Western pond turtle	*Actinemys marmorata*	Reptile	SC	
Sharptail snake	*Contia tenuis*	Reptile	FCo	Local in eastern slope of the Cascade Range

[a] Status codes are FE = federal endangered, FT = federal threatened, FC = federal candidate, FCo = federal species of concern, SE = state endangered, SC = state species of concern.

[b] Recently delisted east of U.S. Highway 97.

Data sources: Oregon or Washington federal status from http://www.fws.gov/oregonfwo/species/Lists/Documents/OregonStateSpeciesList.pdf and http://www.fws.gov/wafwo/pdf/specieslist120613.pdf. Washington state status from http://wdfw.wa.gov/conservation/endangered/All/. Oregon state status from http://www.dfw.state.or.us/wildlife/diversity/species/sensitive_species.asp.

2011). Changing precipitation and fire regimes are expected to compound the effects of warming trends by shifting hydrologic patterns and those of sediment transport and solar radiation (Dunham et al. 2007, Isaak et al. 2010).

Bull trout life histories include migratory forms that spawn and rear in cold streams close to headwaters, then migrate downstream to larger rivers where they live as adults prior to spawning ("migratory") or migrate to lakes following rearing ("adfluvial"). Fragmentation of cold water habitats by stream warming can increase physiological stress to bull trout and decrease interconnectivity of adequate spawning and rearing aggregations. In the "resident" life history form, bull trout remain in cold headwater streams for spawning, rearing, and as adults. Thus, they are subject to these physiological stressors at all life stages.

Cutthroat trout are also common to upland streams in mixed-conifer forests of the eastern Cascades. In small streams within mixed-conifer watersheds of the Wenatchee River sub-basin, cutthroat trout density and total biomass are limited by higher flows (fig. 36) (see footnote 1). Thus, changing flow regimes as a result of precipitation and snowmelt-timing shifts can potentially have consequences for the persistence of populations of this species.

Current Socioeconomic Context

When this science synthesis was initiated, the decision was made to focus on the biophysical science associated with managing MMC forests. However, management of these forests takes place within the broader socioeconomic context of eastern Oregon and Washington, which both influences and is influenced by forest management.[4] In addition, some concepts of resilience include interactions between social and ecological subsystems (see "The Concept of Resilience" on page 18). Thus, some important socioeconomic issues relevant to forest management in the region are briefly reviewed here, recognizing that the topics covered are not all-inclusive. We focus on issues most relevant to accomplishing forest restoration on eastside forests by creating a more enabling social and economic environment. Two such topics—collaboration and the social acceptability of forest restoration treatments—are addressed in chapter 6. The key issues of focus in this section are wood products infrastructure and business capacity to support forest restoration, opportunities for biomass utilization, and some of the tribal concerns associated with forest restoration in the region. Also highlighted are the potential impacts of restoration treatments on recreation. Germane management implications are included in each sub-section. It is important to note that the literature addressing

[4] For purposes of this section, eastern Oregon and Washington refer to all of the counties that lie east of the Cascade crest in Oregon and Washington.

the socioeconomic context of forest management in eastern Oregon and Washington is rarely specific to the MMC forest type; is more extensive for eastern Oregon than for eastern Washington; is largely gray literature (as opposed to peer-reviewed literature); and consists mainly of assessment work rather than scholarly research.

The region is dominated by rural counties where natural resources have historically played an important role in contributing to local economies through agriculture, ranching, forestry, and more recently, recreation. Many of eastern Oregon's counties exhibit high levels of poverty and unemployment however (Davis et al. 2010), and many eastern Washington counties have been characterized as having relatively low socioeconomic resilience (Daniels 2004). Social and economic assessments of interior Columbia River basin communities carried out as part of the Interior Columbia Basin Ecosystem Management Project (Harris et al. 2000, Reyna 1998) found that small population size, geographic isolation, low economic diversity, heavy dependence on one or a few employment sectors, low level of money coming in from the outside, poorly developed infrastructure, and less active leadership all contribute to low community resilience in the region.[5] Conducting forest restoration can potentially increase community resilience by sustaining existing and creating new and future job opportunities, diversifying the local employment base, retaining existing jobs and stimulating new infrastructure development, and spurring innovations and investments to create new business and employment opportunities—all of which contribute to community resilience (Berkes and Ross 2013, Davidson 2010, Magis 2010, Walker and Salt 2006).

The region is dominated by rural counties where natural resources have historically played an important role in contributing to local economies through agriculture, ranching, forestry, and more recently, recreation.

The Forest Service 2012 Planning Rule provides direction to contribute to social and economic sustainability through forest management to help support vibrant communities and rural job opportunities. In addition to employment, other economic benefits resulting from fire hazard reduction through forest restoration may include reduced fire suppression costs over the longer term, increased production of wood products, higher state tax revenues, reduced unemployment payments, and lower expenditures on social services (State of Oregon 2012). Forest restoration to improve forest health can also benefit other ecosystem services associated with national forest lands, including clean water, fish and wildlife habitat, and recreation activities (State of Oregon 2012). Frameworks for identifying the ecosystem services from national forests that are important to stakeholders, and for evaluating the social and ecological tradeoffs between services that are associated with different forest management actions, have been developed by Asah et al. (2012), Kline (2004), Kline and Mazotta (2012), and Smith et al. (2011).

[5] Community resilience is defined here as the ability of a community to successfully cope with, adapt to, and shape change and still retain its basic function and structure.

Forest restoration to reduce hazardous fuels also reduces the risk of wildland fire to communities located in the wildland-urban interface.

Wood Processing Infrastructure and Business Capacity for Forest Restoration

Decreased timber harvesting on federal lands in Oregon and Washington (and elsewhere in the West) since the late 1980s, together with changing technology and markets, industry restructuring, and the recession of 2007–2009 have led to dramatic declines in the production of wood products and mill infrastructure throughout these states (Charnley et al. 2008, Keegan et al. 2006, OFRI 2012). In Oregon and Washington, most of the remaining mills are on the west side of the Cascades (Keegan et al. 2006, Nielsen-Pincus et al. 2012). The increase in severe, large-scale fires that has occurred during the 2000s points to the need for hazardous fuels reduction, in which treatments to remove trees of different sizes play an important role (Keegan et al. 2006). Such removals may include trees having commercial value, as well as trees with smaller diameters than what have traditionally been considered merchantable. Important questions are how to retain what remains of the wood processing infrastructure, whether existing mills can process trees of different size classes (Keegan et al. 2006), and how to develop a new and more diversified infrastructure.

In eastern Oregon, the primary wood processing infrastructure that exists today represents about 20 percent of what it was in the 1980s (Swan et al. 2012). Associated milling capacity has also declined (OFRI 2012). In 2012, there were 45 major primary wood processing facilities and operations in eastern Oregon, including 11 open and three closed sawmills, two open plywood plants, and a number of chipping facilities and operations, post and pole mills, species-specific specialty mills, firewood processors, and whole log shaving operations (Swan et al. 2012). Together, these wood processing facilities employed an estimated 1,730 people (at minimum). Although this diversified mill infrastructure provides opportunities for producing a range of products from logs of different sizes and types, existing mills are not operating at full capacity. Sawmills and chip mills currently operate at about 50 percent of capacity, and whole log mills at 40 percent. Increasing the capacity of their operations could potentially increase employment at eastern Oregon's wood processing facilities by 35 percent (Swan et al. 2012). Doing so would require an increased and sustained supply of logs and fiber of the appropriate sizes and species, favorable market conditions, adequate financing, and a reliable, experienced workforce (OFRI 2012, Swan et al. 2012).

With regard to production from MMC forests to help meet supply needs, eastern Oregon mills that produce lumber graded for strength prefer interior Douglas-fir and white fir, but also use Engelmann spruce, lodgepole pine, and subalpine fir (Swan et al. 2012). Mills that produce lumber graded for appearance prefer ponderosa pine but also use high-quality white fir. Pulp and paper mills that consume wood products from eastern Oregon, located mainly along the Columbia and Snake Rivers, prefer lodgepole pine and white fir, but accept other tree species. Size preferences typically range from 31 to 41 cm (12 to 16 in) diameter for large logs (though some mills take up to 56 cm [22 in] logs), and 10 to 15 cm (4 to 6 in) to 25 to 30 cm (10 to 12 in) for small logs (Swan et al. 2012).

Ownership of eastern Oregon's timberlands is 67 percent Forest Service and 29.2 percent private (OFRI 2012). However, 75 percent of the current supply of wood products for industry in eastern Oregon comes from nonfederal lands, and 25 percent from federal lands (nearly all Forest Service) (Swan et al. 2012). Eastern Oregon sawmills obtain an average of 63 percent of their supply from nonfederal (almost all private) lands (Swan et al. 2012).

Assessments have found that in eastern Oregon, private lands will not be able to support pre-economic recession (2007) harvest levels in a sustainable way, implying that increased wood supplies will need to come largely from federal lands (OFRI 2012). Harvests from federal lands in eastern Oregon are currently estimated at 7 percent of annual growth (OFRI 2012).

The nature of the supply from Forest Service lands has also shifted significantly since the 1980s from predominantly sawlogs greater than 31 cm (12 in) in diameter, to mostly non-saw logs (biomass). Thus, to significantly increase forest restoration treatments in eastside forests, new investment in mill facilities designed to process smaller-diameter material will likely be needed, which in turn will hinge on public support and a reliable supply of raw material over a 10-year time horizon at a minimum (OFRI 2012). The State of Oregon (2012) provided a number of recommendations for how to increase the scale of restoration on Forest Service lands in eastern Oregon.

There is also a need in eastern Oregon to recruit more people into the logging workforce and to increase access to capital to support investments in new equipment (OFRI 2012). One consequence of the loss of forest products industry infrastructure has been a loss of people having the skills and knowledge to work in eastern Oregon's forestry sector (OFRI 2012). The business capacity to engage in forestry support work (e.g., activities in support of timber production, firefighting, and reforestation) has been growing in eastern Oregon over the past decade (Davis et al. 2010). Retaining and increasing the local infrastructure, workforce, equipment, and businesses needed to engage in forest management is critical for carrying out landscape-scale forest restoration to improve the ecological integrity and resilience of forest ecosystems (Kelly and Bliss 2009, State of Oregon 2012).

Another consequence of reduced forestry infrastructure has been that the average distance between mills is often greater than 161 km (100 mi) in eastern Oregon, increasing log haul distances and associated transportation costs (Swan et al. 2012). This leads to reduced competition for logs—driving down stumpage prices—and higher transportation costs to get logs to more distant mills, which together can result in uneconomical timber sales, making it more difficult and costly to engage in forest restoration and hazardous fuels reduction. Nielsen-Pincus et al. (2012) found that national forest ranger districts that are close to sawmills and biomass facilities treated more overall hectares for hazardous fuels reduction, and more hectares in the WUI, than those farther away, and that there was a threshold distance (hauling time) for this effect in Oregon and Washington of 40 minutes. A lack of local wood processing infrastructure constrains the management options available to forest managers and, in turn, affects managers' capacity to address declining forest health and increasing fire risk (Eastin et al. 2009).

In Washington state, there was a steady decline in the number of wood processing facilities between 1991 and the 2000s, with several mills closing along the east side of the Cascades, and the remaining industry shifting from rural to urban areas to be nearer to major transportation corridors (WA DNR 2007). In 2010, there were 14 mills in central and eastern Washington, half of them sawmills (Smith 2012). Three pulp mills, one veneer and plywood mill, and three roundwood chipping mills comprise the remainder of the infrastructure. Few new mills have replaced the mills that closed, though the remaining mills tend to be larger and more efficient (Smith 2012). The distance between mills is greater than 322 km (200 mi) in some parts of eastern Washington however, meaning high transportation costs for what are often low-value, small-diameter logs, and less competitive prices (WA DNR 2007).

The dominant log species consumed by eastern Washington mills in 2010 were Douglas-fir (38 percent), ponderosa pine (25 percent), true firs (18 percent), and lodgepole pine (9 percent) (Smith 2012). Of these logs, 48 percent came from private lands, 21 percent from tribal lands, 17 percent from state lands, and 11 percent from national forest lands. The dominant size classes harvested were 13 to 25 cm (5 to 10 in) (40 percent) and 25 to 51 cm (10 to 20 in) (36 percent) (Smith 2012).

Studies have found that in order to achieve forest restoration goals in eastern Washington, new investments in wood processing infrastructure will be needed (WA DNR 2007). Again, having a stable and adequate supply of wood is critical for stimulating such investments. Increasing harvests from state and private lands to meet supply needs is unlikely to be sustainable however, implying an important role for federal land harvests. Other factors that would help would be government incentives for investing in wood processing facilities, diversification of processing infrastructure (as has occurred in eastern Oregon), establishing markets for carbon and biodiversity, and reducing local regulatory constraints to mill construction and forest products manufacturing (WA DNR 2007).

Ensuring a reliable supply of wood of all size classes from federal lands in eastern Oregon and Washington is critical for maintaining existing and establishing new wood processing infrastructure.

Ensuring a reliable supply of wood of all size classes from federal lands in eastern Oregon and Washington is critical for maintaining existing and establishing new wood processing infrastructure (Keegan et al. 2006). To stay in business, mills also need to remain competitive by investing in improvements such as increasing the efficiency of log conversion, producing higher value products and a more diverse product mix, or constructing drying kilns on site (Dramm 1999). Stewardship contracting, where appropriate, is one tool that can help in this regard because it allows the Forest Service to enter into contracts of up to 10 years in duration and to provide a reliable supply of wood. The Collaborative Forest Landscape Restoration Program is another mechanism that should encourage a more reliable supply of small-diameter wood to support industry investments in infrastructure in participating landscapes (which currently include four areas of eastern Oregon and Washington, plus one affiliate in Washington). This program aims not only to encourage national forests to commit to carrying out restoration projects by providing a source of funding for up to 10 years, but to incentivize restoration treatments on other ownerships within the same landscapes by leveraging funding from other sources (Schultz et al. 2012). Doing so may help increase the supply from other ownerships. The program also requires projects to make use of existing or proposed processing infrastructure to support jobs and local economies (Schultz et al. 2012).

Developing biomass utilization opportunities is also important for retaining and increasing the number of sawmills. Sawmills produce a large volume of residual material associated with log processing that can be used for many biomass utilization applications, as well as for producing pulp and paper and other wood products (Davis et al. 2010). Unless they can market these residuals, sawmills may not be able to operate economically (Swan et al. 2012). The next section addresses this topic.

Biomass Utilization

The development of woody biomass utilization opportunities has received much attention over the past decade because (1) biomass is a domestic source of renewable energy; (2) biomass utilization can help to partially offset the cost of needed hazardous fuels reduction treatments on public lands and contribute to economic development opportunities in forest communities (Aguilar and Garrett 2009, Morgan et al. 2011, Nechodom et al. 2008); and (3) biomass utilization reduces the need for onsite burning of piled material produced by fuels treatments, and associated environmental effects (Daugherty and Fried 2007, Springsteen et al. 2011). To date, biomass utilization infrastructure remains underdeveloped in eastern Oregon and Washington. As of 2012, there was only one stand-alone biomass energy cogeneration plant in eastern Oregon, which was temporarily closed because it could not obtain a favorable power sales agreement (Swan et al. 2012). When market conditions are good, biomass from eastern Oregon may be transported to western Oregon or to mills along the Columbia and Snake Rivers for use, but markets are highly variable. There were no stand-alone biomass energy facilities in eastern Washington in 2010 (Smith 2012), posing a challenge for forest restoration there (WA DNR 2007). Davis et al. (2010) provide an overview of status and trends in woody biomass utilization, including challenges and opportunities, across many eastern Oregon counties.

A number of economic issues constrain the development of viable biomass utilization facilities. To attract investors, there must be an adequate and predictable supply of biomass, a concern in places where federal land is the main potential source of supply (Becker et al. 2011, Hjerpe et al. 2009). The supply problem could be addressed by diversifying sources of raw material, through the use of stewardship contracts, or by addressing internal institutional barriers to biomass utilization within the Forest Service (Becker et al. 2011, Hjerpe et al. 2009, Morgan et al. 2011). Supporting remaining wood products industry infrastructure to prevent its further loss can also help provide opportunities for biomass removal and utilization. For a number of reasons, the presence of wood products industry infrastructure has been found to enhance the development or expansion of biomass utilization, which is difficult to develop as a stand-alone enterprise (Becker et al. 2011).

The cost of harvesting and transporting biomass is another key constraint (Aguilar and Garrett 2009, Becker et al. 2009, Pan et al. 2008). Becker et al. (2009) found that the cost of transporting biomass from the harvest site to the market outlet is the single greatest cost associated with biomass utilization, and that decreasing the travel distance between markets and harvest sites is the only strategy that offsets this cost in a meaningful way. Strategies for addressing the cost issue include establishing a network of decentralized processing facilities of an appropriate size and type closer to the source where biomass is removed (Aguilar and Garrett 2009, Nielsen-Pincus et al. 2012); developing utilization options that focus on higher value products; bundling biomass removal with the removal of larger trees that produce higher value products (e.g., lumber) (Barbour et al. 2008); developing transportation subsidies, which Oregon has done (though these can be problematic) (Becker et al. 2011, Nicholls et al. 2008); and implementing financial incentives such as cost sharing and grant programs for facility development and equipment purchases, and tax incentives for facility development and harvesting and transporting biomass (Sundstrom et al. 2012).

Because biomass produced as a byproduct of forest restoration tends to be of low value, strategies associated with national forest management are likely to focus on establishing smaller processing facilities closer to public lands (Becker et al. 2011). Small and mid-sized facilities that focus on electricity generation, firewood, animal bedding, commercial heating, or combined heat and power systems are likely to be more feasible than large processing facilities because they tend to be less controversial and require a smaller supply of biomass to operate (making it easier to obtain in a reliable manner) (Becker et al. 2011).

Markets are another major constraint on increasing biomass utilization. Key to increasing biomass utilization east of the Cascades is stimulating market demand and opportunities. The abundance of hydropower in the Pacific Northwest limits market opportunities for biomass electricity (Stidham and Simon-Brown 2011). Although low natural gas prices may also threaten the viability of biomass energy projects (OFRI 2012), eastern Oregon has large areas that lack access to natural gas, and woody biomass is more economical for heating than propane or heating oil (Swan et al. 2012).

The main constraints on developing stand-alone, industrial-scale biomass energy plants in eastern Oregon currently are low prices for electricity under power purchase agreements, and uncertain pricing in the California market, making construction of such plants unlikely in the near future (Swan et al. 2012).

Although the supply of biomass for manufacturing value-added products is greater than market demand (Stidham and Simon-Brown 2011), promising opportunities for biomass utilization lie in developing a broader range of products and applications such as pellets, bricks, small-scale institutional thermal applications (such as space and water heating), onsite electricity generation, transportation fuels, and bio-char (Davis et al. 2010, OFRI 2012, Swan et al. 2012). More diversified and integrated biomass utilization projects have been developing in eastern Oregon in recent years (Davis et al. 2010), signaling progress that may enhance forest restoration efforts there.

Tribal Concerns

Tribal concerns associated with the management of MMC forests on federal lands in eastern Oregon and Washington include protecting tribal lands from fire, insects, and disease; creating job opportunities for tribe members in forest restoration; management to address the potential impacts of climate change on forest resources that are important to tribes; and incorporating tribe members and their traditional ecological knowledge into forest restoration decisionmaking and implementation.

The 2004 Tribal Forest Protection Act (TFPA) was passed to protect tribal lands, resources, and rights from fire, insects, disease, and other threats (ITC 2013). The Act allows tribes to propose fire mitigation and environmental restoration activities on Forest Service and Bureau of Land Management lands adjacent to or bordering tribal trust lands and resources. The agencies may enter into contracts or agreements with tribes for this purpose. Despite the potential that TFPA authorities offer for fulfilling federal trust responsibilities, enhancing forest restoration activities, and creating job opportunities for tribe members in fuels reduction and postfire rehabilitation, only a handful of projects have been implemented. Reasons for this lack are described in ITC 2013. Ways that federal forest managers in eastern Oregon and Washington can take greater advantage of the opportunities provided by the Act include strengthening partnerships with tribes through formal agreements, promoting the use of the TPFA authorities internally, and conducting training and outreach to increase understanding and utilization of the TPFA (ITC 2013).

Regarding job opportunities, a survey of 31 of the 42 federally recognized tribes in Oregon, Washington, and Idaho found that tribes had a strong interest in obtaining jobs in fire management, such as work on wildland fire suppression crews and hazardous fuels reduction work (Rasmussen et al. 2007). A number of strategies for promoting tribal economic development through fire management can be found in the Tribal Wildfire Resource Guide (Intertribal Timber Council and Resource Innovations 2006). Obstacles that limit the capacity of tribes to engage in this work include the seasonality of the work, the training required for employees and contractors, the cost of investing in equipment, a lack of financial capital with which to start businesses, and supportive tribal leadership to help form partnerships with public agencies (Rasmussen et al. 2007). Different communication and operating styles and Forest Service bureaucratic processes can also create barriers (Charnley et al. 2007). To the extent that the Forest Service can assist tribes in addressing some of these obstacles, it can help build the capacity of tribal communities to engage in fire management and forest restoration activities.

The environmental impacts of climate change are expected to be disproportionately felt by American Indian and Alaska Native tribes relative to nonnative communities because of their unique rights, economies, and cultures that are linked to the natural environment.

The environmental impacts of climate change are expected to be disproportionately felt by American Indian and Alaska Native tribes relative to nonnative communities because of their unique rights, economies, and cultures that are linked to the natural environment (Lynn et al. 2011). In eastern Oregon and Washington, climate change impacts on forests that may affect tribes include the potential for increased frequency and intensity of wildland fire; increases in invasive species, insects, and disease; and shifts in the quantity, quality, and distribution of culturally important forest resources (plants, animals, fungi, water, and minerals) (Voggesser et al. 2013). As a result, climate impacts on forests may threaten tribal subsistence, culture, and economies. The traditional ecological knowledge held by tribe members can play an important role in helping federal forest managers identify species having social, cultural, and economic importance to tribes in the MMC zone, and in helping to inform management strategies for these species in the context of climate change. Strong federal-tribal relationships around forest management may also help to mitigate climate change impacts on species important to tribes, and assist tribes in developing adaptation strategies (Voggesser et al. 2013). Resources and research to support federal forest management in a manner that is responsive to the climate change-related concerns of tribes in the Pacific Northwest can be found at http://tribalclimate.uoregon.edu/.

The relationship that exists between federally recognized tribes and the United States means that federal agencies are required to consult with tribes when they engage in policy making or undertake actions that affect tribal interests and resources. Consultation processes also promote collaboration between tribes and federal agencies in protecting and managing tribal resources on and off reservation lands (Whyte 2013). Working toward effective consultation processes will help ensure that restoration activities on national forests east of the Cascades address tribal concerns. Traditional ecological knowledge may also make an important contribution to forest restoration on national forest lands and can be fostered through federal-tribal consultation and partnerships and by directly engaging traditional knowledge holders in planning and implementing restoration activities (Vinyeta and Lynn 2013). A number of models for doing this are reviewed in Charnley et al. (2007), Donoghue et al. (2010), Vinyeta and Lynn (2013), and Voggesser et al. (2013).

Recreation

Moist mixed-conifer forests have high recreation value. The status of and trends in recreation activities on national forests in eastern Oregon and Washington can be found in the National Visitor Use Monitoring survey reports for these forests (http://www.fs.fed.us/recreation/programs/nvum/). Assessment information about recreation activities, trends, and issues in eastern Oregon and Washington can also be found in the Statewide Comprehensive Outdoor Recreation Plan (SCORP) reports for these states (http://www.oregon.gov/oprd/PLANS/pages/scrop08_12.aspx; http://www.rco.wa.gov/documents/rec_trends/SCORP_2008.pdf), and in Hall et al. (2009). National forest recreation in MMC forests is likely to be affected by forest restoration treatments, which could be a source of contention. Exactly how recreation values may be affected by restoration treatments is difficult to predict, however, in the absence of site-specific research.

A number of studies have been carried out in the Intermountain West to evaluate the effects of prescribed fire treatments and crown fires on recreation visitation and associated economic benefits. These studies have found that crown fires generally have a negative effect on recreation visitation and economic values over time, although there may be an initial, short-term positive response by some types of visitors (Englin et al. 2001; Hesseln et al. 2003, 2004; Loomis et al. 2001). Hikers, for example, may be drawn to see wildfire effects and wildflower blooms following a crown fire (Englin et al. 2001, Loomis et al. 2001). Longer term declines in recreation visitation may result from the negative aesthetic effects of the fire, reduced access for certain activities, or damage to recreation facilities and infrastructure (Kline 2004). Areas that have substantially recovered from a crown fire may experience a rebound in visitation, however (Englin et al. 2001). In contrast, a study from Oregon's Mount Jefferson Wilderness found that the B&B Fires of 2003 did not dramatically affect recreation in the first few years following the fire (Brown et al. 2008).

Prescribed fires have been found to have either no effect or a positive effect on recreation visitation and associated economic benefits over time in some places (Hesseln et al. 2004, Loomis et al. 2001). These effects can vary by recreation activity (e.g., hiking versus mountain biking) (Loomis et al. 2001). Elsewhere, prescribed fires have been found to have a negative effect on recreation visitation by hikers and mountain bikers and associated economic values, with visitation decreasing as the percent of a burn visible from the trail increased (Hesseln et al. 2003).

These findings have a number of management implications. Because crown fires can be detrimental to recreation visitation and values, fuels reduction treatments may be one way of mitigating the negative social and economic impacts of crown fires on recreation (Hesseln et al. 2004). In some places, mechanical treatments may be more effective at mitigating these impacts than prescribed fire, and have fewer overall social and economic costs, because prescribed fire may also have a negative impact on recreation visitation and values (Hesseln et al. 2003). If prescribed fire is used, minimizing the area of the prescribed burn visible from popular recreation trails may reduce its negative effects. It is important to note, however, that some wildfires, such as fires that take place in areas dominated by shrubs, may have positive effects on recreation by creating more open forest conditions (Kline 2004). The ways in which wildfire and fuels treatments affect recreation vary by recreation activity, geographic location, type of wildfire and fuels treatment, forest characteristics, and over time (Kline 2004). Thus, the best way forward for identifying how different types of recreation activities may be affected by restoration treatments, and the most appropriate approach to forest restoration in places where recreation values are high, is likely to be through place-based research and collaborative processes (Hesseln et al. 2003, Kline 2004).

This brief overview of socioeconomic issues associated with forest restoration in eastern Oregon and Washington is not intended to be all-inclusive. Important issues not addressed include community wildfire protection, the effects of restoration treatments on other ecosystem services provided by national forests, the challenge of forest restoration at the landscape scale across ownership boundaries, and institutional barriers to forest restoration within federal agencies. This overview aims to touch on some of the key socioeconomic issues managers may face in trying to restore MMC forests, and provides some guidance about how to address these issues so that managers can evaluate both ecological and socioeconomic considerations when making decisions about forest restoration. Remaining research gaps with regard to the socioeconomic dimensions of managing MMC forests make it difficult to develop a robust framework for evaluating the social and economic tradeoffs entailed in management decisionmaking.

Summary of Key Scientific Findings and Concepts

The previous sections synthesize scientific findings most relevant to forest managers. We encourage the reader to investigate more thoroughly any of the topics discussed using the extensive reference section (about 400 citations) provided.

Our key scientific findings and concepts are as follows:

Landscapes

- **Broad-scale (top-down), meso-scale (within landscape), and fine-scale (bottom-up) spatial controls (aka, drivers or forcing factors) combine to control ecosystem behavior and ecological outcomes.** Examples of top-down drivers are broad-scale patterns of the regional climate, geology, land surface forms, and broad patterns of biota—lifeforms and land cover types. Meso-scale or within landscape spatial controls are natural and human disturbances, topography, patterns of structure, composition, fuelbeds, and patch size distributions. Examples of fine-scale drivers are patterns of local topography, microsite, soils, plant communities and plant life history and life cycle differences, patterns of micro-climate, and the like. Some patterns at each of the three scales are not likely to be stationary in time or space, while others (geology, topography, geomorphic processes) are relatively more stationary. Any given landscape today is on a trajectory to a future condition that will be a result of these combined forces.

- **Regional landscapes can be viewed as a multi-level hierarchy of mosaics** for landscape planning and management purposes; patches exist within local landscapes, which provide the patchiness and variability of the regional landscape. Patchiness also exists within patches (clumped tree distributions and gaps of various sizes) and this forms the variability of the local landscape.
- **Steep, topographically driven precipitation and temperature gradients have a significant influence** on the vegetation east of the Cascade Range divide; the most significant effect is the rain shadow created by the high divide.
- **Forest productivity and disturbance regimes are highly influenced by these gradients.** Position on the landscape combined with local weather conditions (precipitation and temperature), surface lithologies (parent materials and the soils that are derived from them), and land surface forms contribute to a complex mosaic of patches supporting a diverse array of vegetation. **On any given landscape, dry, moist, and wet mixed-conifer forests occur within a large ecological gradient.** As a consequence it is difficult to neatly disentangle these forest types for management purposes.
- **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.
- **The historical range of variability (HRV) that created pre-Euro-American forest landscapes has been changing and returning to it is no longer feasible or practical** in some places. The pattern and processes of forest ecosystems in a landscape are functions of the range of variability in disturbances, climate, and species movements that has occurred over a long period of time. The range of variability in these processes in the future

will determine how the current patterns and processes of ecosystems and species will develop. Managers can influence the future range of variability to achieve desired future ecosystem conditions for a landscape. The HRV can serve as a guide but not a target.

- **Depending upon the biogeoclimatic setting and physiographic region, historical and future ranges of conditions may be either strongly or weakly overlapping.** Future climate projections suggest that large differences between the past and the future tend to be associated with the extreme ends of ecological gradients, and ecotones may be particularly sensitive areas. For example, at forest and shrubland ecotone margins, and at subalpine-alpine ecotones, digital global vegetation models suggest significant changes, where life forms and plant community composition may suddenly shift after disturbances. Within the MMC forest, which falls within mid-montane and valley bottom environments, there may be only small differences in environments when comparing past and future range of variability, but the degree of difference will vary with topography and ecoregion.

Disturbances

- **Significantly altered disturbance regimes exist as a result of 150 years of Euro-American land use, altering the manner in which systems behave.** In general, mixed-conifer forests are on a trajectory leading to further divergence from a resilient condition.
- **Wildfires, along with insect, pathogen, and weather disturbances, historically did the bulk of the work to continuously restructure and recompose the historical landscape,** providing sustainable patterns of terrestrial and aquatic habitats, and supporting processes. Several forest insects (e.g., bark beetles and defoliators) and pathogens (e.g., dwarf mistletoe and root diseases) are notable disturbances in this region today and they are imparting influences at uncharacteristically high levels.
- **The disturbance ecology and resulting vegetation of the mixed-conifer type is influenced by the dominant disturbance regimes of the local landscape.** Historical fire regimes in dry and MMC forests differed across a broad range of spatial and temporal scales in response to local and regional variation in climate and weather, topography, soils, and fuels. Where dry mixed-conifer forests are a dominant feature within a landscape, the MMC forest often is influenced by more frequent fire. The converse is also true.

- **Small and medium sized fires (<1000 ha in size [2,471 ac]) were most numerous,** comprising 85 to 95 percent of the fires, but representing no more than 5 to 15 percent of the total area burned. **Larger fires (>2000 ha in size [4,942 ac]) were least abundant, but accounted for the majority of area burned.**
- **Low-, mixed-, and high-severity fires all occurred in dry and MMC forest but the relative portion of these fires varied across ecoregions.** Surface fire effects coming from low- and mixed-severity fires tended to dominate in dry forests, and local topography was especially influential during common events. Patches of high-severity fire resulted from rare or extreme weather and climatic events. Low-severity fires within MMC forest may have been more common in drier warmer ecoregions with less topographic relief. High-severity fires within MMC forests appear to be more common in cooler and wetter ecoregions and areas with stronger topographic relief. Low-severity fires occurred commonly in moist forests where they were intermixed with dry forests; high-severity fires in moist forests may have been more common where moist forests were intermixed with wet and subalpine forests.
- **We are in the early stages of anthropogenically induced climate warming in eastern Oregon and Washington, and we anticipate significant vegetation and disturbance impacts to dry and MMC forests during this century.** Over the 21st century, climate change impacts will increase in magnitude and extent, transforming some forests. Impacts to terrestrial ecosystems will include increased fire frequency, severity, and burned area, increased susceptibility to insects and diseases, and increased presence of invasive plant and animal species. Climate change will likely reduce or eliminate some subalpine habitats and snow cover that are essential to the survival of some wildlife species.

Vegetation

- **Fire exclusion, silvicultural practices, agriculture, livestock grazing, intensive road and rail development, and introductions of alien plant species** have significantly influenced the structure, composition, and disturbance regimes of many forests.
- **Dry and MMC forests represent a broad range of potential vegetation types** and include the grand fir, white fir, and Douglas-fir series. MMC forest falls within a gradient between the dry ponderosa pine and dry mixed-conifer forest and woodland types at the lowest end of the moisture gradient and subalpine and wet conifer forests at the upper end of the gradient. MMC forests are

often intermixed with dry mixed-conifer forests on the lower end and wet (e.g., western hemlock, western redcedar [*Thuja plicata* Donn ex D. Don], Pacific silver fir), or subalpine forests at the upper end of the gradient. The neighborhood a forest type lives in can influence the nature and extent of processes that are influential. We do not provide a precise definition of the mixed-conifer plant associations for different ecoregions or national forests. Agency ecologists are best suited to make the final decisions on MMC types occurring on each national forest.

- **A large number of successional pathways and old forest conditions historically existed** in the MMC forests of eastern Oregon and Washington as a result of the environmental complexity and disturbances that shaped them. Some old forests were the result of repeated low-intensity fire that maintained multicohort single-stratum condition of fire-tolerant species, while others developed to complex, multiple-strata structures that occasionally experienced mixed- or high-severity disturbances. Managers have options to shape the trajectories of vegetation patches and landscapes along these different pathways. However, targeting a single developmental stage as a management objective does not fit with variability of the mixed-conifer forest.
- **Fire- and drought-tolerant trees of medium diameter (about 41 to 64 cm [16 to 24.9 in]) and large diameter (more than about 64 cm [25 in]) are the backbone of wildfire- and climate-tolerant landscapes, and are an essential component of wildlife habitat.** The occurrence of older trees and older forests currently is far below the historical range of variability, despite their ecological importance.

Fire- and drought-tolerant trees of medium and large diameter are the backbone of wildfire- and climate-tolerant landscapes, and are an essential component of wildlife habitat.

- **Areas of ecologically diverse, early-seral grass, shrub, and seedling- or sapling-dominated forest patches are in short supply in some landscapes.** Historically, these areas would have been created and often maintained by mixed- and high-severity fires, complete with fire-killed snags and down logs. The largest fire-killed snags and down logs would have lasted a long time and provided needed plant and animal habitat. Grassland and shrubland patches were historically quite abundant on forested PVT settings, owing to variability in wildfire regimes and frequently reburned areas. It is likely, according to climate change projections, that grassland and shrubland conditions will be more common in forested PVT settings.
- **The assumption that plant association groups** (e.g., moist and dry mixed conifer) **can be used to infer historical disturbance regimes is only moderately true.** Limited studies suggest considerable variability in

disturbance regimes within and between moist and dry mixed-conifer types. They both contain components of the mixed-severity regime, which is a highly variable disturbance regime. Managers need a variety of information sources (e.g., known history, landscape context) in addition to plant association types when assessing the historical and future range of variability for a landscape.

- **In many areas of the mixed-conifer type, the density of large fire-tolerant trees has declined from historical levels largely because of past logging and recent high-severity fire.** The density of mid-sized trees may have increased; most of these are shade-tolerant species that reduce resiliency of the MMC forest.

Wildlife and Fish

- **The diversity and juxtaposition of patch types of differing composition and structure that were produced by historical mixed-severity fire regimes provided a variety of habitat resources to resident wildlife.** A key concept for understanding animal distribution and abundance in MMC forests is that animals need to acquire a variety of resources to meet their life-history needs, and most animals use a variety of different habitats and structural features to acquire those resources. The mosaic of conditions provided under historical mixed-severity fire regimes provided a rich variety of habitat features for native wildlife.
- **Different stand development stages provide different habitat structures and resources for wildlife:** Recently disturbed patches tend to support high plant productivity, and often exhibit a particularly high diversity of plant and animal species, including many specialist species.

  Even-age young forest patches generally support less diverse and abundant animal communities than are found in other stand development stages, but can provide important ecological functions, depending on specific habitat conditions and landscape context.

  Older forests with a diversity of tree ages and sizes (including large old trees) provide many unique ecological functions. Big old trees, living and dead, standing and down, provide unique and important habitat structures within old forest stands. The multilayer canopies found in older stands contribute to unique thermal characteristics within stands and provide structural complexity important many species.
- **Natural disturbance processes play a key role in the development of stand structure conditions and landscape patterns that determine habitat values for animals.** Endemic levels of disturbance and pathogens (including

dwarf mistletoe) can contribute to stand and landscape heterogeneity, while large-scale, high-intensity disturbances can reduce stand and landscape heterogeneity. Balancing the risk of long-term loss of stand structure and landscape heterogeneity with the retention of current ecological values often associated with the presence of forest pathogens and local high fuel loads is a fundamental challenge for land managers.

- **Biological legacies (including big old trees, snags, and logs) provide important habitat structures across all stages of stand development.** Within-stand structural diversity is generally much greater in stands with a variety of tree age and size classes, even if those stands are recently disturbed or dominated by younger trees. Big trees and logs are often a legacy of previous stand conditions that have been retained after some disturbance event, like fire or harvest. The variety of extremely valuable habitat structures provided by these large old trees include cavities, large branches, broken tops, brooms, and platforms.
- **Recent wildlife habitat assessments have reported a widespread reduction in habitat values relative to historical conditions for wildlife associated with early-seral, open forest, and postfire habitats.** Old forest habitat conditions show less decline at a regional scale, but patterns are variable across sub-basins. Owing to the variation in habitat trends across the interior Pacific Northwest, managers need to assess the trends in their local subwatersheds for appropriate guidance on wildlife habitat management needs.
- **Management activities focused on restoration of historical habitat patterns and stand structure conditions can have a variety of effects on wildlife populations.** At a stand scale, restoration treatments tend to favor those species associated with more open conditions and increased understory vegetation diversity, and have the potential to reduce abundance of species associated with closed-canopy, old-forest conditions. However, responses can be quite variable depending on the intensity of the treatment, and although the abundance of individual species can change, restoration treatments have generally not been shown to cause substantial changes in species diversity.
- **Loss of old-forest structures to large-scale, high-intensity wildfire is a concern for all old-forest species, including northern spotted owls.** Landscape conditions that are resilient to intermittent disturbances will be most likely to provide a shifting mosaic of old forest habitat conditions for all old forest species over time. Managing for patterns that continually bring stands into old forest conditions is also important because we cannot rely on existing old forest stands to remain in that condition over long time periods. This approach

of maintaining a shifting mosaic of old forest conditions is likely to be an increasingly important concept for old-forest wildlife conservation under future disturbance regimes that are anticipated under many climate change scenarios. Conservation of unique forest structures (particularly big old trees) is important, but it will be most effective if it is done in the context of sustainable landscape patterns.

Management Applications in the Field

- Ecological restoration has become a principle objective that drives current management on national forest system lands in the Western United States. It is defined as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. Ecological restoration focuses on **re-establishing the composition, structure, pattern, and ecological processes necessary to facilitate terrestrial and aquatic ecosystem sustainability, resilience, and health under current and future conditions.**” The scientific thinking on restoration has evolved as we learn more about land use change effects, climate change, paleoecology, and socioecological interactions (Hobbs et al. 2011). In using the term “restoration,” our intended meaning is that restoration involves restoring degraded ecosystems to a desired state considering historical and future ranges of conditions. Where the historical conditions strongly resemble the predicted future range of variability, they will be quite relevant. Where they do not, the focus will be on the future ranges. For example, the concept of socioecological resilience focuses on the adaptive capacity of species, ecosystems, and institutions rather than restoring or recovering a condition. In this report, we define restoration as the applied practice of renewing degraded and damaged landscapes, habitats and ecosystems with active human intervention.
- Rates of environmental change vary across eastern Oregon and Washington, and as a consequence some ecosystems have attributes that may still be similar to their HRV. In other ecosystems, however, **attributes and entire ecosystems are clearly out of their HRV**. For purposes of understanding and prioritization, it is important to assess the degree of landscape departure from the HRV. However, there is obvious importance for considering the future range of variability (FRV) and how that will influence vegetation. It is important to consider the dynamics of ecosystems that are possible as a result of climate change, changing land uses, and evolving social values. Projections or scenarios of FRV recognize the important implications of climate change on forests (Fettig et al. 2013).

Chapter 5—Management Considerations

In the midst of complicated social and political forces, forest managers make decisions that require the application of complex scientific concepts to case-specific project conditions. Decisions often must balance risks (e.g., elimination of fuels hazards vs. preservation of old-forest conditions) while acknowledging and allowing for uncertainties. Decisionmakers also must weigh tradeoffs associated with alternative courses of action to obtain multiple-use policy and land management objectives. We acknowledge this difficult task and the concurrent need to have and thoughtfully apply thoroughly the body of available scientific information.

It is not the role of the research community to direct management decisions, but it is appropriate to synthesize research and its core findings, and underscore key management implications of that research in specific management contexts. It is also the role of researchers to work alongside managers and seek to learn from their successes and failures. Here we provide considerations for management and emphasize that their application to local and regional landscapes requires the skill and knowledge of local practitioners to determine how best to apply them to each local situation they encounter, and its particular management history. Legacy effects matter and one size does not fit all.

One of the roles of researchers is to work alongside managers and seek to learn from their successes and failures.

In the following section we synthesize the principle scientific findings that have been gleaned from the body of scientific literature (summarized in chapter 4) as it pertains to management of moist mixed-conifer (MMC) forests. These constitute the "take home" messages that are intended to assist land managers.

Key Concepts to Consider in Management of MMC Forests

Resilience at regional scales is fostered by heterogeneity in patterns of local landscapes.

- **Resilience at regional scales is fostered by heterogeneity in patterns of local landscapes**, but not just any heterogeneity will do. Heterogeneity that is useful to maintaining resilience is that which is supported by the inherent variability of the local climate, geology, and disturbance regimes of each subregional locale. The variability of the regional landscape mosaic should be represented in unique patterns of local landscapes. Management for resilience at broad scales can promote this variability.
- **Overlap in the historical range of variability (HRV) and future range of variability (FRV) may be useful to defining desired local landscape patterns** (fig. 44). If landscapes reflect these overlapping envelopes of an eco-region, they are likely to be more resilient and conserve more future options for management. If all local landscapes look the same, the regional landscape will have been simplified. We make this suggestion because of high uncertainty about the use of the FRV alone.

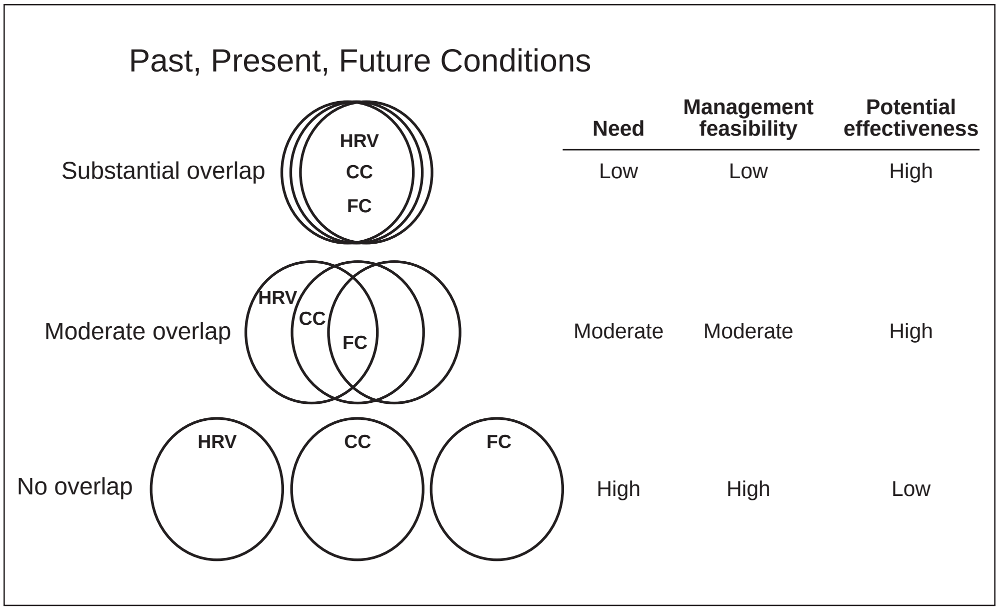

Figure 44—Conceptual illustration of the extent to which current ecosystem conditions (CC) in a place differ from the historical range of variability (HRV) in conditions and from projected future conditions (FC).

- **Managing for resilient landscapes that have patterns consistent with the range of variation under current and expected future disturbance regimes provides a conceptual starting point for capturing the habitat needs of native wildlife.** Native animals evolved in the context of historical disturbance regimes and resultant landscape patterns. Landscape resilience is also important to animal populations because it represents the tendency of a post-disturbance landscape to return to conditions to which the native animal community is adapted. Landscapes that are not resilient may shift into novel environmental conditions that can contribute to substantial changes in the animal community and loss of or change in biodiversity.
- **Some native wildlife species have been affected by threats not related to habitat amount and distribution,** including ecological interactions with invasive species and sensitivity to effects of specific human activities (e.g., livestock grazing or residential, highway, or forest road development). Conservation of these species often requires specific strategies that address habitat in combination with the specific risks to which they are sensitive.
- **Well-connected patterns of habitats across regional landscapes are important to sustaining diversity of native species.** A regional goal is to maintain interconnected networks of habitats that link ecoregions and subregions that would normally provide suitable habitats. Understanding how patterns on the

natural landscape (including vegetation communities and landforms) interact with human-created features (e.g., major highways and residential developments) to influence animal movement and population functions is also important for maintaining landscapes that can sustain populations of wide-ranging species over time.

- **When restoring within-patch heterogeneity, consider varying the tree clump and gap size distribution among patches.** If patches within local landscapes have the same structural characteristics, patch heterogeneity will have been simplified. Consider adapting methods like those developed by Larson and Churchill (2012) and Churchill et al. (2013) to the management of MMC forests. To do that, additional ecoregion-specific datasets will be helpful in fine-tuning innovative silvicultural and forest management strategies.
- **Second-growth 60- to 100-year-old forests offer substantial opportunities for restorative management.** Consider an emphasis on restoring patterns of successional stages that better match the variability of the inherent disturbance regimes. A large acreage of highly productive MMC forest, managed via regeneration and partial harvests during the 20th century, is in this general age class. Resulting forests are intermediate-aged and contribute to simplified landscape patterns.

Second-growth 60- to 100-year-old forests offer substantial opportunities for restorative management.

- **Consider restructuring large areas of the dry and MMC forest on the landscape to return to more resilient forest conditions.** After more than a century of forest and fire management, the current mixed-conifer landscape has developed more than a century's worth of successional inertia (compositional and structural change) and fuel accumulation. Wildfires and insect outbreaks are resetting conditions on the landscape; spatial allocation is driven by ignitions, fire weather, and fuels; changes are occurring suddenly; and the window of treatment opportunity for restoration is growing shorter. Consider identifying patches in key geographic, topographic, and edaphic locations that because of soils, aspect, elevation, and site climate are not likely to sustain these dense, drought- and disturbance-intolerant conditions. Likewise consider reestablishing the inherent landscape heterogeneity, using topography as the underlying template. Where early-seral species would naturally dominate (ponderosa pine [*Pinus ponderosa* Lawson & C. Lawson], western larch [*Larix occidentalis* Nutt.], and western white pine [*Pinus monticola* Douglas ex D. Don]) and natural regeneration is improbable (i.e., grand fir [*Abies grandis* (Douglas ex D. Don) Lindl.], white fir [*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.)], and Douglas-fir [*Pseudotsuga menziesii* (Mirb.) Franco] will likely dominate for a long time), consider regenerating these areas, as is appropriate to the topo-edaphic conditions.

- **To the extent practicable, consider reestablishing the inherent resilience capacities of ecological systems.** This can be done by restoring ecological processes as the foundation of ecosystem resilience. A keystone priority may be to reestablish the central role of the natural or inherent fire regime as the source of reinitiation, development, and maintenance of a heterogeneous landscape mosaic. Where feasible and socially acceptable, create management conditions that enable natural processes to do important work on the ground that is otherwise expensive and less effective to emulate with direct management. Doing this will be economically beneficial, contribute to fire and climate resiliency, and improve diversity of wildlife habitat conditions. Repeated treatments over time will be required to achieve such goals given the century's worth of successional inertia and fuel accumulation that has occurred in many areas.

Expect surprises and consider maintaining a broad range of management options. Expect rare events; they have always influenced the majority of the landscape.

- **Expect surprises and consider maintaining a broad range of management options. Expect rare events; they have always influenced the majority of the landscape.** Twenty-first century climatic changes will result in conditions that differ from the pre-management era, and differ from those we are currently experiencing. Managers and scientists will work to constantly improve the accuracy of future projections, but it may remain prudent to expect the unexpected. There is much uncertainty in the complex interactions between a changing climate, environments, disturbance regimes, and the diversity of organisms and communities in our ecosystems.

- **Consider providing a full complement of vegetation cover types and successional stages within mixed-conifer landscapes.** Patterns and variability of cover types and successional stages consistent with the inherent fire regimes may be desired. Managers can reduce the need for a more highly engineered and costly resilience (that is also more uncertain as to ecological and spatial outcomes) through managing to create or recreate a landscape pattern that would be formed by the patterns and variability of native disturbance regimes.

- **Manage with entire landscapes, all ownerships, and all vegetation types in mind.** Mixed-conifer and other forest types do not exist in isolation and are not confined to ownership boundaries. Disturbance regimes and habitat conditions of neighboring types are interconnected and interrelated, and are best addressed by working together with adjacent landowners.

- **Consider maintaining a backbone of large and old early-seral trees as a primary and persistent structure of all mixed-conifer forests.** If they have been removed by timber harvest, regrow them. This can be a key to restoring climate and wildfire resilience and reestablishing many core wildlife habitat values. Consider favoring ponderosa pine in the dry mixed-conifer forests, and western larch, western white pine, sugar pine (*Pinus lambertiana*), and Douglas-fir in MMC, consistent with local conditions.
- **Consider maintaining a backbone of large standing dead and down trees, and live deformed trees** (those with broken tops and limbs, cavities, and structures for nesting and denning sites) to provide structures for wildlife use. As a result of past timber harvest and "sanitation" treatments, these fine-scale structures are in short supply in most areas. Typically, it takes decades to centuries for a once healthy tree to develop the proper characteristics of such a "wildlife" tree, but techniques are available to create and accelerate their development. Information for both snag and down wood management in green and dead stands can be found in the Decayed Wood Advisor (DecAid) (http://www.fs.fed.us/r6/nr/wildlife/decaid/). Manage these structures as needed (i.e., remove if necessary) in developed recreation areas, such as campgrounds and trailheads, where they could pose a threat to human safety.
- **Old-forest refugia located in suitable locations may be important to retain.** Dense, multilayered mixed-conifer old forest patches are typically not sustainable in many landscapes of eastern Oregon and Washington unless they reside in environmental settings (e.g., valley bottoms and north-facing slopes) that enable them to persist (i.e., suitable topo-edaphic, climatic, and disturbance contexts) by escaping common disturbances. Likewise, focus restoration efforts on developing future old-forest patches in these areas, especially where old forest exists in short supply.
- **Consider reducing the density and layering in the neighborhood surrounding old-forest refugia, and favoring fire-tolerant species to help reduce the risk of loss of multilayered old and otherwise structurally complex forest patches to high-severity fires.** Loss of some existing complex structural patches is an inevitable but reasonable tradeoff for maintaining old forests in fire-prone environments.
- **Restoring the natural variability of wildfire regimes may be a key to restoring the physical processes that create productive aquatic habitats in streams.** Anadromous and cold water fish are fire-adapted species. The natural frequency, severity, and extent of historical fires governed the pulse of erosional

events that carried wood, rocks, and soil to streams. Native fish habitat can be greatly improved through restoration of the natural fire regime.

- **Consider encouraging the natural fire regime within the riparian zones of high-gradient (steep) stream reaches.** With the exception of low-gradient (<5 percent slope) often fish-bearing valley bottom reaches, riparian ecosystems adjacent to high-gradient streams commonly exhibited the fire regime of the adjacent upslope environments. Low-gradient (flatter) reaches are routinely distinguished as flood plain depositional areas whose riparian vegetation is dominated by hardwood tree and shrub vegetation and hydrologic disturbances, whereas the riparian vegetation of higher gradient streams is remarkably similar to the upslope vegetation.

Consider focusing restoration on large, functional landscapes rather than on individual landscape attributes, goals, or risks, such as fuels reduction, timber production, or owl conservation.

- **Consider focusing restoration on large, functional landscapes rather than on individual landscape attributes, goals, or risks,** such as fuels reduction, timber production, or owl conservation. Although we acknowledge the need for short-term measures to protect key habitats, species, and resources that are currently at risk, management emphases with singular resource objectives generally marginalize other important values and eventually fail because they lack this larger focus.
- **Give adaptive management serious consideration.** It simply formalizes learning from successes and failures. The new Forest Service Planning Rule offers yet another opportunity to implement adaptive management. We realize that many prior attempts to conduct adaptive management have failed, but it still represents a rich opportunity for learning, and it is how people learn, through structured trial and experience. Flexible management approaches employing evidence-based, scientifically credible methods promote learning.
- **Consider managing forests in the context of current and potential climate change.** Rates of climate change in the past century and projected for the remainder of this century differ spatially across eastern Oregon and Washington. In some geographic settings, changes in climate in recent decades have altered fire regimes, increased insect infestations, and resulted in high levels of tree mortality. Such changes linked to climate are likely to expand to larger portions of the region in coming decades. Consider identifying rates of change in climate and ecological response over the past century and those projected for future decades and setting management goals and treatments accordingly.

Example of Landscape Management: Okanogan-Wenatchee Landscape Evaluation Decision Support Tool

In eastern Washington, scientists at the Pacific Northwest Research Station Forestry Sciences Laboratory in Wenatchee and on the Okanogan-Wenatchee National Forest have developed and implemented a landscape evaluation and decision support tool (DST) that directly aids managers with implementing a comprehensive landscape restoration strategy. The landscape restoration strategy and the DST have been fully adopted on this national forest and are being implemented on all districts for projects that undertake restoration.

We acknowledge that this particular evaluation and decision support tool represents only one example of how landscape management can be executed. In this instance the forest had access to extensive data sets and skill levels not common on national forests. This example is offered to illustrate what is possible when the necessary resources and expertise are available.

The DST is an outgrowth of a long-standing joint research and management partnership, and a peer-reviewed, forestwide restoration strategy (OWNF 2012). Under the strategy, the objectives of landscape evaluations are to:

- Transparently display how projects move landscapes toward resilience to drought, wildfire, and climate;
- Define and spatially allocate desired ecological outcomes (e.g., adequate habitat networks for listed and focal wildlife species [Gaines et al., in press] and disturbance regimes consistent with major vegetation types);
- Logically and transparently identify potential landscape treatment areas (PLTAs), treatment patches, and the associated rationale; and
- Spatially allocate desired ecological outcomes from landscape prescriptions, and estimate outputs from implemented projects.

Landscape evaluations under the strategy assemble and examine information in five topic areas: (1) current patterns and departures of vegetation structure and composition from historical and climate change reference conditions; (2) spread potential for wildfires, insect outbreaks, and disease pandemics across stands and landscapes given local weather, existing fuel and host conditions; (3) damaging interactions among road, trail, and stream networks; (4) wildlife habitat abundance, distribution, and sustainability; and (5) minimum roads analysis (i.e., which of the existing system roads are essential and affordable, and which are not) (fig. 45).

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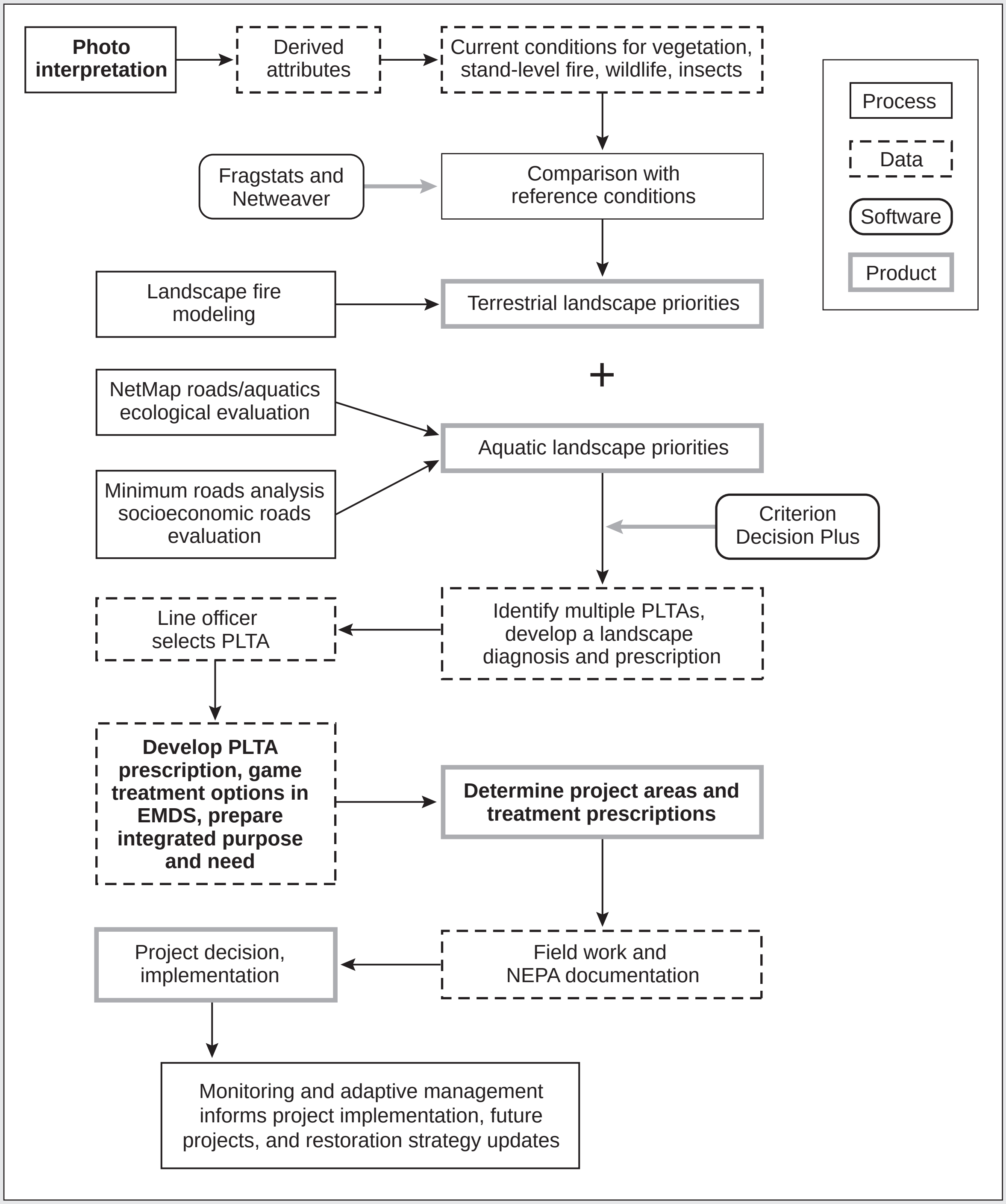

Figure 45—Workflow diagram of the Okanogan-Wenatchee National Forest Decision Support Tool (DST). The DST was implemented in EMDS, the Ecosystem Management Decision Support software (Reynolds and Hessburg 2005, Reynolds et al. 2003) using the NetWeaver and Criterium DecisionPlus software utilities. FRAGSTATS (McGarigal et al. 2002) spatial pattern analysis software was used to characterize spatial patterns of vegetation composition and structure in current subwatersheds and for the historical range of variability and future range of variability reference conditions. FlamMap and Randig (Ager et al. 2007, Finney et al. 2007) were used to simulate probable landscape scale wildfires. PLTAs are potential landscape treatment areas that emerge from landscape evaluations.

The restoration strategy is a living document, which is updated and expanded to include new relevant science and technology. Likewise, with each implemented project, new utilities are added to the decision support tool. Landscape evaluations in the DST examined current vegetation spatial pattern conditions and quantitatively compared them with both historical (past 1900s-era climate, the HRV) and future, mid-21st century climate reference conditions (the FRV), which represent plausible ranges of pattern variability for these periods. Current vegetation conditions were established via interpretation of stereo pairs of aerial photos using standard aerial photogrammetry techniques. The samples for each of the reference conditions were developed using the same vegetation attributes and these same photogrammetry techniques. Evaluated areas were typically small groupings (two or more) of modern-era subwatersheds (12-digit HUCs).

Once current vegetation conditions were interpreted and field verified, the most significant departures of local landscape pattern conditions from reconstructed early 20th century reference conditions (the HRV) were evaluated in the DST. These reconstructed conditions were obtained from the Interior Columbia Basin Ecosystem Management Project (ICBEMP) mid-scale assessment dataset (Hessburg et al. 1999a, 2000a). Since that project, these reconstructed conditions have been summarized for each of the major ecological subregions (Hessburg et al. 2000a) on the two national forests. After the initial departure analysis, departure from a second set of future warmer and drier climate change conditions (FRV) was evaluated. The FRV conditions were represented using historically reconstructed conditions of an ecological subregion whose climatic conditions best match those predicted under warming. This approach used what are often referred to as “climate change analogue conditions” and was intended to show the range of patterns that would emerge with climatic warming and the associated biotic conditions and disturbance regimes. The results of the two departure analyses were used together with equal weighting to determine the most significant changes in forest vegetation structure and composition in the evaluated subwatersheds (Hessburg et al. 2013). Equal weighting was used when considering the HRV and FRV conditions to maximize management options in future landscapes because of the scientific uncertainty surrounding predicted climate change predictions.

Reference conditions were developed for each unique ecological subregion to reflect the range in patch sizes and successional pattern conditions that are

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typical for the biota, climate, geology, and disturbance regimes of a physiographic region (Hessburg et al. 2000a), whether they occurred before the advent of management, or they occur at some time in the future. Results of departure analyses were reported by potential vegetation type (PVT) to provide managers with a mechanism for establishing on-the-ground treatments in response to departure analysis. Departure analysis results can inform development of landscape-level restoration prescriptions that are designed to specifically address the most significant departures. Resulting prescriptions can help restore landscape vegetation conditions that are better adapted to wildfires, native insects, pathogens, and predicted climatic warming.

In addition to the vegetation departure analysis, departure analyses were implemented for habitats of regionally listed and focal wildlife species (Gaines et al., in press), and for patch scale fire behavior attributes under a 90^{th} percentile wildfire burn scenario (Hessburg et al. 1999a). Departure analyses were also completed that reflect landscape changes in vulnerability to a variety of major insects and pathogens using models developed for the ICBEMP project (Hessburg et al. 1999a, 1999b, 2000a).

A landscape-scale wildfire analysis was also completed and programmed into the DST, which characterizes the most probable wildfire flow across the large ecoregional landscape when typical wildfire season wind direction, wind intensity, terrain routing of wind and wildfire, and thousands of randomized ignitions are considered. The landscape-scale wildfire analysis provided concrete map depictions of the anticipated flow of large wildfires across the landscape. Managers can use these maps to determine where wildfire flow may be interrupted via strategically placed fuels treatments to reduce the likelihood of large events (Hessburg et al. 2013). Local landscape evaluations can also incorporate these insights into their landscape-level prescriptions.

New in 2013–14, aquatic landscape evaluation utilities are being added to the DST. Under construction are the addition of a minimum roads analysis subroutine, stream crossing and culvert subroutines (currently handled external to the DST), and a module designed in NetMap (Benda et al. 2007) or similar software that identifies road segments that constrain stream channel migration and natural connection with the historical flood plain, separate existing channels from off-channel features, and contribute the most to in-stream sediment. For additional details about the entire landscape evaluation process, see Gaines et al. (in press) and Hessburg et al. (2013).

Stand Management—Silvicultural Tools and Their Role in Landscape Management

Achieving landscape-level management objectives will require making explicit connections between patch- or stand-level and landscape-level assessments, especially among patch interactions within landscapes. We use the term "stand" to conform to the familiar nomenclature of silviculture. The term "patch" is a similar term used by some to describe a group of trees (O'Hara and Nagel 2013). The planning and operations conducted by land managers will achieve the broader objectives when making this kind of nested spatial relationship explicit in their entire planning process. Stand-level management is therefore the operational level to meet landscape objectives. Creating stand structures that meet these large-scale objectives will be a central concept to management of eastside mixed-conifer forest and riparian ecosystems. Silviculture represents the operational process and offers a matrix of options to achieve these broad-scale objectives.

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Reducing stand density, altering species composition, and enhancing stand and landscape complexity, where appropriate, will be the central silvicultural objectives in many eastside landscapes based on our findings and conclusions. Stand densities are too high in most locations and ecological resilience is generally lacking owing to past management. For example, logging removed a disproportionate number of large early-seral dominant trees and favored simpler stand structures where dense stands of shade-tolerant trees homogenized both stands and landscapes. Land managers are advised to evaluate the fire risks of both current conditions and what might have developed under active fire when considering how to reconstruct forest structure. Most of the ponderosa pine, and the dry and MMC forested landscapes of eastern Oregon and Washington, had a relatively open stand structure under the influence of active fire (for more information, see "Wildfire" and table 5 in chapter 2). Nonetheless, stands had a particular horizontal and vertical complexity that was characterized by a dominant matrix of relatively open canopies with tall fire-tolerant trees, canopy gaps with patches of tree regeneration or shrubs, and dense patches or scattered individuals of shade-tolerant trees. The relative proportion of these fine-scale stand and landscape elements varied with environment and disturbance regimes.

In dry ponderosa pine and dry mixed-conifer types, stands on southerly aspects were relatively open with a few dense patches of shade-tolerant Douglas-fir and grand fir. Tree regeneration and cohort development coincided with wildfires, but frequent fires killed the majority of the regenerating pines, such that relatively few survived from fire to fire, thereby yielding the appearance of many cohorts through time. In wetter areas, valley bottoms, and north aspects with longer fire return intervals, the MMC mosaic would have had more dense patches with shade-tolerant understories or larger canopy gaps created by active or passive crown fire associated with mixed- or high-severity fire.

Restoration goals that achieve both landscape and stand objectives can be met by directing landscape units toward the species composition and diversity in age class structure, and complexity in vertical and horizontal stand structure, that are resilient in the face of fire, insects, disease, and climate change. In many cases, promoting the kind of horizontal heterogeneity of stands that occurred prior to Euro-American settlement will greatly increase resiliency to large-scale disturbances such as insect outbreaks (Fettig et al. 2007, Franklin et al. 2013, Hessburg et al. 1994, O'Hara and Ramage 2013). For many landscapes, development of a multiscale mosaic of age and size structures of fire and drought-tolerant species will be the objectives.

Another key objective, particularly in some stands where fire risk is acute, includes reducing fuels and associated potential for large-scale fires. This involves simplifying some stand structures to reduce ladder fuels or to modify understory vegetation and other fuels. Finally, some stand structures can be modified to restore old-forest single-stratum structures (e.g., O'Hara et al. 1996) where scattered overstory trees dominate and other vegetation is relatively scarce. This condition would have been common in many dry pine and dry mixed-conifer forests but also some MMC forests prior to settlement and management. These old-forest structures may be considered simple compared to contemporary stands where historical fire suppression has favored development of multiple strata, but these simple structures also exhibited a fine-scale heterogeneity of size, age, and occurrence, in clumps of varying size, and among gaps with varying size, which made these patches far more resilient to fire, insects, and drought. The result is that over the entire eastside mixed-conifer zone, and even in individual watersheds, some area (patches, stands, or small landscapes) may be directed on trajectories toward complex structures (e.g., a mosaic of open canopies and dense canopies characterized by multi-storied shade-tolerant tree species that are needed for some objectives such as northern spotted owl habitat) while the majority of others may be directed toward more open structures with a subtle but distinctive spatial, compositional, and age heterogeneity. Landscape-level planning will provide the guidance for these decisions.

Silviculture can encompass both passive and active approaches. A variety of treatment options are available to manage stands so that they develop toward desired structures. A central management objective is implementing silvicultural activities (including both mechanical treatments and prescribed burning) that prepare areas for expected wildfire, to restore stand structures, or alter structures. Treatment options contain a wide range of methods including those that lie between traditional even-aged methods and uneven-aged methods (O'Hara 1998). Desired stand structures may be highly variable across different forest types and the treatments to direct stands toward these structures should also be variable. Implementing forest and riparian management strategies that promote the needed heterogeneity of mixed-conifer forests (dry, moist, and wet) across the landscape will require willingness to innovate on the part of land managers. There may be periods during stand development when the treated forest may not meet, or appear to meet, longer term objectives.

Stand management will often involve social or ecological tradeoffs. Some restored stand structures may not be popular with some stakeholder interests when they result in dramatic reductions in trees or cover. Additionally, many stands may have reached a state where they are prone to wind damage and cannot be manipulated with partial harvest or light thinning treatments, in which case more intensive harvest may be needed to "reset" the ecological trajectory. In other cases, prescribed burning cannot be used without substantial alterations of existing fuels. In these situations, silvicultural treatments can facilitate subsequent prescribed burning treatments. The combination of extensive areas needing treatment, air quality constraints, and work-force requirements presents enormous challenges to land managers, especially at the landscape scale (North 2012, North et al. 2012). Managers are encouraged to view their work as a series of incremental steps intended to move systems in the direction of restoring resilience of mixed-conifer forests to fire and other disturbances. Restoration will require more steps in some places than others.

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There are opportunities for silviculture to be more cost efficient by recognizing the potential for passive treatments to achieve objectives under certain circumstances. Passive silviculture relies on natural processes including wildfire and regeneration to achieve target stand structures. For example, mixtures of tree species with differential growth rates can result in complex, stratified stand structures even though the stands are even-aged. The rapid growth of western larch and lodgepole pine (*Pinus contorta* Douglas ex Loudon) in comparison to grand fir, evident in many mixed-conifer areas in eastern Oregon and Washington, is one example (e.g., see Cobb et al. 1993).

Another cost-saving strategy involves offsetting expenses of active treatments through commercial activities that help maintain local industries. Active components of silviculture include mechanical removal of low- and mid-story vegetation to restore a stand dominated by the largest early-seral trees. These restoration treatments will often involve treatments that result in removal of some commercially valuable trees. To meet the ecological objectives, the commercial volume could come from larger diameter, shade-tolerant species. Ideally, the byproduct of landscape restoration is wood fiber from ecologically undesirable components of stands that can help offset treatment costs, which can sometimes be quite high.

Silvicultural treatments of stands will be more effective when directly tied to a landscape analysis and prescription, and tailored to resolve many concurrent issues. Both single- and multi-aged stand structures are naturally part of eastside forest landscapes. Single cohort structures often result from stand-replacement fires, which can be structurally mimicked by several types of silvicultural techniques, if a significant large dead wood component is provided. However, there is no direct substitute for fire and its effects (McIver et al. 2012, Stephens et al. 2012). Fine-scale multi-aged conditions, on the other hand, were much more common in dry ponderosa pine, dry mixed conifer, and MMC forests, especially where surface fire effects dominated, and fires were relatively frequent. Multi-aged stand management at a variety of scales offers a means to create complex structures and emulate the structural complexity found in forests subject to mixed-severity fire regimes. However, traditional uneven-aged management methods are generally inappropriate to achieve these ends. These tools focus on achieving certain diameter frequency distributions rather than meaningful structural characteristics such as numbers of canopy strata, crown coverage, or light penetration for regeneration (O'Hara 1996, 1998; O'Hara and Valappil 1999). These tools also tend to encourage stand structures that are constant from one location to the next and over time.

Instead, managers can focus on retaining meaningful structures that build on existing structural features that vary from site to site and where patchiness of cohorts within stands may vary from fine (groups of trees) to rather coarse (acres to parts of acres) grains. For example, some large and old ponderosa pine and other large young forest dominants would ideally be retained on sites where they are scarce, regardless of the effect on a diameter distribution curve. Simpler two-aged stands offer the opportunity to form structures that fit historical disturbance regimes, retain some cover at all times, and provide structural diversity within stands and across landscapes. The multi-aged approaches in eastside forests may therefore not resemble traditional uneven-aged silviculture (e.g., O'Hara and Gersonde 2004). Instead, guides with a high level of flexibility will be needed to encompass a wide range of stand structural features. Examples of flexible prescriptions and stand structures are available for ponderosa pine (O'Hara et al. 2003) and lodgepole pine (O'Hara and Kollenberg 2003).

In some MMC forests where native disturbance regimes previously created patches of high-severity fire and, in turn, patches of early-seral grass, shrubs, or forest, the wildlife habitat associated with early-seral conditions may be lacking. This may be due to a lack of stand-replacement disturbance or because recent timber harvest has consisted of partial removal treatments that consistently left some of the more dominant trees. In situations where early-seral structure is clearly underrepresented, treatments targeted at creating early-seral grass, shrub, seedling, sapling, and pole-sized tree patches within the forest may be needed, complete with large snags and down logs, as appropriate. Variable retention systems that leave residual trees after harvest (e.g., Beese et al. 2003, Mitchell and Beese 2002) are one way to develop these stand structures.

Existing stands without desired variability in stand structure may be managed to enhance stand-level variation with variable-density thinning (VDT) and other similar methods. VDT is used to enhance variability in homogeneous stands by thinning to a range of densities within a single stand (Carey 2003, O'Hara et al. 2012). This general concept can be applied to any stand to any degree where increased variability is an objective (e.g., see Harrod et al. 1999). However, generalized VDT protocols do not exist because these protocols are usually not transferable from one stand to another, so prescriptions will be needed for individual stands.

Systematically achieving variability in managed forest landscapes is a major change from past practices because traditional silviculture emphasized homogeneity rather than heterogeneity.

Systematically achieving variability in managed forest landscapes is a major change from past practices because traditional silviculture emphasized homogeneity rather than heterogeneity. The challenge will be finding ways to systematically integrate variability into stand management. Recent efforts are making meaningful progress toward development of operational marking rules and procedures for implementing VDT in a variety of vegetation types (O'Hara et al. 2012). Larson and Churchill (2012) cautioned that typical global pattern analysis ignores within-pattern variation that is often the key feature of forest restoration. They recommended maintaining mosaic structures within patches that often exist at quite small spatial scales (<0.4 ha [1 ac]) and developing marking guidelines that focus on creating or maintaining individuals, clumps, and openings (ICO) as key structural elements (Churchill et al. 2013, Larson and Churchill 2012) (fig. 46). This approach has been employed recently and demonstrates the innovation that is emerging in forest management to achieve these varied landscape objectives (Churchill et al. 2013, Franklin et al. 2013, Stine and Conway 2012). More extensive datasets are needed from additional physiographic regions and potential vegetation type settings to better understand the variability in within-patch clumpiness that can inform marking guidelines and silvicultural prescriptions throughout the mixed-conifer forest.

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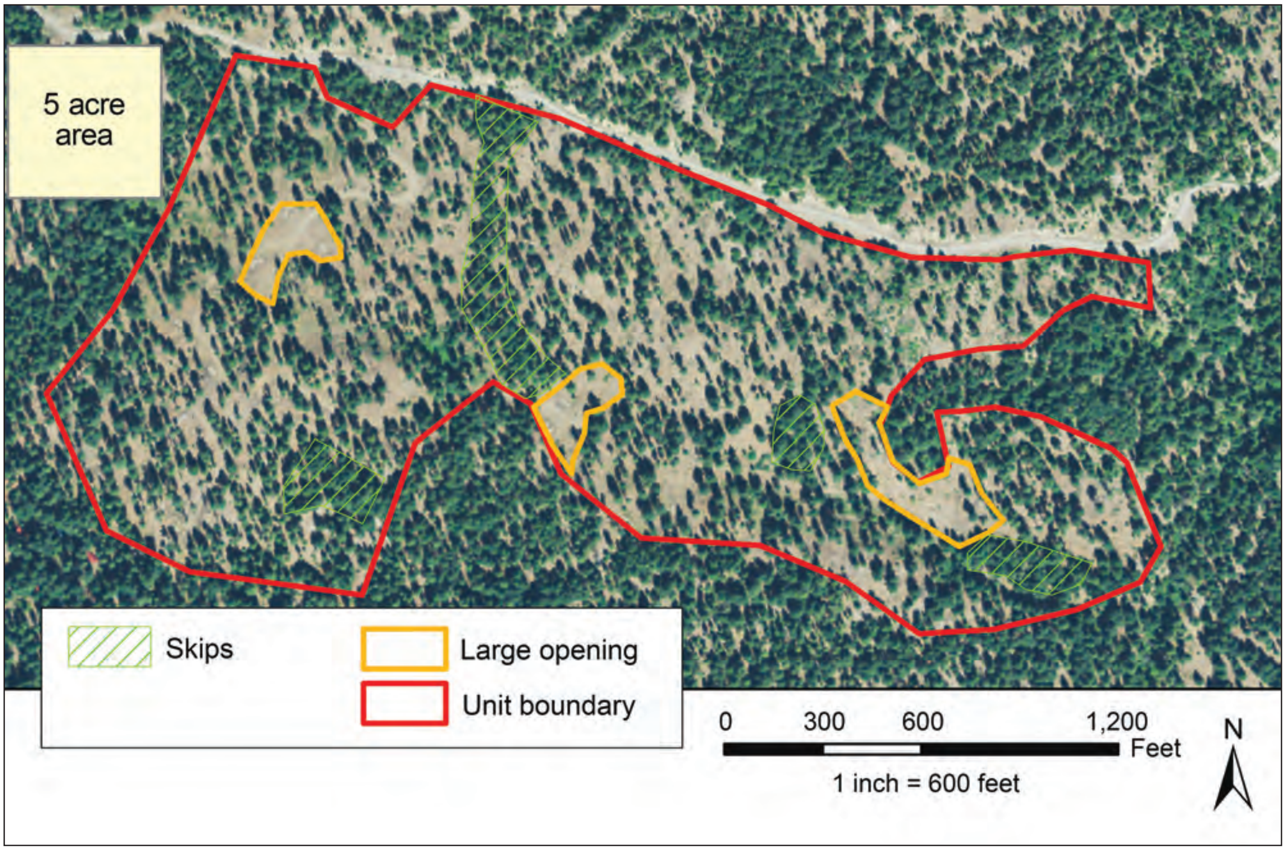

Figure 46—Example of a 32-ha (80-ac) ponderosa pine and Douglas-fir treatment from the Okanogan-Wenatchee National Forest. The treatment was designed to restore a pattern of individual trees, clumps, and openings consistent with prefire exclusion conditions. The prescription included specific guidelines for retentions of clumps of different sizes, as well as openings (Churchill et al. 2013). Large openings were flagged out during layout and were often sinuous in shape. Smaller openings occurred as a result of leaving clumps. Large skips (0.2 to 1.2 ha [0.5 to 3 ac]) were included to protect a riparian buffer in the middle of the unit and other biological hotspots such as large Douglas-fir with dwarf mistletoe, concentrations of downed logs, and multistory patches. Note how the unit boundaries were used to create an additional "finger" skip that extends into the stand. A 2-ha (5-ac) square area is shown to indicate the approximate size of an area that should contain a skip or large clump to break up siting distances. For more details on this approach, see http://www.cfc.umt.edu/forestecology/files/ICO_Manager_Guide_V2.pdf.

In the case of the diverse mixed-conifer forests of eastern Oregon and Washington (dry, moist, wet) the ICO method identifies important components of heterogeneity at fine spatial scales that should be considered in any silviculture prescription. Knowledge about the landscape context, disturbance history, environment, and other management goals will need to inform how the ICO goals are set. For example, in MMC where grand-fir invasion has created large areas of dense canopies, a stand-level silvicultural prescription may be focused first on creating relatively open conditions by removal of most grand fir stems (subject to other management objectives) and second on the spatial patterning of individuals, clumps, and openings in the remaining early-seral species-dominated stand. These considerations may be sequenced or concurrent, depending upon the particular vegetation conditions, and other relevant contexts.

Offsite or nonlocal seed sources were commonly used to replant many plantations in previous decades. These stands frequently exhibit poor survival and growth. Individual trees that are clearly maladapted to a site can be removed, but the presence of these trees should not necessitate special operations to remove them. Often such trees are not producing either pollen or seed and their presence may be of limited ecological importance.

Salvage treatments are an option to produce some timber from stands where trees have been killed by fire, insects, or pathogens. In addition to potentially providing wood resources to local communities, salvage treatments can mitigate future damages from wildfires, especially if salvage removes the smaller trees and associated fine fuels. However, there is typically little ecological justification for salvage of dead trees (Lindenmayer et al. 2008, Spies et al. 2012). Treatments to remove fire- or insect-killed trees are best focused on first achieving residual stand structural goals determined through landscape analysis including retention of snags and downed logs for wildlife habitat considerations (O'Hara and Ramage 2013). Additionally, salvage treatments can be a necessary safety measure in developed recreation areas, near communities, and along roads to reduce hazards.

Upper diameter limit restrictions on tree harvesting have been implemented to improve retention of larger trees on some public lands (e.g., OWNF 2012). Conservation of remaining large and old trees makes sense in many ecological settings where these structures are in short supply, especially where remaining old trees are fire-tolerant, shade-intolerant species. However, strict application of this restriction can prevent managers from having flexibility to achieve ecological objectives. For example, Abella and Covington (2006) found that in Southwestern ponderosa pine forests, upper diameter limits hindered abilities to achieve ecosystem goals and instead served primarily as an economic constraint. Likewise, in some forest patches, large and old trees may be more susceptible to supporting and sustaining bark beetle infestations (Fettig et al. 2007), so some discretion is needed.

Similar constraints exist in eastside forests where these arbitrary limits on harvested tree diameters lead to an unsustainable abundance of large, shade-tolerant trees that actually impede regeneration of shade-intolerant and fire-tolerant trees. This happens on relatively productive sites where shade-tolerant trees, established after the advent of fire exclusion, have grown to relatively large diameters that exceed these arbitrary limits. The presence of such trees may also prevent restoration of openings or a patchy heterogeneity in stand structure. Many of the post-harvest stands contain large (more than about 53 cm [21 in] diameter at breast height) young trees with species compositions and stand structures that are not representative of the native forests that we might expect to be resilient to drought stress, beetle infestation, changing climate, and other ecosystem stressors. In those situations, active management that may include the harvest of some medium- and large-size trees may produce ecologically enhanced stand structure and species compositions. Management options in these young, productive stands should take into account historical stand structure and species composition, using a landscape-scale context to help ensure that project implementation fits into overall restoration and desired future conditions. That said, large and old trees, particularly of shade-intolerant and fire-tolerant species, are essential elements to include in any eastside landscape. However, strict age or size limits on tree harvest that are not sensitive to site conditions, disturbance history, and topo-edaphic settings can hinder some restoration efforts and may reduce resiliency. Rules of thumb provide helpful guidelines but departures from these may be allowed with well-reasoned explanations.

Chapter 6—Social Agreement and Institutional Capacity for Restoring Moist Mixed-Conifer Forests

The social agreement and institutional capacity for restoring moist mixed-conifer (MMC) forests is every bit as important as the scientific foundation for doing so. The ability to institute the kinds of management changes managers will consider is directly a function of the capacity of the entire affected community to form working partnerships and a common vision. Thus, this section focuses on (1) the social acceptability of different types of restoration treatments among the general public; (2) collaboration as a means for developing, broadening, and sustaining social agreement around specific restoration projects among local stakeholders; and (3) some ideas for how the Forest Service and its partners could approach restoration on national forests that would enhance the agency's institutional capacity to implement the kinds of management strategies discussed in this synthesis document.

The social agreement and institutional capacity for restoring moist mixed-conifer forests is every bit as important as the scientific foundation for doing so.

Social Acceptability of Restoration Treatments Among Members of the Public

Public acceptance is a critical factor influencing whether or not an agency like the Forest Service can carry out its forest management goals (Shindler 2007). It is generally not enough for decisions about forest management to be scientifically sound and economically feasible; they must also be socially acceptable (Shindler 2007). Scientific evidence clearly attests to the need for forest restoration, as this science synthesis demonstrates, and most people living in the wildland-urban interface express a high level of support for reducing fuels on public lands that exhibit high fire risk (McCaffrey et al. 2013). Research from Oregon and Washington also finds that many members of the public recognize the need for restoration treatments on federal lands because of poor forest health and high fire risk (Abrams et al. 2005, Brunson and Shindler 2004, Shindler and Toman 2003). Nevertheless, there is often social disagreement about the methods used to accomplish these treatments, and the locations where they should be carried out. Furthermore, public acceptance of fuels reduction treatments can vary by both geography and social group (Raish et al. 2007, Shindler 2007). For example, a study of nine different stakeholder groups in Oregon found that they were much more supportive of active management of lower elevation ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) forests than of upper elevation mixed-conifer forests, where the perceived need for treatments was controversial (Stidham and Simon-Brown 2011). This finding suggests that science-based planning—in which the ecological need for fuels reduction is demonstrated by a solid foundation of scientific evidence—is important for improving the social acceptability of restoration treatments in the mixed-conifer zone (Stidham and Simon-Brown 2011).

Social acceptability of forest restoration activities is greatest when the perceived risk of wildfire is high, forest health is believed to be poor, and proposed treatments are perceived as being cost effective and successful at achieving desired outcomes (Bright et al. 2007, Brunson and Shindler 2004, Winter et al. 2002). There is less support for management actions that are perceived as being costly, producing long-duration smoke, posing a risk of escaped or catastrophic fire, or reducing the aesthetic qualities of the landscape (Brunson and Shindler 2004, Winter et al. 2002). Moreover, when citizens believe that forest restoration and fuels treatments are likely to have positive outcomes—economic, ecological, or social—they are more likely to be supportive of them (Shindler 2007).

This science synthesis suggests that both mechanical thinning and the use of prescribed fire will be needed to restore the resilience of MMC forests. In general, there is social support for both types of treatments, though prescribed fire has been found to be less socially acceptable than mechanical thinning in some places (Abrams et al. 2005, Brunson and Shindler 2004). Regarding mechanical treatments, one survey of Oregon and Washington residents (Abrams et al. 2005) found that 88 percent of respondents supported the selective thinning of overstocked forests, and 50 percent supported selective thinning of healthy forests. There was virtually no support for clearcutting. Another study of public acceptance of mechanical fuels reduction treatments from Oregon and Utah (Toman et al. 2011) found a high level of support for mechanical thinning among survey respondents (83 percent), followed by mowing understory vegetation (68 percent).

The social acceptability of prescribed fire use appears to increase when there is confidence in those carrying out the treatments; when it is conducted in remote areas away from development; when it is implemented in a reasonably sized area and resources are present to assure its control; when people are knowledgeable about fire and fuels management and prescribed fire treatments; when people believe it will be effective in producing desired outcomes; when it is cost effective; when mitigation measures are taken to reduce negative air quality and aesthetic impacts; and when stakeholders are involved in planning and preparing the treatments (Bright et al. 2007, Brunson and Shindler 2004, Nelson et al. 2004, Ostergren et al. 2008, Shindler and Toman 2003, Toman et al. 2011, Winter et al. 2002). Ideological perspectives may also inform opinions about the use of fire as a restoration treatment. A study from Montana found that landowners who valued forests as working landscapes that produced economic value disapproved of wildland fire use and prescribed fire because they result in "wasted" timber resources (Cacciapaglia et al. 2012). These landowners also saw fire as a detrimental or unnatural force that needed to be suppressed. They tended to view mechanical thinning in a positive light, however. In contrast, almost all of the landowners who valued land for its naturalness and ecological values supported wildland fire use (in wilderness) as well as prescribed burning, as appropriate, citing the regenerative properties of fire.

Trust and communication are key components of social acceptability. Researchers have found that the greater the trust in an agency to implement fuels treatments, the more supportive people are of their use (Olsen et al. 2012, Toman et al. 2011, Winter et al. 2002). In addition, the more informed people are about fuels management practices, the more supportive they are (Brunson and Schindler 2004). Activities that build trust between agencies and the public and increase communication and knowledge about fuels treatments are important. Communication and outreach efforts that emphasize the benefits of mechanical thinning and prescribed fire treatments for improving forest health and reducing fire risk to local residents are more likely to increase public acceptance of these treatments (Ascher et al. 2013). Highlighting the amount of control forest managers have over prescribed fire may also help increase support for it. Creating pilot demonstration projects in the places and forest types where restoration activities would be located is another approach to increasing support for them (Stidham and Simon-Brown 2011). Finally, developing fuels reduction and restoration activities through collaborative processes that include stakeholders in planning, decisionmaking, and partnerships (discussed below) is an important approach for overcoming social disagreement and lack of trust (Becker et al. 2011, Hjerpe et al. 2009, Stidham and Simon-Brown 2011, Sundstrom et al. 2012).

Shindler (2007) identified five strategies that forest managers can pursue in order to increase public acceptance of forest management activities: (1) view public acceptance as a process that evolves through building understanding, exchanging and discussing ideas, and evaluating alternatives; (2) develop agency capacity to respond to public concerns; (3) recognize trust building as the goal of public communication and outreach; (4) be sensitive to the local social and community context in which fire management activities are to be carried out, and how these activities may affect communities and forest users; and (5) encourage stable leadership so that shared understanding of forest conditions and fire management practices can develop, and so that strong agency leadership is evident in agency interactions with the public. It is important to remember, however, that social acceptability is dynamic in nature, and can change as people learn more, or when external variables shift (Shindler 2007). Thus, it is important for managers to pay continual attention to public values associated with forest restoration.

Collaboration

Decisionmaking processes play an important role in influencing public acceptance of fire and fuels management activities (Shindler 2007). One of the major constraints to increasing the rate and scale of forest restoration on national forest lands in eastern Oregon is the capacity to reach social agreement about how to achieve it (Economic Assessment Team 2012). The same is likely true in eastern Washington. The Forest Service has emphasized collaboration as a means for developing the social agreement needed among diverse stakeholder groups to carry out forest restoration projects, as demonstrated by recent investments in the Collaborative Forest Landscape Restoration Program and language in the 2012 Forest Service planning rule. Collaboration can be defined as "an approach to solving complex environmental problems in which a diverse group of autonomous stakeholders deliberates to build consensus and develop networks for translating consensus into results" (Margerum 2011: 6). Consensus can range from a simple majority to unanimous agreement regarding a decision, but usually means reaching a decision that everyone can live with. Collaboration in identifying approaches to forest restoration in MMC forests is particularly important because these forests have relatively high economic value in addition to high recreation value, and the methods needed to restore them may be complex and controversial.

There are many models for collaboration associated with forest and fire management on national forest lands (reviewed in Charnley et al. 2013); the best model will depend on local context and the nature of the issues. A number of community-based collaborative groups have formed around eastside national forests to address the ecological and economic issues associated with forest restoration, and to help implement solutions. In eastern Oregon, these include Blue Mountains Forest Partners, Umatilla Forest Collaborative Group, Wallowa-Whitman Forest Collaborative, Harney County Restoration Collaborative, and Ochoco Forest Restoration Collaborative. In eastern Washington, they include the Tapash Sustainable Forest Collaborative and the Northeast Washington Forestry Coalition.

Research has addressed what is needed for successful community-based collaborative processes. For example, McDermott et al. (2011) identified three sets of features that contribute to successful collaboration. The first factor is external sources of support. This includes the support and involvement of elected officials, agency leaders, and key decisionmakers; enabling laws and policies; and active community involvement. The second factor involves having access to sufficient resources to enable meaningful engagement of the many entities, such as funding, staffing, and information. The third factor involves the capacity to perform, which depends on effective leadership, trust, and social capital.

Bartlett (2012) laid out a logical and promising pathway for collaboration based on the collaborative process used to reach stakeholder agreement about hazardous fuels reduction projects at Dinkey Creek, on the Sierra National Forest in California. Successful collaboration there was based on a process that included five main stages: assessment, organization, education, negotiation, and implementation (see Bartlett 2012 for a description of these stages). Her experiences from California provide some key insights about what promotes effective collaboration:

- Include a broad range of participants;
- Establish a common conceptual framework, purpose and need, and long-term desired condition;
- Include scientific experts who serve as technical resources during meetings;
- Move some intractable issues forward without complete consensus if necessary;
- Include site visits to support decisionmaking and reach agreements; and
- Have an impartial mediator to promote trust and problem solving.

Other suggestions that seem to have a positive effect on collaboration include timely engagement, building trust, and developing patience with the process.

The organizational capacity of collaborative groups is critical for sustaining effective collaborative processes. Cheng and Sturtevant (2012) provide a framework for assessing the collaborative capacity of community-based collaboratives to engage in federal forest management. They identify six arenas of collaborative action: organizing, learning, deciding, acting, evaluating, and legitimizing. Within each of these arenas there are different kinds of capacities associated with individuals, the collaborative group itself, and other organizations that the group engages with. Their framework can be used to evaluate what capacities exist within local collaboratives, and what capacities could be enhanced, so that investments in building and sustaining these groups can be targeted.

Davis et al. (2012) examined the organizational capacity of community-based collaborative groups in Oregon that are engaged in natural resource management issues on public as well as private lands. They found that many of these groups have limited financial capacity and insecure funding. They also have small numbers of staff who are expected to carry out a broad range of activities, and are heavily dependent on volunteers. Most accomplish their work through partnerships, often with federal agencies. They make a valuable contribution to local-level natural resource management and community economic well-being. Finding ways to successfully engage, support, and help build the capacity of local community-based collaboratives east of the Cascades in Oregon and Washington is an important

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strategy for building social agreement around the management of MMC forests, and for implementing forest restoration activities there.

Management of public lands today is as much a social experiment as it is an ecological experiment, especially when viewed through the lens of social-ecological resilience. The public wants to be involved. The best efforts to engage them in this work are not only well advised but critical to the success of restoration programs. It will be prudent to engage expertise in the areas of public involvement, social science, and economics in the cadre of players who help guide these national forest planning and management efforts.

Institutional Capacity

Land managers operate within established policy, constraints, and limitations as well as in accordance with established practices and conventions. However, some of the potential changes in forest management evoked within this document represent a departure from "business as usual." Land managers will decide how to proceed, and this will depend in large part on budget, policy, local circumstances, and, ultimately, the judgment of line officers. However, there are some ideas and observations from past work, both research and management, that suggest some prudent adjustments in management approach.

Create Landscape Management Demonstration Areas

We suggest that the identified approach to project and forest planning described above dovetails quite seamlessly with the intentions and expectations of the 2012 Forest Service planning rule (36 CFR Part 219) and has value for all forms of ownership. In that vein, perhaps the findings and conclusions contained herein can be implemented both formally within the planning rule context as well as informally for current project planning on public and private lands. On public lands, Forest Service staff (or other public land managers) can identify the appropriate landscapes, sometimes including mixed ownerships, and treat them as "landscape laboratories" or landscape demonstration areas. Although the geographic scope of these "learning" landscapes is yet to be fully defined, we are generally referring to large drainage basins on the order of tens to hundreds of thousands of hectares (1 ha = about 2.5 ac).

These large landscapes can provide the context in which we evaluate past, current, and potential future conditions and then develop project plans and desired future conditions for stands, watersheds, and the entire landscape (i.e., a nested hierarchical spatial organization of a landscape) accordingly. In this way, the project and forest planners can include the crucial perspective of broader spatial and temporal scales as they consider what to do on any given project area. It is prudent to put local project evaluations within this context of a nested landscape that can effectively consider the broader scales where the full implications of ecosystem drivers (e.g., fire, insects, and disease) and the periodicity of their occurrences can be accurately assessed.

National Forest System (NSF) staff, in collaboration with Research and Development (R&D) staff, and a variety of potential partners and collaborators will likely work together to develop this approach. Information from a wide variety of sources (e.g., experimental forests and ranges situated in the landscapes, other research results, NFS legacy data) can be used to develop better management options for public land managers and private landowners. This will also enable implementation of ecosystem restoration activities across the landscape, as well as the Department of Agriculture's guidance to consider "all lands" in our planning and management activities. Within these "landscape laboratories," Forest Service R&D, universities, and other partners with relevant information will be able to contribute long-term data records. This approach is a significant administrative and scientific challenge but it represents an important learning opportunity. The Forest Service has already made some strides in developing this concept through the relatively new Collaborative Forest Landscape Restoration Program (CFLRP). The purpose of the CFLRP is to encourage the collaborative, science-based ecosystem restoration of priority forest landscapes. Twenty projects have been established across the United States since 2010 to implement this concept. Perhaps this effort can expand to encompass larger experimental areas and a more robust role for the assistance from the scientific community.

Incorporate the "All-Lands" Initiative

The Department of Agriculture's 2010–2015 Strategic Plan contains a goal that strives to "ensure our national forests and private working lands are conserved, restored, and made more resilient to climate change, while enhancing our water resources." There are many facets to this ambitious goal and one new initiative in support of this goal involves use of a collaborative, "all lands" approach to bring public and private owners together across landscapes and ecosystems.

An all-lands approach would move landscape-level assessment and planning beyond the limits of the MMC forest type to other forest types, other land designations, and other ownerships that are often intermingled on logical landscape-level planning units. Managers can strive to involve all owners and management agencies and include all lands, even wilderness, in this kind of landscape planning. Additionally, these all-lands assessment efforts may indicate the need to treat areas such as wilderness-designated areas with controlled ignitions and burning to achieve landscape-level goals such as fuels reduction and achieving target future ranges of variation.

Revisit Current Management Constraints; Eastside Screens

In August 1993, the Pacific Northwest Region regional forester issued a letter providing interim direction to eastside national forests on retaining old-growth attributes at the local scale and moving toward the historical range of variability (HRV) across the landscape. These became known as the "eastside screens." A subsequent decision notice in May 1994 amended all eastside forest plans to include these standards. These provisions involved a three-stage process for screening projects to evaluate effects:

- Riparian Screen. Defers timber harvest in riparian areas.
- Ecosystem Screen. Compares the acres of old forest stages in a watershed with the HRV for that structural stage.
- Wildlife Screen. Maintains options for future wildlife habitat requirements for old growth-dependent species.

Eastside screens for large trees (>21 in [about 53 cm] in diameter) were intended to be an 18-month interim measure to screen out projects that could potentially harvest significant amounts of old forest and of remnant medium- and large-size trees. Managers were advised to avoid removal of trees above a certain diameter limit for a variety of reasons. The eastside forest health assessment in eastern Oregon and Washington had shown that remnant large trees and old-forest patches had been heavily targeted for timber harvest, and were seriously depleted (Everett et al. 1994). Public concern for loss of large and old trees also became significant beginning in the late 1980s and has continued to the present. Forest managers had to rapidly "screen" timber sales and then exclude parcels from those sales that included these large trees. Tree diameter was also used as a rapid and conservative but crude surrogate for old growth to limit removal of larger and older trees—because analyses had not been completed to characterize old forests and old trees—across the variety of forest types and productivities.

The team that developed the large tree screen designed it to be replaced by more in-depth landscape evaluations that considered the key departures in forest structural and compositional patterns from HRV. In fact, in 1994 the regional forester stated that "projects need to be designed according to the principles of landscape ecology and conservation biology." Nonetheless, these screens have been employed for 20 years now. We are not aware of how much variation in these procedures has been exercised since the screens were instituted, but the screens are still being used. Restoration of patterns that support natural processes and terrestrial species habitat arrangements is better achieved through evaluations that lead to landscape prescriptions. We know of no research that has evaluated the effects of the screens since they were implemented. However, other research on the ecology and history of pine and mixed-conifer forest (Hessburg et al. 1999a, 2005; Merschel 2012; Perry et al. 2004) indicates that many shade-tolerant trees more than 51 cm (20 in) in diameter were established after fire exclusion in the early 1900s. Consequently, restoration guided by size alone will not remove all of the individuals of species and ages of trees that are products of the altered disturbance regimes of these forests.

Furthermore, a recent report on climate change impacts on Western forests indicates that increased fire frequency and severity, insect outbreaks, and drought stress will challenge managers with novel conditions that fall outside what was identified from the 500+ years prior to Euro-American settlement (Fettig et al. 2013). To develop adaptation strategies, including ones that allow for new species composition and changes in forest structure across the landscape, managers will need more flexibility than is provided under one-size-fits-all rules for silviculture, such as those imparted by the eastside screens. Local variability coupled with an uncertain climate future requires an ability to adjust to site conditions and be nimble, as better and more complete information becomes available.

Rather than simple rules based only on stand structure, managers can be guided by multi-scale evaluations that include landscape and site-level criteria.

Rather than simple rules based only on stand structure, managers can be guided by multi-scale evaluations that include landscape and site-level criteria. Such approaches would consider and remedy the key departures in forest structural and compositional patterns relative to their HRV and future range of variability (FRV), the latter of which incorporates climate change. This integration of historical and potential future conditions is likely to conserve more desired components of ecosystems than would occur under the use of HRV or the FRV alone. The departure analyses using the HRV and the FRV are not meant as a recipe, but as guidance about the nature of pre-management-era patterns that supported native species habitats and disturbance regimes. With climate warming we use the FRV in the same way as HRV—as guidance about the nature of the patterns that may emerge as species distributions and disturbance regimes (e.g., ecological envelopes) shift as climate interacts with land use to create novel environmental conditions and behaviors. As we are seeing in eastern Washington ecoregions, these envelopes of HRV and FRV can actually overlap quite a bit, but are clearly being deformed by warmer and drier conditions.

As managers reevaluate their planning approaches using the concepts and findings contained in this report, they might also consider phasing out these interim directions. As mentioned, these screens were a short-term solution to prevent harvesting of larger trees. The diameter limits were intended as a crude and conservative filter to avoid harvesting old trees, pending the development of more precise definitions and tools. These limits are arbitrary with respect to meeting any ecological objectives because they applied to all sites regardless of the number or species of large trees present, or the potential of these areas to grow large trees. The limits on removing any tree larger than about 53 cm (21 in) whatsoever, regardless of geographic context, or age, or species, or relative abundance, or other considerations (e.g., forest health) within a patch can inhibit regeneration in some stands, lack any real landscape objectives, and impede landscape-level management and restoration.

The concepts presented in this report address the underlying ecological objectives of the eastside screens. As these concepts are explored and implemented by managers, the ecological goals intended by the eastside screens can be met. Some individual features (i.e., nesting, roosting, and resting sites for wildlife) may continue to merit short-term conservation measures. However, employing a landscape approach to assessment, planning, and execution of forest management will improve our ability to effectively restructure forests to a more resilient condition and cope with impending change. Moving landscapes toward these desired conditions will be expensive, thus some flexibility in treatment options is needed. Treatments designed based on the considerations herein are capable of generating some revenue that will offset a portion of the costs of achieving landscape-level objectives. Flexibility will also help managers address long-term socio-political and operational issues related to eastside screens.

Chapter 7—Conclusions

Active management of vegetation in moist mixed-conifer (MMC) forests in eastern Oregon and Washington has declined over the last two decades because of uncertainties about management. During this time, shade-tolerant and fire-intolerant trees have regenerated, grown, and densified forests in many locations. Consequently much of the landscape of ponderosa pine and dry and MMC forests of eastern Oregon and Washington is in a non-resilient condition. Some of the MMC landscape would benefit from active vegetation and fire management to restore or create vegetation conditions and landscape patterns that are better adapted to fire, drought, insects and disease, and climate change than conditions and patterns managed passively. Although managers have a scientific foundation and some amount of social license to conduct active restoration in pine and dry mixed-conifer types, there is far less consensus on what should be done in MMC types. This report offers scientific findings and management considerations to help managers take informed steps toward restoring resilience to MMC forests.

This report offers scientific findings and management considerations to help managers take informed steps toward restoring resilience to moist mixed-conifer forests.

Dry, moist, and wet mixed-conifer forests coexist in landscape mosaics that have been altered by settlement and land use activities. On balance, the array of changes over the last 100 to 150 years make these forests and landscapes subject to high-severity and sometimes large fires (with varying degrees of mixed- and low-severity components), insect outbreaks, and drought-related tree mortality that negatively affect many ecological and social values. During the presettlement era, the dry **and** MMC forest landscapes were sculpted by low-, mixed-, and high-severity wildfires (tables 8 through 13 in app. 3) that maintained relatively open canopies and understories. Landscapes were mosaics, dominated by a mix of tree species, sizes, and ages (tables 8 through 13 in app. 3). Some ecoregions exhibited more mixed- and high-severity fire than others.

Environments and vegetation patterns are diverse and complex in the MMC forest (e.g., historical disturbance regimes differ within potential vegetation types as a result of topography and landscape context). We lack uniform vegetation classifications, maps, and disturbance regime information that can be applied at local landscape scales. This information is sorely needed. Despite these challenges, we were able to identify and develop a working definition of the MMC types (potential vegetation types, disturbance regimes, ecoregions) within the broader mixed-conifer mosaic, where structure, composition, and processes have been significantly altered by Euro-American activities.

In many respects the impacts of postsettlement activity on forests and implications to restoration are the same in MMC forests as those in the ponderosa pine and dry mixed-conifer forests. The disturbance regimes were similar in the dry and moist environments, but in the productive MMC forest environments the large increases in understory density were composed of shade-tolerant species like grand fir or Douglas-fir rather than ponderosa pine. We identify that in the wettest mixed-conifer types, restoration of resilient vegetation patterns is much less needed or inappropriate given their low frequency disturbance regimes.

Given the stated restoration goals of the Forest Service, we identify a set of objectives and actions at stand and landscape levels that would move forests and landscapes on a trajectory toward resilience in the face of fire, insects, disease, and climate change. These include creating diverse, fire-resilient vegetation types over a large portion of the MMC landscape, creating seral-stage diversity and patterns at stand and landscape scales, using topography as a guide, maintaining habitats for key wildlife species, including those that need landscapes with patches of dense mixed-conifer forests, and maintaining ecological and physical processes that favorably reconnect terrestrial and aquatic ecosystems and support habitat for fish.

Stands (patches) and local landscapes, conjoined with regional landscape-scale perspectives, are needed to achieve restoration goals. Although we address all three scales in this report, we have emphasized local landscape perspectives because they are newer and the theory and practice is rapidly developing through research and innovative management. For those seeking more detail on the latest stand-level considerations for mixed-conifer forests, other resources exist (e.g., Franklin et al. 2013). The geographic scope and context of forest restoration dictates an “all lands” approach to address the challenges of the 21st century. Vegetation succession and disturbance dynamics emerging from wildfires, insect and disease outbreaks, exotic species invasions, and multiple ownerships and management directions—set against a backdrop of climate change—do not observe local forest patch boundaries, land lines, or administrative boundaries. More effective land management can be realized through action grounded in collaboration and a landscape perspective.

A list of management considerations is provided for planners and managers when managing for restoration and resilience at landscape scales (see app. 1). The list includes guidance on development and use of historical reference conditions (historical range variation is a useful guide), addressing climate change (using potential future range of variation in designing landscapes), and a workflow plan. Several eastside national forests and other forest landowners in this region now actively employ some principles of restoration management at a landscape scale. The example provided from the Okanogan-Wenatchee National Forest is one of a number of bottom-up initiatives that explore innovative approaches to forest management. Many of the concepts presented in this report may be known to land managers, but they are not yet in common practice. Understandably, new innovations and scientific insights take time and often require experimentation to integrate into standard operations.

It makes sense for both the management and research communities to work together to learn about restoration and collectively stop every few years and take stock of where we are and what we have learned in implementing these ideas. This is, in essence, the conceptual core of adaptive management, and we encourage such flexible management approaches that promote learning and sharing. It also makes sense to build stronger day-to-day ties between management and research to enhance the real-time flow of information, improve researcher insights about ongoing and newly emergent problems, and to improve management by experimentation and adaptation. Regional land management, with all of its uncertainties and risks, and the significant and growing public scrutiny, is an enterprise that could benefit from increased adaptive management and enhanced science-management collaboration.

Land management activities necessarily integrate many policy and societal considerations, while utilizing the best available science. Managers can rely on well-tested, dependable methods and approaches of our respective professions, but can also continually avail themselves of developing innovations and technologies to better enable decisions and implementation. That, we believe, is evident in the synthesis presented in this report. We have presented a synopsis of the best available science, old and new, combined with an array of management concepts and principles that have yet to become common practice. We have also assessed the best of both traditional scientific and management approaches with some recent innovations that substantially improve our ability to understand how complex forest ecosystems function and how management influences those processes.

Finally, when viewed through the lens of socioecological resilience, creating ecologically resilient landscapes cannot occur without support and consideration of the institutions and socioeconomic components. Restoration and renewal of resiliency in eastside forest landscapes requires economic support, forest management infrastructure, and social license and partnerships between management agencies and various publics. Human activities over the last 150 years have reduced the resilience of mixed-conifer forests across millions of acres of federal lands. This problem, a long time building and currently widespread, will require a sustained, comprehensive, and collaborative approach to remedy.

Restoration and renewal of resiliency in eastside forest landscapes requires economic support, forest management infrastructure, and social license and partnerships between management agencies and various publics.

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Metric Equivalents

When you know:	Multiply by:	To get:
Inches (in)	2.54	Centimeters
Feet (ft)	.305	Meters
Miles (mi)	1.609	Kilometers
Square miles (mi^2)	2.59	Square kilometers
Acres (ac)	.405	Hectares

Literature Cited

Abella, S.R.; Covington, W.W. 2006. Forest ecosystems of an Arizona *Pinus ponderosa* landscape: multifactor classification and implications for ecological restoration. Journal of Biogeography. 33: 1368–1383.

Abrams, J.B.; Kelly, E.; Shindler, B.; Wilton, J. 2005. Value orientation in forest management: the forest health debate. Environmental Management. 36: 495–505.

Agee, J.K. 1988. Successional dynamics in forest riparian zones. In: Raedeke, K.J., ed. Streamside management: riparian wildlife and forestry interactions. Proceedings of a symposium on riparian wildlife interactions. Seattle, WA: University of Washington: 31–44.

Agee, J.K. 1990. The historical role of fire in Pacific Northwest forests. In: Walstad, J.D.; Radosevich, S.R; Sandberg, D.V., eds. Natural and prescribed fire in Pacific Northwest forests. Corvallis, OR: Oregon State University Press: 25–38.

Agee, J.K. 1993. Fire ecology of Pacific Northwest Forests. Washington, DC: Island Press. 493 p.

Agee, J.K. 1994. Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. In: Hessburg, P.F., ed. Eastside forest ecosystem health assessment. Gen. Tech. Rep. PNW-GTR-320. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 52 p.

Agee, J.K. 1998. The landscape ecology of western forest fire regimes. Northwest Science. 72: 24–34.

Agee, J.K. 2003. Historical range of variability in eastern Cascades forests, Washington, USA. Landscape Ecology. 18: 725–740.

Agee, J.K.; Skinner, C.N. 2005. Basic principles of fuel reduction treatments. Forest Ecology and Management. 211: 83–96.

Ager, A.A.; Finney, M.A.; Kerns, B.K.; Maffei, H. 2007. Modeling wildfire risk to northern spotted owl (*Strix occidentalis caurina*) habitat in central Oregon, USA. Forest Ecology and Management 246: 45–56.

Aguilar, F.; Garrett, H.E. 2009. Perspectives of woody biomass for energy: survey of state foresters, state energy biomass contacts, and national council of forestry association executives. Journal of Forestry. 107: 297–306.

Alexander, M.E.; Hawksworth, F.G. 1975. Wildland fires and dwarf mistletoe: a literature review of ecology and prescribed burning. Gen. Tech. Rep. RM-GTR-14. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 12 p.

Anderson P.D.; Larson D.J.; Chan S.S. 2007. Riparian buffer and density management influences on microclimate of young headwater forests of western Oregon. Forest Science. 53: 254–269.

Arno, S.F. 1988. Fire ecology and its management implications in ponderosa pine forests. In: Baumgartner, D.M.; Lotan, J.E., eds. Ponderosa pine: the species and its management: Proceedings of a symposium. Pullman, WA: Washington State University Extension: 133–139.

Asah, S.T.; Blahna, D.J.; Ryan, C.M. 2012. Involving forest communities in identifying and constructing ecosystem services: millennium assessment and place specificity. Journal of Forestry. 110: 149–156.

Ascher, T.J.; Wilson, R.S.; Toman, E. 2013. The importance of affect, perceived risk, and perceived benefit in understanding support for fuels management among wildland-urban interface residents. International Journal of Wildland Fire. 22: 267–276.

Aubry, K.B.; Koehler, G.M.; Squires, J.R. 2000. Ecology of Canada lynx in southern boreal forests. In: Ruggiero, L.F.; Aubry, K.B.; Buskirk, S.W.; Koehler, G.M.; Krebs, C.J.; McKelvey, K.S.; Squires, J.R., eds. Ecology and conservation of lynx in the United States. Gen. Tech. Rep. RMRS-GTR-30. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 373–396.

Aubry, K.B.; McKelvey, K.S.; Copeland, J.P. 2010. Distribution and broadscale habitat relationships of the wolverine in the contiguous United States. Journal of Wildlife Management. 71: 2147–2158.

Ault, T.R.; Cole, J.E.; St. George, S. 2012. The amplitude of decadal to multi-decadal variability in precipitation simulated by state-of-the-art climate models. Geophysical Research Letters. 39:10.1929/2012GL053424.

Bailey, R.G. 1995. Description of the ecoregions of the United States. Misc. Publ. 1391. Washington, DC: U.S. Department of Agriculture. 108 p.

Baker, C.S.; Batchelder, H.P.; Epps, C.; Garcia, T.S.; Haig, S.M.; Letelier, R.M.; Lytle, D.A.; Menge, B.A.; Miller, J.C.; Noakes, D.L.G.; Peterson, W.T.; Rice, J.M.; Rumrill, S.S.; Schreck, C.B.; Suryan, R.M.; Sytsma, M.D.; White, A.E. 2010. Oregon's fish and wildlife in a changing climate. In: Dello, K; Mote, P., eds. Oregon Climate Assessment Report. Corvallis, OR: Oregon State University, Oregon Climate Change Research Institute. Chapter 7. http://occri.net/wp-content/uploads/2011/04/chapter7ocar.pdf. (17 January 2014).

Baker, W.L. 2012. Implications of spatially extensive historical data from surveys for restoring dry forests of Oregon's eastern Cascades. Ecosphere. 3(3): 23.

Barbour, J.R.; Fried, J.S.; Daugherty, P.J.; Christensen, G.; Fight, R. 2008. Potential biomass and logs from fire hazard reduction treatments in Southwest Oregon and Northern California. Forest Policy and Economics. 10: 400–407.

Bartlett, G. 2012. Developing collaboration and cooperation. In: North, M., ed. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 81–88.

Bartos, D.L. 2000. Landscape dynamics of aspen and conifer forests. Aspen Bibliography. Paper 782. http://digitalcommons.usu.edu/aspen_bib/782. (17 January 2014).

Baxter C.V.; Fausch K.D.; Murakami, M.; Chapman, P.L. 2004. Fish invasion restructures stream and forest food webs by interrupting reciprocal prey subsidies. Ecology. 85: 2656–2663.

Becker, D.R.; Larson, D.; Lowell, E.C. 2009. Financial considerations of policy options to enhance biomass utilization for reducing wildfire hazards. Forest Policy and Economics. 11: 628–635.

Becker, D.R.; McCaffrey, S.M.; Abbas, D.; Halvorsen, K.E.; Jakes, P.; Moseley, C. 2011. Conventional wisdoms of woody biomass utilization on federal public lands. Journal of Forestry. 109: 208–218.

Bednarz, J.C.; Ripper, D.; Radley, P.M. 2004. Emerging concepts and research directions in the study of cavity-nesting birds: keystone ecological processes. The Condor. 106: 1–4.

Beebe, J.; Everett, R.; Scherer, G.; Davis, C. 2002. Effect of fertilizer applications and grazing exclusion on species composition and biomass in wet meadow restoration in Eastern Washington. Res. Pap. PNW-RP-542. Portland OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 15 p.

Beechie, T.; Bolton, S. 1999. An approach to restoring salmonid habitat-forming processes in Pacific Northwest watersheds. Fisheries. 24: 6–15.

Beese, W.J.; Dunsworth, B.G.; Zielke, K.; Bancroft, B. 2003. Maintaining attributes of old-growth forests in coastal BC through variable retention. Forestry Chronicle. 79: 570–578.

Belsky, J.A.; Matzke, A.; Uselman, S. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation. 54: 419–431.

Benda, L.; Miller, D.; Andras, K.; Bigelow, P.; Reeves, G.; Michael, D. 2007. NetMap: a new tool in support of watershed science and resource management. Forest Science. 52: 206–219.

Benda, L.; Miller, D.; Bigelow, P.; Andras, K. 2003. Effects of post-wildfire erosion on channel environments, Boise River, Idaho. Forest Ecology and Management. 178: 105–119.

Benitez-Lopez, A.; Alekemade, R.; Verweij, P.A. 2010. The impacts of roads and other infrastructure on mammal and bird populations: a meta-analysis. Biological Conservation 143: 1307–1316.

Bennett, A.F. 2003. Linkages in the landscape: the role of corridors and connectivity in wildlife conservation. IUCN Forest Conservation Programme, Conserving Forest Ecosystems Series No. 1. Gland, Switzerland: IUCN. 254 p.

Bentz, B.J.; Régnière, J.; Fettig, C.J.; Hansen, E.M.; Hayes, J.L.; Hicke, J.A.; Kelsey, R.G.; Negrón, J.F.; Seybold, S.J. 2010. Climate change and bark beetles of the western U.S. and Canada: Direct and indirect effects. Bioscience. 60: 602–613.

Berkes, F.; Ross, H. 2013. Community resilience: toward an integrated approach. Society and Natural Resources. 26: 5–20.

Beschta, R.L.; Donahue, D.L.; DellaSala, D.A.; Rhodes, J.J.; Karr, J.R.; O'Brien, M.H.; Fleischner, T.L.; Deacon Williams, C. 2012. Adapting to climate change on western public lands: addressing the ecological effects of domestic, wild, and feral ungulates. Environmental Management. 51: 474–491.

Beschta, R.L.; Ripple, W.J. 2009. Large predators and trophic cascades in terrestrial ecosystems of the western United States. Biological Conservation. 142: 2401–2414.

Betancourt, J.L.; Van Devender, T.R.; Martin, P.S. 1990. Packrat middens: the last 40,000 years of biotic change. Tucson, AZ: University of Arizona Press. 469 p.

Betts, M.G.; Hagar, J.; Rivers, J.; Alexander, J.; McGarigal, K.; McComb, B.C. 2010. Thresholds in songbird occurrence in relation to early seral broadleaf forest. Ecological Applications. 20: 2116–2130.

Binckley, C.A.; Wipfli, M.S.; Medhurst, R.B.; Polivka, K.; Hessburg, P.; Salter, R.B.; Kill, J.Y. 2010. Ecoregion and land-use influence invertebrate and detritus transport from headwater streams. Freshwater Biology. 55: 1205–1218.

Bisson, P.A.; Rieman, B.E.; Luce, C.; Hessburg, P.F.; Lee, D.C.; Kershner, J.; Reeves, G.H.; Gresswell, R.E. 2003. Fire and aquatic ecosystems of the western USA: current knowledge and key questions. Forest Ecology and Management. 178: 213–229.

Bolsinger, C.L. 1978. The extent of dwarf mistletoe in six principal softwoods in California, Oregon, and Washington, as determined from forest survey records. In: Scharpf, R.F.; Parmeter, J.R. Jr., eds. Proceedings of a symposium on dwarf mistletoe control through forest management. Gen. Tech. Rep. PSW-GTR-31. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 45–54.

Bright, A.D.; Newman, P.; Carroll, J. 2007. Context, beliefs, and attitudes toward wildland fire management: an examination of residents of the wildland-urban interface. Human Ecology Review. 14: 212–222.

Brown, D.K.; Echell, A.A.; Propst, D.L.; Brooks, J.E.; Fisher, W.L. 2001. Catastrophic wildfire and number of populations as factors influencing risk of extinction for Gila trout (*Oncorhynchus gilae*). Western North American Naturalist. 61: 139–148.

Brown, R.N.K.; Rosenberger, R.S.; Kline, J.D.; Hall, T.E.; Needham, M.D. 2008. Visitor preferences for managing wilderness recreation after wildfire. Journal of Forestry. 106: 9–16.

Brunson, M.W.; Shindler, B.A. 2004. Geographic variation in social acceptability of wildland fuels management in the western United States. Society and Natural Resources. 17: 661–678.

Buchanan, J.B.; Irwin, L.L.; McCutchen, E.L. 1995. Within-stand nest site selection by spotted owls in the Eastern Washington Cascades. Journal of Wildlife Management. 59: 301–310.

Bull, E.L.; Heater, T.W.; Shepherd, J.F. 2005. Habitat selection by the American marten in northeastern Oregon. Northwest Science. 79: 37–43.

Bull, E.L.; Henjum, M.G. 1990. Ecology of the great gray owl. Gen. Tech. Rep. PNW-GTR-265. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 39 p.

Bull, E.L.; Holthausen, R.S. 1993. Habitat use and management of pileated woodpeckers in northeastern Oregon. Journal of Wildlife Management. 57: 335–345.

Bull, E.L.; Holthausen, R.S.; Henjum, M.G. 1992. Roost trees used by pileated woodpeckers in northeastern Oregon. Journal of Wildlife Management. 56: 786–793.

Bull, E.L., Parks, C.G.; Torgerson, T.R. 1997. Trees and logs important to wildlife in the interior Columbia River basin. Gen. Tech. Rep. PNW-GTR-391. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 55 p.

Bull, E.L.; Torgersen, T.R.; Blumton, A.K.; McKenzie, C.M.; Wyland, D.S. 1995. Treatment of an old-structure stand and the effect on birds, ants, and large woody debris: a case study. Gen. Tech. Rep. PNW-GTR-353. Portland, OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 12 p.

Bull, E.L.; Wright, A.L.; Henjum, M.G. 1989. Nesting and diet of long-eared owls in conifer forests, Oregon. Condor. 91: 908–912.

Bull, W.B. 1991. Geomorphic responses to climate change. New York: Oxford University Press. 352 p.

Bunnell, F.L. 1995. Forest-dwelling vertebrate faunas and natural fire regimes in British Columbia: patterns and implications for conservation. Conservation Biology. 9: 636–644.

Burton, T.A. 2005. Fish and stream habitat risks from uncharacteristic wildfire: observations from 17 years of fire-related disturbances on the Boise National Forest, Idaho. Forest Ecology and Management. 211: 140–149.

Cacciapaglia, M.A.; Yung, L.; Patterson, M.E. 2012. Place mapping and the role of spatial scale in understanding landowner views of fire and fuels management. Society & Natural Resources. 25: 453–467.

Camp, A.E. 1999. Age structure and species composition changes resulting from altered disturbance regimes on the eastern slopes of the Cascades Range, Washington. Journal of Sustainable Forestry. 9: 39–67.

Camp, A.E.; Oliver, C.D.; Hessburg, P.F.; Everett, R.L. 1997. Predicting late-successional fire refugia from physiography and topography. Forest Ecology and Management. 95: 63–77.

Carey, A.B. 2003. Biocomplexity and restoration of biodiversity in temperate coniferous forest: inducing spatial heterogeneity with variable-density thinning. Forestry. 76: 127–136.

Carey, A.B. 2007. AIMing for healthy forests: active, intentional management for multiple values. Gen. Tech Rep. PNW-GTR-721. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 447 p.

Carpenter, S., Walker, B.; Anderies, J.M.; Ablel, N. 2001. From metaphor to measurement: Resilience of what to what? Ecosystems. 4: 765–781.

Cayan, D.R.; Kammerdiener, S.A.; Dettinger, M.D.; Caprio, J.M.; Peterson, D.H. 2001. Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society. 82: 399–415.

Chambers, J.; Leger, E.; Goergen, E. 2009. Cold desert fire and invasive species management: resources, strategies, tactics, and response. Rangelands. 31: 14–20.

Charnley, S.; Fischer, A.P.; Jones, E.T. 2007. Integrating traditional and local ecological knowledge into forest biodiversity conservation in the Pacific Northwest. Forest Ecology and Management. 246: 14–28.

Charnley, S.; Long, J.; Lake, F.K. 2014. Collaboration in national forest management. In: Long, J.W.; Quinn-Davidson, L.; Skinner, C.N., eds. Science synthesis to promote socioecological resilience in the Sierra Nevada and southern Cascade Range. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 661–701. Chapter 9.6.

Cheng, A.; Sturtevant, V.E. 2012. A framework for assessing collaborative capacity in community-based public forest management. Environmental Management. 49: 675–689.

Choat, B.; Jansen, S.; Brodribb, T.J.; Cochard, H.; Delzon, S.; Bhaskar, R.; Bucci, S.J.; Feild, T.S.; Gleason, S.M.; Hacke, U.G.; Jacobsen, A.L.; Lens, F.; Maherali, H.; Martínez-Vilalta, J.; Mayr, S.; Mencuccini, M.; Mitchell, P.J.; Nardini, A.; Pittermann, J.; Pratt, R.B.; Sperry, J.S.; Westoby, M.; Wright, I.J.; Zanne, A.E. 2012. Global convergence in the vulnerability of forests to drought. Nature. 491: 752–755.

Churchill, D.J.; Larson, A.J.; Dahlgreen, M.C.; Franklin, J.F.; Hessburg, P.F.; Lutz, J.A. 2013. Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management. 291: 442–457.

Climate Impacts Group [CIG]. 2009. The Washington climate change impacts assessment: evaluating Washington's future in a changing climate. Seattle, WA: Center for Science in the Earth System, Joint Institute for the Study of the Atmosphere and Oceans, University of Washington. 407 p. http://cses.washington.edu/db/pdf/wacciareport681.pdf. (17 January 2014).

Cobb, D.F.; O'Hara, K.L.; Oliver, C.D. 1993. Effect of variations in stand structure on development of mixed species stands in eastern Washington. Canadian Journal of Forest Research. 23: 545–552.

Cook, R.C.; Cook, J.G.; Vales, D.J.; Johnson, B.K.; McCorquodale, S.M.; Shipley, L.A.; Riggs, R.A.; Irwin, L.L.; Murphie, S.L.; Murphie, B.L.; Schoenecker, K.A.; Geyer, F.; Briggs Hall, P.; Spencer, R.D.; Immell, D.A.; Jackson, D.H.; Tiller, B.L.; Miller, P.J.; Schmitz, L. 2013. Regional and seasonal patterns of nutritional condition and reproduction in elk. Wildlife Monographs. 184: 1–45.

Converse, S.J.; White, G.C.; Farris, K.L.; Zack, S. 2006. Small mammals and forest fuel reduction: national-scale responses to fire and fire surrogates. Ecological Applications. 16: 1717–1729.

Courtney, S.; Blakesley, J.; Bigley, R.; Cody, M.; Dumbacher, J.; Fleischer, R.; Franklin, A.; Franklin, J.; Gutierrez, R.; Marzluff, J.; Sztukowski, L. 2004. Scientific evaluation of the status of the northern spotted owl. Portland, OR: Sustainable Ecosystems Institute. 258 p.

Crisafulli, C.M. 2005. Larch Mountain salamander*, Plethodon larselli* Burns. In: Jones, L.L.C.; Leonard, W.P.; Olson, D.H, eds. Amphibians of the Pacific Northwest. Seattle, WA: Seattle Audubon Society: 132–135.

Crooks, K.R.; Sanjayan, M. 2006. Connectivity conservation. Cambridge, United Kingdom: Cambridge University Press. 712 p.

Cushman, K.A.; C.A. Pearl. 2007. A conservation assessment for the Oregon spotted frog. Portland, OR: U.S. Department of Agriculture Forest Service and U.S. Department of Interior Bureau of Land Management. 46 p.

Dale, V.H.; Joyce, L.A.; McNulty, S.; Nielson, R.P.; Ayres, M.P.; Flannigan, M.D.; Hanson, P.J.; Irland, L.C.; Lugo, A.E.; Peterson, C.J.; Simerloff, D.; Swanson, F.J.; Stocks, B.J.; Wotton, B.M. 2001. Climate change and forest disturbances. Bioscience. 51: 723–734.

Dalton, M.M.; Mote, P.W.; Snover, A.K. 2013. Climate change in the Northwest: implications for our landscapes, waters, and communities. Washington, DC: Island Press. 270 p. http://cses.washington.edu/db/pubs/allpubs.shtml. (14 January 2014).

Daly, C.; Smith, J.W.; Smith, J.I.; McKane, R.B. 2007. High-resolution spatial modeling of daily weather elements for a catchment in the Oregon Cascade Mountains, United States. Journal of Applied Meteorology and Climatology. 46: 1565–1586.

Daniels, J.M. 2004. Assessing socioeconomic resiliency in Washington Counties. Gen. Tech. Rep. PNW-GTR-607. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 35 p.

D'Antonio, C.M.; Chambers, J.C.; Loh, R.; Tunison, J.T. 2009. Applying ecological concepts to the management of widespread grass invasions. In: Inderjit, R.L., ed. Management of invasive weeds. Chapter 7. Dordrecht, The Netherlands: Springer: 123–149.

Daugherty, P.J.; Fried, J.S. 2007. Jointly optimizing selection of fuel treatments and siting of forest biomass-based energy production facilities for landscape-scale fire hazard reduction. INFOR-OTTAWA. 45: 17–30.

Davidson, D.J. 2010. The applicability of the concept of resilience to social systems: some sources of optimism and nagging doubts. Society and Natural Resources. 23: 1135–1149.

Davis, E.J. 2012. Economic development and sustainable forest stewardship in the dry forest zone: a mid-term report. Portland, OR and Eugene, OR: Sustainable Northwest and University of Oregon, Ecosystem Workforce Program, Institute for a Sustainable Environment. 24 p.

Davis, E.J.; Moseley, C.; Evers, C.; Macfarland, K.; Nielsen-Pincus, M.; Pomeroy, A.; Enzer, M.J. 2012. Community-based natural resource management in Oregon: a profile of organizational capacity. Ecosystem Workforce Program Working Paper Number 39. Eugene, OR: University of Oregon. 17 p.

Davis, E.J.; Moseley, C.; Nielsen-Pincus, M. 2010. The state of dry zone forest and its communities. Eugene, OR and Portland, OR: University of Oregon, Ecosystem Workforce Program, Institute for a Sustainable Environment. 94 p.

Daw, S.K.; DeStefano, S. 2001. Forest characteristics of northern goshawk nest stands and post-fledging areas in Oregon. Journal of Wildlife Management. 65: 59–65.

Donoghue, E.M.; Thompson, S.A.; Bliss, J.C. 2010. Tribal-federal collaboration in resource management. Journal of Ecological Anthropology. 14: 22–38.

Dramm, J.R. 1999. Small-diameter utilization issues and opportunities. In: Conference proceedings, wood technology clinic & show. Portland, OR: Miller Freeman: 30–38.

Dugger, K.M.; Anthony, R.G.; Andrews, L.S. 2011. Transient dynamics of invasive competition: barred owls, spotted owls, habitat, and the demons of competition present. Ecological Applications. 21: 2459–2468.

Dunham, J.B.; Rosenberger, A.E.; Luce, C.H.; Rieman, B.E. 2007. Influences of wildfire and channel reorganization on spatial and temporal variation in stream temperature and the distribution of fish and amphibians. Ecosystems. 10: 335–346.

Dunham J.B.; Young M.K.; Gresswell R.E.; Rieman B.E. 2003. Effects of fires on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasions. Forest Ecology and Management. 178: 183–196.

Eastin, I.; Ganguly, I.; Sasatani, D.; Mason, L.; Lipple, B. 2009. Economic contribution of the forestry and wood processing sectors in the State of Washington, executive summary. Working paper 116. Seattle, WA: Center for International Trade in Forest Products. 2 p.

Ebersole, J.L.; Liss, W.J.; Frissell, C.A. 1997. Restoration of stream habitats in the western United States: restoration as reexpression of habitat capacity. Environmental Management. 21: 1–14.

Englin, J.; Loomis, J.; Gonzalez-Caban, A. 2001. The dynamic path of recreational values following a forest fire: a comparative analysis of states in the Intermountain West. Canadian Journal of Forestry Research. 31: 1837–1844.

Everett R.L.; Hessburg, P.F.; Jensen, M.; Bormann, B. 1994. Volume I: executive summary. Eastside forest health assessment. Gen. Tech. Rep. PNW-GTR-317. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 61 p.

Everett, R.L.; Schellhaas, R.; Keenum, D.; Spurbeck, D.; Ohlson, P. 2000. Fire in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. Forest Ecology and Management. 129: 207–225.

Everett, R.; Schellhaas, R.; Ohlson, P.; Spurbeck, D.; Keenum, D. 2003. Continuity in fire disturbance between riparian and adjacent sideslope Douglas-fir forests. Forest Ecology and Management. 175: 31–47.

Everett, R.; Schellhaas, D.; Spurbeck, D.; Ohlson, P.; Keenum, D.; Anderson, T. 1997. Structure of northern spotted owl nest stands and their historical conditions on the eastern slope of the Pacific Northwest Cascades, USA. Forest Ecology and Management. 94: 1–14.

Fetherston, K.L.; Naiman, R.J.; Bilby, R.E. 1995. Large woody debris, physical process, and riparian forest development in montane river networks of the Pacific Northwest. Geomorphology. 13: 133–144.

Fettig, C.J. 2012. Forest health and bark beetles. In: North, M., ed. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 13–22. Chapter 2.

Fettig, C.J.; Klepzig, K.D.; Billings, R.F.; Munson, A.S.; Nebeker, T.E.; Negrón, J.F.; Nowak, J.T. 2007. The effectiveness of vegetation management practices for prevention and control of bark beetle outbreaks in coniferous forests of the western and southern United States. Forest Ecology and Management. 238: 24–53.

Fettig, C.J.; Reid, M.L.; Bentz, B.J.; Sevanto, S.; Spittlehouse, D.L.; Wang, T. 2013. Changing climates, changing forests: a western North American perspective. Journal of Forestry. 111: 214–228.

Filip, G.M. 1990. Effect of tree harvesting on Armillaria root disease in an old-growth mixed-conifer stand in northeastern Oregon. Northwest Environmental Journal. 6: 412–413.

Filip, G.M.; Goheen, D.J. 1982. Tree mortality caused by a root pathogen complex on the Deschutes National Forest, Oregon. Plant Disease. 66: 240–243.

Filip, G.M.; Goheen, D.J. 1984. Root diseases cause severe mortality in white and grand fir stands of the Pacific Northwest. Forest Science. 30: 138–142.

Filip, G.M., Maffei, H.M.; Chadwick, K.L.; Max, T.A. 2010. Armillaria root disease-caused tree mortality following silvicultural treatments (shelterwood or group selection) in an Oregon mixed-conifer forest: insights from a 10-year case study. Western Journal of Applied Forestry. 25: 136–143.

Filip, G.M.; Schmitt, C.L. 1979. Susceptibility of native conifers to laminated root rot east of the Cascade Range in Oregon and Washington. Forest Science. 25: 261–265.

Filip, G.M.; Schmitt, C.L.; Hosman, K.P. 1992. Effects of harvesting season and stump size on incidence of annosum root disease of true fir. Western Journal of Applied Forestry. 7: 54–57.

Finney, M.A.; Seli, R.C.; McHugh, C.W.; Ager, A.A.; Bahro, B.; Agee, J.K. 2007. Simulation of long-term landscape-level fuel treatment effects on large wildfires. International Journal of Wildland Fire. 16: 712–727.

Fischer, W.C.; Bradley, A.F. 1987. Fire ecology of western Montana forest habitat types. Gen. Tech. Rep. INT-GTR-223. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. 95 p.

Folke, C. 2006. Resilience: the emergence of a perspective for socio-ecological systems analyses. Global Environmental Change. 16: 253–267.

Forsman, E.D. 1983. Methods and materials for locating and studying spotted owls. Gen. Tech. Rep. PNW-GTR-162. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 8 p.

Forsman, E.D.; Anthony, R.; Dugger, K.; Glenn, E.; Franklin, A.; White, G.; Schwarz, C.; Burnham, K.; Anderson, D.; Nichols, J.; Hines, J.; Lint, J.; Davis, R.; Ackers, S.; Andrews, L.; Biswell, B.; Carlson, P.; Diller, L.; Gremel, S.; Herter, D.; Higley, J.; Horn, R.; Reid, J.; Rockweit, J.; Schaberl, J.; Snetsinger, T.; Sovern, S. 2011. Population demography of northern spotted owls. Studies in Avian Biology No. 40. Los Angeles, CA: University of California Press. 120 p.

Forsman, E.D.; Anthony, R.G.; Meslow, E.C.; Zabel, C.J. 2004. Diets and foraging behavior of northern spotted owls in Oregon. Journal of Raptor Research. 38: 214–230.

Forsman, E.D.; Otto, I.A.; Sovern, S.G.; Taylor, M.; Hays, D.W.; Allen, H.; Roberts, S.L.; Seaman, D.E. 2001. Spatial and temporal variation in diets of Spotted Owls in Washington. Journal of Raptor Research. 35: 141–150.

Franceschi, V.R.; Krokene, P.; Christiansen, E.; Krekling, T. 2005. Anatomical and chemical defenses of conifer bark against bark beetles and other pests. New Phytologist. 167: 353–376.

Franklin, J.F.; Dyrness, C.T. 1973. Natural vegetation of Oregon and Washington. Gen. Tech. Rep. PNW-GTR-008. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 427 p.

Franklin, J.F.; Dyrness, C.T. 1988. Natural vegetation of Oregon and Washington. Corvallis, OR: Oregon State University Press. 464 p.

Franklin, J.F.; Johnson, K.N. 2012. A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry. 110: 429–439.

Franklin, J.F.; Johnson, K.N.; Churchill, D.; Hagmann, K.; Johnson, D.; Johnston, J. 2013. Restoration of dry forests in eastern Oregon: a field guide. Portland, OR: The Nature Conservancy. 202 p.

Franklin, J.F.; Lindenmayer, D.; MacMahon, J.A.; McKee, A.; Magnuson, J.; Perry, D.A.; Waide, R.; Foster, D. 2000. Threads of continuity: there are immense differences between even-aged silvicultural disturbances (especially clearcutting) and natural disturbances, such as windthrow, wildfire, and even volcanic eruptions. Conservation in Practice. 1: 8–17.

Frissell, C.A.; Liss, W.J.; Gresswell, R.E.; Nawa, R.K.; Ebersole, J.L. 1997. A resource in crisis: changing the measure of salmon management. In: Stouder, D.J.; Bisson, R.A.; Naiman, R.J., eds. Pacific salmon and their ecosystems: status and future options. New York: Chapman & Hall: 411–444.

Gaines, W.L.; Harrod, R.J.; Dickinson, J.D.; Lyons, A.L.; Halupka, K. 2010. Integration of northern spotted owl habitat and fuels treatments in the eastern Cascades, Washington, USA. Forest Ecology and Management. 260: 2045–2052.

Gaines, W.L.; Singleton, P.H.; Gold, A.L. 2000. Conservation of rare carnivores in the North Cascades Ecosystem, western North America. Natural Areas Journal. 20: 366-375.

Gaines, W.L.; Singleton, P.H.; Ross, R.C. 2003. Assessing the cumulative effects of linear recreation routes on wildlife habitats on the Okanogan and Wenatchee National Forests. Gen. Tech. Rep. PNW-GTR-586. Portland OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station. 79 p.

Gaines, W.L.; Wales, B.C.; Suring, L.H.; Begley, J.S.; Mellen-McLean, K.; Mohoric, S. [In press]. Terrestrial species viability assessments for the national forests in northeastern Washington. Gen. Tech. Rep. Portland OR: U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station.

Galbraith, W.A.; Anderson, E.W. 1991. Grazing history of the Northwest. Rangelands. 13: 213–218.

Garza, E.S. 1995. Fire history and fire regimes of East Humbug and Scorpion Creeks, and their relation to the range of *Pinus lambertiana* Dougl. Portland, OR: Oregon State University. 75 p. M.F. thesis.

Gedalof, Z.; Peterson, D.L.; Mantua, N.J. 2005. Atmospheric, climatic, and ecological controls on extreme wildfire years in the northwestern United States. Ecological Applications. 15: 154–174.

Goheen, D.J.; Filip, G.M. 1980. Root pathogen complexes in Pacific Northwest forests. Plant Disease. 64: 793–794.

Goheen, E.M.; Willhite, E.A. 2006. Field guide to common diseases and insect pests of Oregon and Washington conifers. R6-NR-FID-PR-01-06. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 327 p.

Graf, M.; Reid, M.L.; Aukema, B.H.; Lindgren, B.S. 2012. Association of tree diameter with body size and lipid content of mountain pine beetles. The Canadian Entomologist. 144: 467–477.

Grant, G.; Tague, C.; Allen, C.D. 2013. Watering the forest for the trees: an emerging priority for managing water in forest landscapes. Frontiers in Ecology and the Environment. 11: 314–321.

Gregory, S.V.; Swanson, F.J.; McKee, W.A.; Cummins, K.W. 1991. An ecosystem perspective of riparian zones: focus on links between land and water. BioScience. 41: 540–551.

Greenwald, D.N.; Crocker-Bedford, D.C.; Broberg, L.; Suckling, K.F.; Tibbitts, T. 2005. A review of northern goshawk habitat selection in the home range and implications for forest management in the western United States. Wildlife Society Bulletin. 33: 120–128.

Gresswell, R.E. 1999. Fire and aquatic ecosystems in forested biomes of North America. Transactions of the American Fisheries Society.128: 193–221.

Gresswell, R.E.; Liss, W.J.; Larson, G.L. 1994. Life-history organization of Yellowstone cutthroat trout (*Oncorhynchus clarki bouvieri*) in Yellowstone Lake. Canadian Journal of Fisheries and Aquatic Sciences. 51: 298–309.

Gude, P.H.; Rasker, R.; van den Noort, J. 2008. Potential for future development on fire-prone lands. Journal of Forestry. 106: 198–205.

Gunderson, L.H. 2000. Ecological resilience—in theory and application. Annual Review of Ecological Systems. 31: 425–439.

Gutierrez, R.J.; Cody, M.; Courtney, S.; Franklin, A.B. 2007. The invasion of barred owls and its potential effect on the spotted owl: a conservation conundrum. Biological Invasions. 9: 181–196.

Hadfield, J.S.; Goheen, D.G.; Filip, G.M.; Schmitt, C.L.; Harvey, R.D. 1986. Root diseases in Oregon and Washington conifers. R6-FPM-250-86. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Forest Pest Management. 27 p.

Hagar, J.C. 2007. Wildlife species associated with non-coniferous vegetation in Pacific Northwest conifer forests: A review. Forest Ecology and Management. 246: 108–122.

Hagmann, R.K.; Franklin, J.F.; Johnson, K.N. 2013. Historical structure and composition of ponderosa pine and mixed-conifer forests in south-central Oregon. Forest Ecology and Management. 304: 492–504.

Hall, T.E.; Heaton, H.; Kruger, L.E. 2009. Outdoor recreation in the Pacific Northwest and Alaska: trends in activity participation. Gen. Tech. Rep. PNW-GTR-778. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 108 p.

Halofsky, J.E.; Donato, D.C.; Hibbs, D.E.; Campbell, J.L.; Donaghy Cannon, M.; Fontaine, J.B.; Thompson, J.R.; Anthony, R.G.; Bormann, B.T.; Kayes, L.J.; Law, B.E.; Peterson, D.L.; Spies, T.A. 2011. Mixed-severity fire regimes: lessons and hypotheses from the Klamath-Siskiyou ecoregion. Ecosphere. 2(4): art40. doi: 10.1890/ES10-00184.1.

Hamlet, A.F.; Mote, P.W.; Clark, M.P.; Lettenmaier, D.P. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate. 18: 4545–4561.

Hamlet, A.F.; Mote, P.W.; Clark, M.P.; Lettenmaier, D.P. 2007. Twentieth-century trends in runoff, evapotranspiration, and soil moisture in the western United States. Journal of Climate. 20: 1468–1486.

Hann, W.J.; Jones, J.L.; Karl, M.G.; Hessburg, P.F.; Keane, R.E.; Long, D.G.; Menakis, J.P.; McNicoll, C.H.; Leonard, S.G.; Gravenmier, R.A.; Smith, B.G. 1997. An assessment of landscape dynamics of the basin. In: Quigley, T.M.; Arbelbide, S.J., eds. An assessment of ecosystem components in the Interior Columbia Basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 337–1055.

Hansen, A.J.; Knight, R.; Marzluff, J.; Powell, S.; Brown, K.; Hernandez, P.; Jones, K. 2005. Effects of exurban development on biodiversity: patterns, mechanisms, research needs. Ecological Applications. 15: 1893–1905.

Hansen, A.J.; Phillips, L.; Flather, C.H.; Robison-Cox, J. 2011. Carrying capacity for species richness as a context for conservation: a case study of North American breeding birds. Global Ecology and Biogeography. 20: 817–831.

Hansen, A.J.; Phillips, L.; Loehle, C.; Waring, R.H. 2006. Species energy theory and regional scale bird diversity: implications for conservation. Technical Report. Bozeman, MT: Montana State University, Landscape Biodiversity Lab.

Harrington, M.G. 1991. Fire management in interior Douglas-fir forests. In: Baumgartner, D.M.; Lotan, J.E., eds. Interior Douglas-fir: the species and its management. Proceedings of a symposium. Pullman, WA: Washington State University Cooperative Extension: 209–214.

Harris, C.; McLaughlin, W.; Brown, G.; Becker, D.R. 2000. Rural communities in the Inland Northwest: An assessment of small rural communities in the Interior and Upper Columbia River Basins. Gen. Tech. Rep. PNW-GTR-477. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station and U.S. Department of the Interior, Bureau of Land Management. 120 p.

Harris, J.A.; Hobbs, R.J.; Higgs, E.H.; Aronson, J. 2006. Ecological restoration and global climate change. Restoration Ecology. 14: 170–176.

Harrod, R.J.; McRae, B.H.; Hartl, W.E. 1999. Historical stand reconstruction in ponderosa pine forests to guide silvicultural prescriptions. Forest Ecology and Management. 114: 433–446.

Harvey, A.E.; Geist, J.M.; McDonald, G.I.; Jurgensen, M.F.; Cochran, P.H.; Zabowski, D.; Meurisse, R.T. 1994. Biotic and abiotic processes in eastside ecosystems: the effects of management on soil properties, processes, and productivity. In: Hessburg, P.F., ed., Eastside Forest Ecosystem Health Assessment. Gen. Tech. Rep. PNW-GTR-323. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 71 p.

Haugo, R.D.; Hall, S.A.; Gray, E.M.; Gonzalez, P.; Bakker, J.D. 2010. Influences of climate, fire, grazing, and logging on woody species composition along an elevation gradient in the eastern Cascades, Washington. Forest Ecology and Management. 260: 2204–2213.

Haugo, R.D.; Halpern, C.B. 2007. Vegetation responses to conifer encroachment in a western Cascade meadow: a chronosequence approach. Canadian Journal of Botany. 85: 285–298.

Hawksworth, F.G. 1958. Rate of spread and intensification of dwarf mistletoe in young lodgepole pine stands. Journal of Forestry. 56: 404–407.

Hawksworth, F.G. 1960. Distribution of ponderosa pine dwarf mistletoe in the vicinity of an Arizona volcano. Ecology. 41: 799–800.

Hawksworth, F.G.; Nicholls, T.H.; Merrill, L.M. 1987. Long-distance dispersal of lodgepole pine dwarf mistletoe. In: Troendle, C.A.; Kaufmann, M.R.; Hamre, R.H.; Winokur, R.P., tech. coords. Management of subalpine forests: building on 50 years of research. Gen. Tech. Rep. RM-GTR-149. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 220–226.

Hayes, J.L.; Daterman, G.E. 2001. Bark beetles (Scolytidae) in eastern Oregon and Washington. Northwest Science. 75: 21–30.

Hayes, J.L.; Ragenovich, I. 2001. Non-native invasive forest insects of eastern Oregon and Washington. Northwest Science. 75: 77–84.

Healey, M.C.; Prince, A. 1995. Scales of variation in life history tactics of Pacific salmon and the conservation of phenotype and genotype. In: Nielsen, J., ed. Evolution and the aquatic ecosystem: defining unique units in population conservation. Bethesda, MD: American Fisheries Society: 176–184.

Hedrick, D.W.; Young, J.; McArthur, J.; Keniston, R. 1968. Effects of forest and grazing practices on mixed coniferous forests of northeastern Oregon. Technical Bulletin 103. Corvallis, OR: Oregon State University Agricultural Experiment Station. 24 p.

Heinselman, M.L. 1973. Fire in the virgin forests of the Boundary Waters Canoe Area, Minnesota. Quaternary Research. 3: 329–382

Heller, N.E.; Zavaleta, E.S. 2009. Biodiversity management in the face of climate change: a review of 22 years of recommendations. Biological Conservation. 142: 14–32.

Henderson, J.A.; Lesher, R.D.; Peter, D.H.; Ringo, C.D. 2011. A landscape model for predicting potential natural vegetation of the Olympic Peninsula USA using boundary equations and newly developed environmental variables. Gen. Tech. Rep. PNW-GTR-841. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 35 p.

Hessburg, P.F.; Agee, J.K.; Franklin, J.F. 2005. Dry mixed-conifer forests and wildland fires of the Inland Northwest: contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management. 211: 117–139.

Hessburg, P.F.; James, K.M.; Salter, R.B. 2007. Re-examining fire severity relations in pre-management era mixed-conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology. 22: 5–24.

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

Hessburg, P.F.; Reynolds, K.M.; Salter, R.B.; Dickinson, J.D.; Gaines, W.L.; Harrod, R.J. 2013. Landscape evaluation for restoration planning on the Okanogan-Wenatchee National Forest, USA. Sustainability. 5: 805–840.

Hessburg, P.F.; Reynolds, K.M.; Salter, R.B.; Richmond, M.B. 2004. Using a decision support system to estimate departures of present forest landscape patterns from historical conditions: an example from the Inland Northwest Region of the United States. In: Perera, A.H.; Buse, L.J.; Weber, M.G., eds. Emulating natural forest landscape disturbances: concepts and applications. New York: Columbia University Press: 158–175.

Hessburg P.F.; Salter, R.B.; Richmond, M.B.; Smith, B.G. 2000a. Ecological subregions of the Interior Columbia Basin, USA. Applied Vegetation Management. 3: 163–180.

Hessburg, P.F.; Smith, B.G.; Kreiter, S.G.; Miller, C.A.; Salter, R.B.; McNicholl, C.H.; Hann, W.J. 1999a. Historical and current forest and range landscapes in the Interior Columbia River Basin and portions of the Klamath and Great Basins. Part 1. Linking vegetation patterns and landscape vulnerability to potential insect and pathogen disturbances. Gen. Tech. Rep. PNW-GTR-458. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 357 p.

Hessburg, P.F.; Smith, B.G.; Miller, C.A.; Kreiter, S.G.; Salter, R.B. 1999b. Modeling change in potential landscape vulnerability to forest insect and pathogen disturbances: methods for forested subwatersheds sampled in the mid-scale Interior Columbia River Basin assessment. Gen, Tech. Rep. PNW-GTR-454. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 56 p.

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

Hessburg, P.F.; Smith, B.G.; Salter, R.B. 1999d. Using natural variation estimates to detect ecologically important change in forest spatial patterns: a case study of the eastern Washington Cascades. Res. Pap. PNW-RP-514. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 65 p.

Hessburg, P.F.; Smith, B.G.; Salter, R.B.; Ottmar, R.D.; Alvarado, E. 2000b. Recent changes (1930s–1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management. 136: 53–83.

Hesseln, H.; Loomis, J.B.; Gonzalez-Caban, A. 2004. Comparing the economic effects of fire on hiking demand in Montana and Colorado. Journal of Forest Economics. 10: 21–35.

Hesseln, H.; Loomis, J.B.; Gonzalez-Caban, A.; Alexander, S. 2003. Wildfire effects on hiking and biking demand in New Mexico: a travel cost study. Journal of Environmental Management. 69: 359–368.

Hessl, A.E.; McKenzie, D.; Schellhaas, R. 2004. Fire and climatic variability in the inland Pacific Northwest. Ecological Applications. 14: 425–442.

Heyerdahl, E.K.; Brubaker, L.B.; Agee, J.K. 2001. Spatial controls of historical fire regimes: a multi-scale example from the Interior West, USA. Ecology. 82: 660–678.

Heyerdahl, E.K.; Falk, D.A.; Loehman, R.A. 2013. Data archived with the International Multiproxy Paleofire Database, IGBP PAGES/World Data Center for Paleoclimatology. Boulder, CO: NOAA/NCDC Paleoclimatology Program. http://www.ncdc.noaa.gov/paleo/impd/paleofire.html. (4 February 2014).

Heyerdahl, E.K.; McKenzie, D.; Daniels, L.D.; Hessl, A.E.; Littell, J.S.; Mantua, N.J. 2008. Climate drivers of regionally synchronous fires in the inland Northwest (1651–1900). International Journal of Wildland Fire. 17: 40–49.

Hicke, J.A.; Jenkins, J.C. 2008. Mapping lodgepole pine stand structure susceptibility to mountain pine beetle attack across the western United States. Forest Ecology and Management. 255: 1536–1547.

Hicke, J.A.; Johnson, M.C.; Hayes, J.L.; Preisler, H.K. 2012. Effects of bark beetle-caused tree mortality on wildfire. Forest Ecology and Management. 271: 81–90.

Hilty, J.A.; Lidicker, W.Z.; Merenlender, A.M. 2006. Corridor ecology: the science and practice of linking landscapes for biodiversity and conservation. Washington, DC: Island Press. 323 p.

Hjerpe, E.; Abrams, J.; Becker, D.R. 2009. Socioeconomic barriers and the role of biomass utilization in southwestern ponderosa pine restoration. Ecological Restoration. 27: 169–177.

Hobbs, R.J.; Hallett, L.M.; Ehrlich, P.R.; Mooney, H.A. 2011. Intervention ecology: applying ecological science in the twenty-first century. BioScience. 61: 442–450.

Hollenbeck, J.P.; Bate, L.J.; Saab, V.A.; Lehmkuhl, J.F. 2013. Snag distributions in relation to human access in ponderosa pine forests. Wildlife Society Bulletin. 37: 256–266.

Huff, M.H.; Ottmar, R.D.; Lehmkuhl, J.F.; Hessburg, P.F.; Everett, R.L.; Alvarado, E.; Vihanek, R.H. 1995. Historical and current forest landscapes of eastern Oregon and Washington. Part II: potential fire behavior and smoke production. In: Hessburg, P.F., ed., Eastside Forest Ecosystem Health Assessment. Gen. Tech. Rep. PNW-GTR-355. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 43 p.

Intergovernmental Panel on Climate Change [IPCC]. 2013. Summary for policymakers. In: Climate change 2013: the physical science basis. Stocker, D.F.; Qin, D.; Plattner, G.-K.; Tignor, M.; Allen, S.K.; Boschung, J.; Nauels, A.; Xia, Y.; Bex, V.; Midgley, P.M., eds. Working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom, and New York: Cambridge University Press. 27 p.

Intertribal Timber Council [ITC]. 2013. Fulfilling the promise of the Tribal Forest Protection Act of 2004, executive summary. Portland, OR: Intertribal Timber Council.

Intertribal Timber Council and Resource Innovations. 2006. Tribal wildfire resource guide. Eugene, OR: University of Oregon, Resource Innovations. 146 p.

Isaak, D.J.; Luce, C.H.; Rieman, B.E.; Nagel, D.E.; Peterson, E.E.; Horan, D.L.; Parkes, S.; Chandler, G. 2010. Effects of climate change and wildfire on stream temperatures and salmonid thermal habitat in a mountain river network. Ecological Applications. 20: 1350–1371.

Jain, T.B.; Battaglia, M.A.; Han, H.; Graham, R.T.; Keyes, C.R.; Fried, J.S.; Sandquist, J.E. 2012. A comprehensive guide to fuel management practices for dry mixed-conifer forests in the northwestern United States. Gen. Tech. Rep. RMRS-GTR-292. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 344 p.

Jain, S.; Lall, U. 2001. Floods in a changing climate: Does the past represent the future? Water Resources Research. 37: 3193–3206.

Jenkins, M.J.; Runyon, J.B.; Fettig, C.J.; Page, W.G.; Bentz, B.J. 2014. Interactions among the mountain pine beetle, fires, and fuels. Forest Science. 60: 489–501.

Johnson, C. 2007. Rangeland exclosures of northeastern Oregon: stories they tell (1936–2004). Gen. Tech. Rep. PNW-GTR-724. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 33 p.

Johnson, C.G., Jr. 2004. Alpine and subalpine vegetation of the Wallowa, Seven Devils, and Blue Mountains. R6-NR-ECOL-TP-03-04. Portland OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 612 p.

Johnson, C.G., Jr.; Clasunitzer, R.R. 1992. Plant associations of the Blue and Ochoco Mountains. R6-ERW-TP-036-92. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 208 p.

Johnson, C.G.; Clausnitzer, R.R.; Mehringer, P.J.; Oliver, C.D. 1994. Biotic and abiotic processes in eastside ecosystems: the effects of management on plant and community ecology, and on stand and landscape vegetation dynamics. Gen. Tech. Rep. PNW-GTR-322. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 66 p.

Johnson, C.G., Jr.; Simon, S.A. 1987. Plant associations of the Wallowa-Snake Province. R6-ECOL-TP-255A-86. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 400 p. plus appendixes.

Johnson, D.H. 1980. The comparison of usage and availability measurements for evaluating resource preference. Ecology. 61: 65–71.

Johnson, D.J.; O'Neil, T.A. 2001. Wildlife-habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State University Press. 736 p.

Jolly, W.M.; Parsons, R.A.; Hadlow, A.M.; Cohn, G.M.; McAllister, S.S.; Popp, J.B.; Hubbard, R.M.; Negrón, J.F. 2012. Relationships between moisture, chemistry, and ignition of *Pinus contorta* needles during the early stages of mountain pine beetle attack. Forest Ecology and Management. 269: 52−59.

Kauffman, J.B. 1988. The status of riparian habitats in Pacific Northwest forests. In: Raedeke, K.J., ed. Streamside management: riparian wildlife and forestry interactions. College of Forest Resources Institute of Forest Resources Contribution 59. Seattle, WA: University of Washington: 45–55.

Keane, R.E.; Hessburg, P.F.; Landres, P.B.; Swanson, F.J. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management. 258: 1025–1037.

Keegan III, C.E.; Morgan, T.A.; Gebert, K.M.; Brandt, J.P.; Blatner, K.A.; Spoelma, T.P. 2006. Timber-processing capacity and capabilities in the western United States. Journal of Forestry. 104: 262–268.

Kellogg, L.K.B.; McKenzie, D.; Peterson, D.L.; Hessl, A.E. 2008. Spatial models for inferring topographic controls on historical low-severity fire in the eastern Cascade Range of Washington, USA. Landscape Ecology. 23: 227–240.

Kemp, W.P.; Everson, D.O.; Wellington, W.G. 1985. Regional climate patterns and western spruce budworm outbreaks. Tech. Bull. 1693. Washington, DC: U.S. Department of Agriculture, Forest Service, Cooperative State Research Service. 31 p.

Kennedy, P.L.; Fontaine, J.B. 2009. Synthesis of knowledge on the effects of fire and fire surrogates on wildlife in U.S. dry forests. Special Report 1096. Corvallis, OR: Oregon State University, Agricultural Experiment Station. 132 p.

Kiffney, P.M.; Greene C.M.; Hall J.E.; Davies J.R. 2006. Tributary streams create spatial discontinuities in habitat, biological productivity and diversity in mainstem rivers. Canadian Journal of Fisheries and Aquatic Sciences. 63: 2518–2530.

Kline, J.D. 2004. Issues in evaluating the costs and benefits of fuel treatments to reduce wildfire in the nation's forests. Res. Note PNW-RN-542. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 46 p.

Kline, J.D.; Mazzotta, M.J. 2012. Evaluating tradeoffs among ecosystem services in the management of public lands. Gen. Tech. Rep. PNW-GTR-865. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 48 p.

Klutsch, J.G.; Battaglia, M.A.; West, D.R.; Costello, S.L.; Negrón, J.F. 2011. Evaluating potential fire behavior in lodgepole pine-dominated forests after a mountain pine beetle epidemic in north-central Colorado. Western Journal of Applied Forestry. 26: 101–109.

Koonce, A.L.; Roth, L.F. 1980. The effects of prescribed burning on dwarf mistletoe in ponderosa pine. In: Proceedings, sixth conference on fire and forest meteorology.. Washington, DC: Society of American Foresters: 197–203.

Kotliar, N.B.; Hejl, S.J.; Hutto, R.L.; Saab, V.A.; Melcher, C.P.; McFadzen, M.E. 2002. Effects of fire and post-fire salvage logging on avian communities in conifer-dominated forests of the western United States. In: George, T.L.; Dobkin, D.S., eds. Effects of habitat fragmentation on birds in western landscapes: contrasts with paradigms from the eastern United States. Studies in Avian Biology No. 25. Camarillo, CA: Cooper Ornithological Society: 49–64.

Kruse, C.G.; Hubert, W.A.; Rahel, F.J. 2001. An assessment of headwater isolation as a conservation strategy for cutthroat trout in the Absaroka Mountains of Wyoming. Northwest Science. 75: 1–11.

Langston, N. 1995. Forest dreams, forest nightmares: the paradox of old growth in the Inland Northwest. Seattle, WA: University of Washington Press. 368 p.

Larson, A.J.; Churchill, D.C. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management. 267: 74–92.

Latta, G.; Temesgen, H.; Adams, D.; Barrett, T. 2010. Analysis of potential impacts of climate change on forests of the United States Pacific Northwest. Forest Ecology and Management. 259: 720–729.

Lee, D.; Sedell, J.; Rieman, B.; Thurow, R.; Williams, J. 1997. Broad-scale assessment of aquatic species and habitats. In: Quigley, T.M.; Arbelbide, S.J., eds. An assessment of ecosystem components in the Interior Columbia Basin and portions of the Klamath and Great Basins, vol. 3. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1057–1430.

Lehmkuhl, J.F. 2005. Wildlife adaptations and management in eastside interior forests with mixed-severity fire regimes. Proceedings of the symposium on mixed-severity fire regimes: ecology and management, Spokane, Washington, November 17–19, 2004. Misc. Pub. 3. Davis, CA: The Association for Fire Ecology: 177–186.

Lehmkuhl, J.F.; Burger, E.D.; Drew, E.K.; Lindsey, J.P.; Haggard, M.; Woodruff, K.Z. 2007a. Breeding birds in riparian and upland dry forests of the Cascade Range. Journal of Wildlife Management. 71: 2632–2643.

Lehmkuhl, J.F.; Hessburg; P.F.; Everett; R.L.; Huff; M.H.; Ottmar, R.D. 1994. Historical and current vegetation pattern and associated changes in insect and disease hazard conditions in eastern Oregon and Washington. In: Hessburg, P.F., ed., Eastside Forest Ecosystem Health Assessment: Volume III: Assessment. Gen. Tech. Rep. PNW-GTR-328. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 88 p.

Lehmkuhl, J.F.; Kennedy, M.; Ford, E.D.; Singleton, P.H.; Gaines, W.L.; Lind, R.L. 2007b. Seeing the forest for the fuel: integrating ecological values and fuels management. Forest Ecology and Management. 246: 73–80.

Lehmkuhl, J.F.; Kistler, K.D.; Begley, J.S. 2006a. Bushy-tailed woodrat abundance in dry forests of eastern Washington. Journal of Mammalogy. 87: 371–379.

Lehmkuhl J.F.; Kistler, K.D.; Begley, J.S.; Boulanger, J. 2006b. Demography of northern flying squirrels informs ecosystem management of western interior forests. Ecological Applications. 16: 584–600.

Lehmkuhl, J.F.; Lyons, A.L.; Bracken, E.; Leingang, J.; Gaines, W.L.; Dodson, E.K.; Singleton, P.H. 2013. Forage composition, productivity, and utilization in the eastern Washington Cascade Range. Northwest Science. 87: 267–291.

Lehmkuhl, J.F.; Peffer, R.D.; O'Connell, M.A. 2008. Riparian and upland small mammals on the east slope of the Cascade Range, Washington. Northwest Science. 82: 94–107.

Lewis, J.C.; Stinson, D.W. 1998. Washington State status report for the fisher. Olympia, WA: Washington Department of Fish and Wildlife. 64 p.

Lillybridge, T.R.; Kovalchik, B.L.; Williams, C.K.; Smith, B.G. 1995. Field guide for forested plant associations of the Wenatchee National Forest. Gen. Tech. Rep. PNW-GTR-359. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station and Pacific Northwest Region, Wenatchee National Forest. 352 p.

Lindenmayer, D.; Burton, P.J.; Franklin, J.F. 2008. Salvage logging and its ecological consequences. Washington, DC: Island Press. 248 p.

Littell, J.S.; McKenzie, D.; Peterson, D.L.; Westerling, A.L. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications. 19: 1003–1021.

Littell, J.S.; O'Neil, E.E.; McKenzie, D.; Hicke, J.A.; Lutz, J.A.; Norheim, R.A.; Elsner, M.M. 2010. Forest ecosystems, disturbance, and climatic change in Washington State, USA. Climatic Change. 102: 129–158

Loehle, C. 2011. Criteria for assessing climate change impacts on ecosystems. Ecology and Evolution 1: 63–72.

Long, R.A.; Rachlow, J.L.; Kie, J.G.; Vavra, M. 2008. Fuels reduction in a western coniferous forest: effects on quantity and quality of forage for elk. Rangeland Ecology and Management. 61: 302–313.

Loomis, J.; Gonzalez-Caban, A.; Englin, J. 2001. Testing for differential effects of forest fires on hiking and mountain biking demand and benefits. Journal of Agricultural and Resource Economics. 26: 508–522.

Lotan, J.E.; Alexander, M.E.; Arno, S.F.; French, R.E.; Langdon, O.G.; Loomis, R.M.; Morum, R.A.; Rothermel, R.C.; Schmidt, W.C.; van Wagtendonk, J. 1981. Effects of fire on flora: a state of knowledge review. Gen. Tech. Rep. WO-GTR-16. Washington, DC: U.S. Department of Agriculture, Forest Service. 71 p.

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

Luce, C.; Morgan, P.; Dwire, K.; Isaak, D.; Holden, Z.; Rieman, B. 2012. Climate change, forests, fire, water, and fish: Building resilient landscapes, streams, and managers. Gen. Tech. Rep. RMRS-GTR-290. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 207 p.

Luce, C.H.; Rieman, B.E.; Dunham, J.B.; Clayton, J.L.; King, J.G.; Black, T.A. 2001. Incorporating aquatic ecology into decisions on prioritization of road decommissioning. Water Resources Impact. May: 8–14.

Luce, C.; Staab, B.; Kramer, M.; Wenger, S.; Isaak, D.; McConnell, C. 2014. Sensitivity of summer stream temperatures to climate variability in the Pacific Northwest. Water Resources Research 50, doi:10.1002/2013WR014329.

Ludwig, D.; Hilborn, R.; Walters, C. 1993. Uncertainty, resource exploitation, and conservation: lessons from history. Science. 260: 35–36.

Lyon, L.; Christensen, A.G. 2002. Elk and land management. In: Toweill, D.E.; Thomas, J.W., eds., North American Elk: Ecology and Management. Washington, DC: Wildlife Management Institute: 531–556.

Lyons, A.L.; Gaines, W.L.; Lehmkuhl, J.F.; Harrod, R.J. 2008. Short-term effects of fire and fire surrogate treatments on foraging tree selection by cavity-nesting birds in dry forests of central Washington. Forest Ecology and Management. 255: 3201–3211.

Lyons, A.L.; Gaines, W.L.; Servheen, C. 2003. Black bear resource selection in the northeast Cascades, Washington. Biological Conservation. 113: 55–62.

Magis, K. 2010. Community resilience: an indicator of social sustainability. Society and Natural Resources. 23: 401–416.

Malamud, B.D.; Millington, J.D.A.; Perry, G.L.W. 2005. Characterizing wildfire regimes in the United States. Proceedings of the National Academy of Sciences of the United States of America. 102: 4694–4699.

Malamud, B.D.; Morein, G.; Turcotte, D.L. 1998. Forest fires: an example of self-organized critical behavior. Science. 281: 1840–1842.

Mantua, N.J.; Hare, S.R. 2002. The Pacific decadal oscillation. Journal of Oceanography. 58: 35–44.

Mantua, N.J.; Hare, S.R.; Zhang, Y.; Wallace, J.M.; Francis, R.C. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society. 78: 1069–1079.

Marcot, B.G.; Mellen, K.; Livingston, S.A.; Ogden, C. 2002. The DecAID advisory model: wildlife component. In: Shea, P.J.; Laudenslayer Jr., W.F.; Valentine, B.; Weatherspoon, C.P., eds. Proceedings of the symposium on the ecology and management of dead wood in western forests. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 561–590.

Margerum, R.D. 2011. Beyond consensus: improving collaborative planning and management. Cambridge, MA: The MIT Press. 416 p.

Marsh, R.; Helliwell, R.; Rodgers, J. 1987. Plant association guide for the commercial forest of the Warm Springs Indian reservation. Warm Springs, OR: Warm Springs Indian Reservation. 142 p.

Martin, K.; Aitken, K.E.H.; Wiebe, K.L. 2004. Nest sites and nest webs for cavity-nesting communities in interior British Columbia. The Condor. 106: 5–19.

Maser, C.; Anderson, R.G.; Cromak Jr., K.; Williams, J.T.; Martin, R.E. 1979. Dead and down woody material. In: Thomas, J.W., ed. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agricultural Handbook 553. Washington, DC: U.S. Department of Agriculture, Forest Service: 78–95.

McAllister, K.R.; Leonard, W.P. 1997. Washington State status report for the Oregon spotted frog. Olympia, WA: Washington Department of Fish and Wildlife. 38 p.

McCaffrey, S.; Toman, E.; Stidham, M.; Shindler, B. 2013. Social science research related to wildfire management: an overview of recent findings and future research needs. International Journal of Wildland Fire. 22: 15–24.

McComb, W.C. 2001. Management of within-stand forest habitat features. In: Johnson, D.H.; O'Neil, T.A., managing directors. Wildlife–habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State University Press: 140–153.

McDermott, M.H.; Moote, M.A.; Danks, C. 2011. Effective collaboration: overcoming external obstacles. In: Dukes, E.F.; Firehock, K.E.; Birkhoff, J.E., eds. Community-based collaboration: bridging socio-ecological research and practice. Charlottesville: University of Virginia Press: 81–110.

McGarigal, K.; Cushman, S.A.; Neel, M.C.; Ene, E. 2002. FRAGSTATS 3.0: spatial pattern analysis program for categorical maps. Amherst, MA: University of Massachusetts. http://www.umass.edu/landeco/research/fragstats/documents/fragstats_documents.html. (July 3, 2014).

McGlone, C.M.; Springer, J.D.; Laughlin, D.C. 2009. Can pine forest restoration promote a diverse and abundant understory and simultaneously resist nonnative invasion? Forest Ecology and Management. 258: 2638–2646.

McIver, J.; Stephens, S.; Agee, J.; Barbour, J.; Boerner, R.; Edminster, C.; Erickson, K.; Farris, K.; Fettig, C.; Fiedler, C.; Haase, S.; Hart, S.; Keeley, J.; Knapp, E.; Lehmkuhl, J.; Moghaddas, J.; Otrosina, W.; Outcalt, K.; Schwilk, D.; Skinner, C.; Waldrop, T.; Weatherspoon, P.; Yaussy, D.; Youngblood, A.; Zack, S. 2012. Ecological effects of alternative fuel reduction treatments: highlights of the national Fire and Fire Surrogate study (FFS). International Journal of Wildland Fire. 22: 63–82.

McKelvey, K.S.; Copeland, J.P.; Schwartz, M.K.; Littell, J.S.; Aubry, K.B.; Squires, J.R.; Parks, S.A.; Elsner, M.M.; Mauger, G.S. 2011. Climate change predicted to shift wolverine distributions, connectivity, and dispersal corridors. Ecological Applications. 21: 2882–2897.

McKinney, M.L. 2002. Urbanization, biodiversity, and conservation. BioScience. 52: 883–890.

Medhurst, R.B.; Wipfli, M.S.; Binckley, C.; Polivka, K.M.; Hessburg, P.F.; Salter, R.B. 2010. Headwater streams and forest management: Does ecoregional context influence logging effects on benthic communities? Hydrobiologia. 641: 71–83.

Merschel, A.G. 2012. Mixed-conifer forests of central Oregon: structure, composition, history of establishment, and growth. Portland, OR: Oregon State University. 167 p. M.S. thesis.

Meyer, G.A.; Pierce, J. 2003. Climatic control of fire-induced sediment pulses in Yellowstone National Park and Central Idaho: a long-term perspective. Forest Ecology and Management. 178: 89–104.

Meyer, G.A.; Wells, S.G.; Balling Jr., R.C.; Jull, A.J.T. 1992. Response of alluvial systems to fire and climate change in Yellowstone National Park. Nature. 357: 147–150.

Miller, D.; Luce, C.; Benda, L. 2003. Time, space, and episodicity of physical disturbance in streams. Forest Ecology and Management. 178: 121–140.

Miller, J.D.; Skinner, C.N.; Safford, H.D.; Knapp, E.E.; Ramirez, C.M. 2012. Trends and causes of severity, size, and number of fires in northwestern California, USA. Ecological Applications. 22: 184–203.

Minshall, G.W. 2003. Responses of stream benthic macroinvertebrates to fire. Forest Ecology and Management. 178: 155–161.

Mitchell, S.J.; Beese, W.J. 2002. The retention system: reconciling variable retention with the principles of silvicultural systems. The Forestry Chronicle. 78: 397–403.

Morgan, T.A.; Brandt, J.P.; Baldridge, J.D.; Loeffler, D.R. 2011. Use of financial and economic analyses by federal forest managers for woody biomass removal. Western Journal of Applied Forestry. 26: 5–12.

Moritz, M.A.; Hessburg, P.F.; Povak, N.A. 2011. Native fire regimes and landscape resilience. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 51–86.

Moritz, M.A.; Hurteau, M.D.; Suding, K.N.; D'Antonio, C.M. 2013. Bounded ranges of variation as a framework for future conservation and fire management. Annals of the New York Academy of Sciences. 1286: 92–107.

Morrison, D.J.; Williams, R.E.; Whitney, R.D. 1991. Infection, disease development, diagnosis, and detection. In: Shaw, C.G., III; Kile, G.A., eds. Armillaria root disease. Agric. Handb. 691. Washington, DC: U.S. Department of Agriculture, Forest Service: 62–75.

Morrison, M.L.; Marcot, B.G.; Mannan, R.W. 1998. Wildlife-habitat relationships: concepts and applications. 2nd ed. Madison, WI: University of Wisconsin Press. 435 p.

Mote, P.W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30: 1601. doi:10.1029/2003GL017258.

Mote, P.W. 2009. Update to data originally published in: Mote, P.W.; Hamlet, A.F.; Clark, M.P.; Lettenmaier, D.P. 2005. Declining mountain snowpack in Western North America. Bulletin of the American Meteorological Society. 86: 39–49.

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

Mote, P.W.; Parson, E.A.; Hamlet, A.F.; Ideker, K.N.; Keeton, W.S.; Lettenmaier, D.P.; Mantua, N.; Miles, E.L.; Peterson, D.W.; Peterson, D.L.; Slaughter, R.; Snover, A.K. 2003. Preparing for climate change: the water, salmon, and forests of the Pacific Northwest. Climatic Change. 61: 45–88.

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

Munger, T.T. 1917. Western yellow pine in Oregon. USDA Bulletin No. 418. Washington, D.C.: U.S. Government Printing Office. 48 p.

Munzing, D.; Gaines, W.L. 2008. Monitoring American marten on the east-side of the North Cascades of Washington. Northwestern Naturalist. 89: 67–75.

Naiman, R.J.; Bilby, R.E.; Bisson, P.A. 2000. Riparian ecology and management in the Pacific coastal rain forest. Bioscience 50: 996–1011.

Naiman, R.J.; Decamps, H. 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics. 28: 621–658.

Naiman, R.J.; Decamps, H.; Pollock, M. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications. 3: 209–212.

Naiman, R.J.; Fetherston, K.L.; McKay, S.J.; Chen, J. 1998. Riparian forests. In: Naiman, R.J.; Bilby, R.E., eds. River ecology and management: lessons from the Pacific Coastal Ecoregion. New York: Springer-Verlag: 289–323.

Nakano, S.; Miyasaka, H.; Kuhara, N. 1999. Terrestrial-aquatic linkages: riparian arthropod inputs alter trophic cascades in a stream food web. Ecology 80: 2435–2441.

National Research Council (NRC). 1992. Restoration of aquatic ecosystems. Washington, DC: The National Academy Press. 532 p.

National Research Council (NRC). 1996. Upstream: salmon and society in the Pacific Northwest. Washington, DC: The National Academy Press. 472 p.

Naylor, L.M.; Wisdom, M.J.; Anthony. R.G. 2009. Behavioral responses of North American elk to recreational activity. Journal of Wildlife Management. 73: 328–338.

Nechodom, M.; Becker, D.R.; Haynes, R. 2008. Evolving interdependencies of community and forest health. In: Donoghue, E.M.; Sturtevant, V.E., eds. Forest community connections: implications for research, management, and governance. Washington, DC: Resources for the Future: 91–108.

Nelson, K.; Monroe, M.; Johnson, J.; Bowers, A. 2004. Living with fire: homeowner assessment of landscape values and defensible space in Minnesota and Florida, USA. International Journal of Wildland Fire. 13: 413–425.

Neville, H.; Dunham, J.D.; Rosenberger, A.; Umek, J.; Wilson, B. 2009. Influences of wildfire, habitat size, and connectivity on trout in headwater streams revealed by patterns of genetic diversity. Transactions of the American Fisheries Society 138: 1314–1327.

Nicholls, D.L.; Monserud, R.A.; Dykstra, D.P. 2008. A synthesis of biomass utilization for bioenergy production in the western United States. Gen. Tech. Rep. PNW-GTR 753. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 52 p.

Nielsen-Pincus, M.; Charnley, S.; Moseley, C. 2012. The influence of market proximity on national forest hazardous fuels treatments. Forest Science. 59: 566–577.

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

North, M. 2012. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 184 p.

North, M.P.; Collins, B.M.; Stephens, S.L. 2012. Using fire to increase the scale, benefits and future maintenance of fuels treatments. Journal of Forestry. 110: 392–401.

North M.; Stine, P.; O'Hara, K.; Zielinski, W.; Stephens, S. 2009. An ecosystem management strategy for Sierran mixed-conifer forests. Gen. Tech. Rep. PSW-GTR-220. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 49 p.

Nuñez, T.A.; Lawler, J.J.; McRae, B.H.; Pierce, D.J.; Krosby, M.B.; Kavanagh, D.M.; Singleton, P.H.; Tewksbury, J.J. 2013. Connectivity planning to address climate change. Conservation Biology. 27: 407–416.

O’Hara, K.L. 1996. Dynamics and stocking-level relationships of multi-aged ponderosa pine stands. Forest Science. 42: 1–34, Suppl. 33.

O’Hara, K.L. 1998. Silviculture for structural diversity: a new look at multi-aged systems. Journal of Forestry. 96: 4–10.

O’Hara, K.L.; Gersonde, R.F. 2004. Stocking control concepts in uneven-aged silviculture. Forestry. 77: 131–143.

O’Hara, K.L.; Kollenberg, C.L. 2003. Stocking control procedures for multi-aged lodgepole pine stands in the northern Rocky Mountains. Western Journal of Applied Forestry. 18: 15–21.

O’Hara, K.L.; Latham, P.A.; Hessburg, P.; Smith, B.G. 1996. A structural classification for inland northwest forest vegetation. Western Journal of Applied Forestry. 11: 97–102.

O’Hara, K.L.; Leonard, L.; Keyes, C.R. 2012. Variable-density thinning and a marking paradox: comparing prescription protocols to attain stand variability in coast redwood. Western Journal of Applied Forestry. 27: 143–149.

O’Hara, K.L.; Nagel, L.M. 2013. The stand: revisiting a central concept in forestry. Journal of Forestry. 111: 335–340.

O'Hara, K.L.; Ramage, B.S. 2013. Silviculture in an uncertain world: utilizing multi-aged management systems to integrate disturbance. Forestry. doi: 10.1093/forestry/cpt012.

O'Hara, K.L.; Valappil, N.I. 1999. MASAM: a flexible stand density management model for meeting diverse structural objectives in multi-aged stands. Forest Ecology and Management. 118: 57–71.

O'Hara, K.L.; Valappil, N.I.; Nagel, L.M. 2003. Stocking control procedures for multi-aged ponderosa pine stands in the Inland Northwest. Western Journal of Applied Forestry. 18: 5–14.

Okanogan-Wenatchee National Forest [OWNF]. 2012. The Okanogan-Wenatchee National Forest Restoration Strategy: adaptive ecosystem management to restore landscape resiliency. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 119 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5340103.pdf. (17 January 2014).

Oliver, C.D.; Larson, B.C. 1996. Forest stand dynamics-update edition. New York: John Wiley & Sons. 544 p.

Olsen, C.S.; Mallon, A.L.; Shindler, B. 2012. Public acceptance of disturbance-based forest management: factors influencing support. Forestry 2012: Article ID 594067, 10 p. doi:10.5402/2012/594067.

Olson, D.H.; Hagar, J.; Carey, A.; Cissel, J.; Swanson, F. 2001. Wildlife of westside and high montane forests. In Johnson, D.H.; O'Neil, T.A., eds. Wildlife-habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State University Press: 187–212.

Olson, D.L. 2000. Fire in riparian zones: a comparison of historical fire occurrence in riparian and upslope forests in the Blue Mountains and southern Cascades of Oregon. Seattle, WA: University of Washington. 274 p. M.S. thesis.

Olson, D.L; Agee, J.K. 2005. Historical fires in Douglas-fir dominated riparian forests of the southern Cascades, Oregon. Fire Ecology. 1: 50–74.

Oregon Forest Resources Institute [OFRI]. 2012. The 2012 forest report: an economic assessment of Oregon's forest and wood products manufacturing sector. 181 p. http://theforestreport.org/downloads/2012_Forest_Report.pdf. (17 January 2014).

Ormsbee, P.C.; McComb, W.C. 1998. Selection of day roosts by female long-legged myotis in the central Oregon Cascade Range. Journal of Wildlife Management. 62: 596–603.

Ostergren, D.M.; Abrams, J.B.; Lowe, K.A. 2008. Fire in the forest: public perceptions of ecological restoration in north-central Arizona. Ecological Restoration. 26: 51–60.

Otrosina, W.J.; Cobb Jr., F.W. 1989. Biology, ecology, and epidemiology of *Heterobasidion annosum*. In: Otrosina, W.J.; Scharpf, R.F., eds. Research and management of annosus root disease (*Heterobasidion annosum*) in western North America. Gen. Tech. Rep. PSW-GTR-116. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 26–33.

Page, W.G.; Jenkins, M.J. 2007. Predicted fire behavior in selected mountain pine beetle infested lodgepole pine. Forest Science. 53: 662−674.

Page, W.G.; Jenkins, M.J.; Runyon, J.B. 2012. Mountain pine beetle attack alters the chemistry and flammability of lodgepole pine foliage. Canadian Journal of Forest Research. 42: 1631−1647.

Pan, F.; Han, H.-S.; Johnson, L.R.; Elliot, W.J. 2008. Production and cost of harvesting, processing, and transporting small-diameter (≤ 5 inches) trees for energy. Forest Products Journal. 58: 47−53.

Parks, C.G.; Bull, E.L.; Tinnin, R.O.; Shepherd, J.F.; Blumton, A.K. 1999. Wildlife use of dwarf mistletoe brooms in Douglas-fir in northeast Oregon. Western Journal of Applied Forestry. 14: 100−105.

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

Parmeter, J.R., Jr. 1978. Forest stand dynamics and ecological factors in relation to dwarf mistletoe spread, impact, and control. In: Scharpf, R.F.; Parmeter, J.R., Jr., eds. Dwarf mistletoe control through forest management. Gen. Tech. Rep. PSW-GTR-31. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station: 16–30.

Pederson, J.L.; Smith, G.A.; Pazzaglia, F.J. 2001. Comparing the modern, quaternary, and neogene records of climate-controlled hillslope sedimentation in southeast Nevada. Geological Society of America Bulletin. 113: 305–319.

Peñuelas, J.; Canadell, J.G.; Ogaya, R. 2011. Increased water-use efficiency during the 20th century did not translate into enhanced tree growth. Global Ecology and Biogeography. 20: 597–608.

Perry, D.A.; Hessburg, P.F.; Skinner, C.N.; Spies, T.A.; Stephens, S.L.; Taylor, A.H.; Franklin, J.F.; McComb, B.; Riegel, G. 2011. The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management. 262: 703–717.

Perry, D.A.; Jing, H.; Youngblood, A.; Oetter, D.R. 2004. Forest structure and fire susceptibility in volcanic landscapes of the Eastern High Cascades, Oregon. Conservation Biology. 18: 913–926.

Perry, D.A.; Ram, O.; Hart, S.C. 2008. Forest ecosystems. 2nd edition. Baltimore, MD: Johns Hopkins University Press. 632 p.

Pess, G.R.; Montgomery, D.R.; Steel, E.A.; Bilby, R.E.; Feist, B.E.; Greenberg, H.M. 2002. Landscape characteristics, land use, and coho salmon (*Oncorhynchus kisutch*) abundance, Snohomish River, Washington, USA. Canadian Journal of Fisheries and Aquatic Sciences. 59: 613–623.

Peterson, G.D. 2002. Contagious disturbance, ecological memory, and the emergence of landscape pattern. Ecosystems. 5: 329–338.

Peterson, G.D.; Allen, C.R.; Holling, C.S. 1998. Ecological resilience, biodiversity, and scale. Ecosystems. 1: 6–18.

Pfister, R.D.; Arno, S.F. 1980. Classifying forest habitat types based on potential climax vegetation. Forest Science. 26: 52–70.

Pickett S.T.A.; Cadenasso, M.L.; Grove, J.M.; Nilson, C.H.; Pouyat, R.V.; Zipperer, W.C.; Costanza, R. 2001. Urban ecological systems: linking terrestrial ecological, physical, and socioeconomic components of metropolitan areas. Annual Review of Ecology and Systematics. 32: 127–157.

Pilliod, D.S.; Bull, E.L.; Hayes, J.L.; Wales, B.C. 2006. Wildlife and invertebrate response to fuel reduction treatments in dry coniferous forests of the Western United States: a synthesis. Gen. Tech. Rep. RMRS-GTR-173. Fort Collins, CO: U. S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 34 p.

Poff, N.L.; Allan, J.D.; Bain, M.B.; Karr, J.R.; Prestegaard, K.L.; Richter, B.D.; Sparks, R.E.; Stromber, J.C. 1997. The natural flow regime: a paradigm for river conservation and restoration. Bioscience. 47: 769–784.

Poff, N.L.; Brinson, M.M.; Day Jr., J.W. 2002. Aquatic ecosystems and global climate change: potential impacts on inland freshwater and coastal wetland ecosystems in the United States. Report from the Pew Center on Global Climate Change. http://www.c2es.org/docUploads/aquatic.pdf. (17 January 2014).

Polivka, K.M.; Friedli, L.M.; Green, E.C. 2013. Stream inflow and predation risk affect littoral habitat selection by benthic fish. Freshwater Biology. 58: 986–994.

Poole, G.; Dunham, J.; Hicks, M.; Keenan, D.; Lockwood, J.; Materna, E.; McCullough, D.; Mebane, C.; Risley, J.; Sauter, S.; Spalding, S.; Sturdevant, D. 2001. Scientific issues relating to temperature criteria for salmon, trout, and char native to the Pacific Northwest. A Summary Report submitted to the Policy Workgroup of the EPA Region 10 Water Temperature Criteria Guidance Project, EPA Region 10 Office, Seattle, WA.

Powell, D.C.; Johnson Jr., C.G.; Crowe, E.A.; Wells, A.; Swanson, D.K. 2007. Potential vegetation hierarchy for the Blue Mountains section of northeastern Oregon, southeastern Washington, and west-central Idaho. Gen. Tech. Rep. PNW-GTR-709. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 87 p.

Preisler, H.K.; Hicke, J.A.; Ager, A.A.; Hayes, J.L. 2012. Climate and weather influences on spatial temporal patterns of mountain pine beetle populations in Washington and Oregon. Ecology. 93: 2421–2434.

Radeloff, V.C.; Hammer, R.B.; Stewart, S.I.; Fried, J.S.; Holcomb, S.S.; McKeefry, J.F. 2005. The wildland–urban interface in the United States. Ecological Applications. 15: 799–805.

Rainville, R.; White, R.; Barbour, J., tech. eds. 2008. Assessment of timber availability from forest restoration within the Blue Mountains of Oregon. Gen. Tech. Rep. PNW-GTR-752. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 65 p.

Raish, C.; Gonzalez-Caban, A.; Martin, W.; Martin, I.; Bender, H. 2007. Cultural variation in public perceptions concerning fire use and management. In: Daniel, T.C.; Carroll, M.S.; Moseley, C.; Raish, C. eds. People, fire, and forests. Corvallis, OR: University of Oregon Press: 70–88.

Rasmussen, K.; Hibbard, M.; Lynn, K. 2007. Wildland fire management as conservation-based development:an opportunity for reservation communities? Society and Natural Resources. 20: 497–510.

Reeves, G.H.; Benda, L.E.; Burnett, K.M.; Bisson, P.A.; Sedell, J.R. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. In: Nielsen, J., ed. Evolution and the aquatic ecosystem, Proceedings of the 17th Symposium of the Bethesda, MD: American Fisheries Society: 334–349.

Reeves, G.H.; Bisson, P.A.; Dambacher, J.M. 1998. Fish communities. In: Naiman, R.J.; Bilby, R.E., eds. River ecology and management: lessons from the Pacific Coastal ecoregion. New York: Springer-Verlag: 200–234.

Rehfeldt, G.E.; Crookston, N.L.; Saenz-Romero, C.; Campbell, E.M. 2012. North American vegetation model for land-use planning in a changing climate: a solution to large classification problems. Ecological Applications. 22: 119–141.

Reid, M.L.; Purcell, J.R.C. 2011. Condition-dependent tolerance of monoterpenes in an insect herbivore. Arthropod-Plant Interactions 5: 331–337.

Reyna, N.E. 1998. Economic and social characteristics of communities in the Interior Columbia Basin. In: Interior Columbia Basin Ecosystem Management Project, Economic and Social Conditions of Communities. Walla Walla, WA: U.S. Department of Agriculture, Forest Service and U.S. Department of Interior, Bureau of Land Management: 3–81.

Reynolds, K.M.; Hessburg, P.F. 2005. Decision support for integrated landscape evaluation and restoration planning. Forest Ecology and Management. 207: 263–278.

Reynolds, K.M.; Rodriguez, S.; Bevans, K. 2003. User guide for the ecosystem management decision support system, version 3.0. Redlands, CA: Environmental Systems Research Institute. 82 p.

Rieman, B.E.; Dunham, J.B. 2000. Metapopulation and salmonids: a synthesis of life history patterns and empirical observations. Ecology of Freshwater Fish. 9: 51–64.

Rieman, B.E.; Hessburg, P.F.; Lee, D.C.; Thurow, R.F.; Sedell, J.R. 2000. Toward an integrated classification of ecosystems: defining opportunities for managing fish and forest health. Environmental Management. 25: 425–444.

Rieman B.E.; Hessburg, P.F.; Luce, C.; Dare, M.R. 2010. Wildfire and management of forests and native fishes: Conflict or opportunity for convergent solutions? BioScience. 60: 460–468.

Rieman, B.E.; Isaak, D.J.; Adams, S.; Horan, D.; Nagel, D.; Luce, C.; Myers, D. 2007. Anticipated climate warming effects on bull trout habitats and populations across the Interior Columbia River Basin. Transactions of the American Fisheries Society. 136: 1552–1565.

Rieman, B.E.; Lee, D.C.; Burns, D.; Gresswell, R.; Young, M.; Stowell, R.; Rinne, J.; Howell, P. 2003. Status of native fishes in the western United States and issues for fire and fuels management. Forest Ecology and Management. 178: 197–212.

Riggs, R.A.; Tiedemann, A.R.; Cook, J.G.; Ballard, T.M.; Edgerton, P.J.; Vavra, M.; Krueger, W.C.; Hall, F.C.; Bryant, L.D.; Irwin, L.L.; Delcurto, T. 2000. Modification of mixed-conifer forests by ruminant herbivores in the Blue Mountains ecological province. Res. Pap. PNW-RP-527. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 77 p.

Roath, L.R.; Krueger, W.C. 1982. Cattle grazing influences on a mountain riparian zone. Journal of Range Management. 35: 100–104.

Robbins, W.G. 1997. Landscapes of promise: the Oregon story, 1800–1940. Seattle, WA: University of Washington Press. 392 p.

Robbins, W.G. 1999. Landscape and environment: ecological change in the Intermontane Northwest. In: Boyd, R., ed. Indians, fire, and the land in the Pacific Northwest. Corvallis, OR: Oregon State University Press: 219–237.

Robbins, W.G.; Wolf, D.W. 1994. Landscape and the Intermontane Northwest: an environmental history. In: Hessburg, P.F., ed. Eastside Forest Ecosystem Health Assessment. Gen. Tech. Rep. PNW-GTR-319. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 32 p.

Roghair, C.N.; Dollof, C.A.; Underwood, M.K. 2002. Response of a brook trout population and instream habitat to a catastrophic flood and debris flow. Transactions of the American Fisheries Society. 131: 718–730.

Roni, P.; Beechie, T.J.; Bilby, R.E.; Leonetti, F.E.; Pollock, M.M.; Pess, G.R. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. North American Journal of Fisheries Management. 22: 1–20.

Rosenberger, A.E.; Dunham, J.B.; Neuswanger, J.R.; Railsback, S.F. [N.d.]. Population and individual responses of rainbow trout to wildfire and channel reorganization: an integrated analysis of observation and models. In review. Freshwater Science. On file with: A.E. Rosenberger, U.S. Geological Survey, Missouri Cooperative Fish and Wildlife Research Unit, University of Missouri, Columbia, Missouri 65211.

Rosenzweig, C.; Karoly, D.; Vicarelli, M.; Neofotis, P.; Wu, Q.; Casassa, G.; Menzel, A.; Root, T.L.; Estrella, N.; Seguin, B.; Tryjanowski, P.; Liu, C.; Rawlins, S.; Imeson, A. 2008. Attributing physical and biological impacts to anthropogenic climate change. Nature. 453: 353–358.

Running, S.W.; Nemani, R.R.; Heinsch, F.A.; Zhao, M.; Reeves, M.; Hashimoto, H. 2004. A continuous satellite-derived measure of global terrestrial primary productivity: future science and applications. Bioscience. 56: 547–560.

Russell, R.E; Royle, J.A.; Saab, V.A.; Lehmkuhl, J.F.; Block, W.M.; Sauer, J.R. 2009. Modeling the effects of environmental disturbance on wildlife communities: avian responses to prescribed fire. Ecological Applications. 19: 1253–1263.

Ryan, R.B. 1997. Before and after evaluation of biological control of the larch casebearer (Lepidoptera: Coleophoridae) in the Blue Mountains of Oregon and Washington, 1972-1995. Environmental Entomology. 26: 703–715.

Sabo, K.E.; Sieg, C.H.; Hart, S.C.; Bailey, J.D. 2009. The role of disturbance severity and canopy closure on standing crop of understory plant species in ponderosa pine stands in northern Arizona, USA. Forest Ecology and Management. 257: 1656–1662.

Sallabanks, R.; Heglund, P.; Haufler, J.B.; Gilbert, B.; Wall, W. 1999. Forest fragmentation in the inland West: issues, definitions, and potential study approaches for birds. In: Rochelle, J.; Lehmann, L.; Wisniewski, J., eds. Forest fragmentation: wildlife and management implications. Leiden, The Netherlands: Brill Publishers: 187–199.

Sallabanks, R.; Marcot, B.; Riggs, R.; Mehl, C.; Arnett, E. 2001. Wildlife of eastside (interior) forests and woodlands. In: Johnson, D.H.; O'Neil, T.A., managing directors. Wildlife–habitat relationships in Oregon and Washington. Corvallis, OR: Oregon State University Press: 213–238.

Sallabanks R.; Riggs, R.A.; Cobb, L.E. 2002. Bird use of forest structural classes in grand fir forests of the Blue Mountains, Oregon. Forest Science. 48: 311–321.

Schmitt, C.L.; Goheen, D.G.; Gregg, T.F.; Hessburg, P.F. 1991. Effects of management activities and stand type on pest-caused losses in true fir and associated species on the Wallowa-Whitman National Forest, Oregon. BMPMZ-01-91. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Wallowa-Whitman National Forest. 78 p.

Schmitt, C.L.; Goheen, E.M.; Frankel, S.J. 1984. Effects of management activities and dominant species type on pest-caused mortality losses in true fir on the Fremont and Ochoco National Forests. Portland, OR: U.S. Department of Agriculture, Forest Service, Forest Pest Management. 34 p.

Schommer, T.J.; Woolver, M.M. 2008. A review of disease related conflicts between domestic sheep and goats and bighorn sheep. Gen. Tech. Rep. RMRS-GTR-209. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 16 p.

Schultz, C.A.; Jedd, T.; Beam, R.D. 2012. The Collaborative Forest Landscape Restoration Program: a history and overview of the first projects. Journal of Forestry. 110: 381–391.

Schumm, S.A.; Hadley, R.J. 1957. Arroyos and the semi-arid cycle of erosion. American Journal of Science. 225: 161–174.

Seibert, J.; McDonnell, J.J.; Woodsmith, R.D. 2010. Effects of wildfire on catchment runoff response: a modelling approach to detect changes in snow-dominated forested catchments. Hydrology Research. 41: 378–390.

Shindler, B. 2007. Public acceptance of wildland fire conditions and fuel reduction practices: challenges for federal forest managers. In: Daniel, T.C.; Carroll, M.S.; Moseley, C.; Raish, C. eds. People, fire, and forests. Corvallis, OR: University of Oregon Press: 37–54.

Shindler, B.; Toman, E. 2003. Fuel reduction strategies in forest communities. Journal of Forestry. 101: 8–15.

Shirk, A.J.; Wallin, D.O.; Cushman, S.A.; Rice, C.G.; Warheit, K.I. 2010. Inferring landscape effects on gene flow: a new model selection framework. Molecular Ecology. 19: 3603–3619.

Shirley, D.M.; Erickson, V. 2001. Aspen restoration in the Blue Mountains of northeast Oregon. In: Shepperd, W.D.; Binkley, D.; Bartos, D.L.; Stohlgren, T.J.; Eskew, L.G., comps. Sustaining aspen in western landscapes: symposium proceedings. Proceedings RMRS-P-18. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station: 13–15.

Shmida, A.; Wilson, M.V. 1985. Biological determinants of species diversity. Journal of Biogeography. 12: 1–20.

Simpson, W.G. 2009. American pikas inhabit low-elevation sites outside the species' previously described bioclimatic envelope. Western North American Naturalist. 69: 243–250.

Singleton, P.H. 2013. Barred owls and northern spotted owls in the eastern Cascade Range, Washington. Seattle, WA: University of Washington. 198 p. Ph.D. dissertation.

Singleton, P.H.; Gaines, W.L.; Lehmkuhl, J.F. 2002. Landscape permeability for large carnivores in Washington: a geographic information system weighted-distance and least-cost corridor assessment. Res. Pap. PNW-RP-549. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 89 p.

Singleton, P.H.; Lehmkuhl, J.F.; Gaines, W.L.; Graham, S.A. 2010. Barred owl space use and habitat selection in the eastern Cascades, Washington. Journal of Wildlife Management. 74: 285–294.

Simpson, M. 2007. Forested plant associations of the Oregon East Cascades. Tech. Pap. R6-NR-ECOL-TP-03-2007. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 590 p.

Skinner, C.N. 1997. Fire history in riparian reserves of the Klamath Mountains. In: Cooper, S.; Sugihara, N., eds. Proceedings—fire in California ecosystems: integrating ecology, prevention, and management. Davis, CA: California Association for Fire Ecology.

Skovlin, J.M.; Bryant, L.D.; Edgerton, P.J. 1989. Timber harvest affects elk distribution in the Blue Mountains of Oregon. Res. Pap. PNW-RP-415. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 10 p.

Skovlin, J.M.; Thomas, J.W. 1995. Interpreting long-term trends in Blue Mountains ecosystems from repeat photography. Gen. Tech. Rep. PNW-GTR-315. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 102 p.

Smith, D. 2012. Washington mill survey, 2010. Olympia, WA: Washington State Department of Natural Resources. 100 p.

Smith, J.K. ed. 2000. Wildland fire in ecosystems: effects of fire on fauna. Gen. Tech. Rep. RMRS-GTR-42-vol. 1. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 83 p.

Smith, N.; Deal, R.; Kline, J.; Blahna, D.; Patterson, T.; Spies, T.A.; Bennett, K. 2011. Ecosystem services as a framework for forest stewardship: Deschutes National Forest overview. Gen. Tech Rep. PNW-GTR-852. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 46 p.

Southwood, T.R.E. 1977. Habitat, the template for ecological strategies? Journal of Animal Ecology. 46: 337–365.

Sovern, S.G.; Taylor, M.; Forsman, E.D. 2011. Nest reuse by northern spotted owls on the east slope of the Cascade Range, Washington. Northwestern Naturalist. 92: 101–106.

Spies, T.A. 1998. Forest structure: a key to the ecosystem. Northwest Science. 72: 34–39.

Spies, T.A.; Duncan, S.L. 2009. Searching for old growth. In: Spies, T.A.; Duncan, S.L., eds. Old growth in a new world: a Pacific Northwest icon reexamined. Washington, DC: Island Press. 344 p.

Spies, T.A.; Hemstrom, M.A.; Youngblood, A.; Hummel, S. 2006. Conserving old-growth forest diversity in disturbance-prone landscapes. Conservation Biology. 20: 351–362.

Spies, T.A.; Lindenmayer, D.B.; Gill, A.M.; Stephens, S.L.; Agee, J.K. 2012. Challenges and a checklist for biodiversity conservation in fire-prone forests: perspectives from the Pacific Northwest of USA and southeastern Australia. Biological Conservation. 145: 5–14.

Spies, T.A.; Turner, M.G. 1999. Dynamic forest mosaics. In: Hunter, M.L., ed. Maintaining biodiversity in forest ecosystems. Cambridge, United Kingdom: Cambridge University Press: 95–160.

Springsteen, B.; Christofk, T.; Eubanks, S.; Mason, T.; Clavin, C.; Storey, B. 2011. Emission reductions from woody biomass waste for energy as an alternative to open burning. Journal of the Air and Waste Management Association. 61: 63–68.

Squires, J.R.; Kennedy, P.L. 2006. Northern goshawk ecology: an assessment of current knowledge and information needs for conservation and management. Studies in Avian Biology. 31: 8–62.

State of Oregon. 2012. National forest health restoration: an economic assessment of forest restoration on Oregon's eastside national forests. Prepared for Governor John Kitzhaber and Oregon's legislative leaders. 84 p. http://www.oregon.gov/gov/docs/OR_Forest_Restoration_Econ_Assessment_Nov_2012.pdf. (17 January 2014).

Staudinger, M.C.; Carter, S.L.; Cross, M.S.; Dubois, N.S.; Duffy, J.E.; Endquist, C.; Griffis, R.; Hellmann, J.J.; Lawler, J.L.; O'Leary, J.; Morrison, S.A.; Sneddon, L.; Stein, B.; Thompson, L.M.; Turner, W. 2013. Biodiversity in a changing climate: a synthesis of current and projected trends in the U.S. Frontiers in Ecology and the Environment. 11: 465–473.

Stephens, G.L.; L'Ecuyer, T.; Forbes, R.; Gettlemen, A.; Golaz, J.-C.; Bodas-Salcedo, A.; Suzuki, K.; Gabriel, P.; Haynes, J. 2010. Dreary state of precipitation in global models. Journal of Geophysical Research. 115: D24211.

Stephens, S.L.; McIver, J.D.; Boerner, R.E.J.; Fettig, C.J.; Fontaine, J.B.; Hartsough, B.R.; Kennedy, P.L.; Schwilk, D.W. 2012. Effects of forest fuel-reduction treatments in the United States. Bioscience. 62: 549–560.

Stewart, I.; Cayan, D.; Dettinger, M. 2005, Changes towards earlier streamflow timing across western North America. Journal of Climate. 18: 1136–1155.

Stidham, M.; Simon-Brown, V. 2011. Stakeholder perspectives on converting forest biomass to energy in Oregon, USA. Biomass and Bioenergy. 35: 203–213.

Stine, P.A.; Conway, S. 2012. Applying GTR-220 Concepts on the Sagehen Experimental Forest. In: North, M. ed. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 184 p.

Strong, N.; Stringer, D.; Welch, T.; Littlefield, B. 2010. Land manager's guide to aspen management in Oregon. Corvallis, OR: Oregon State University, Agricultural Extension Service. http://ir.library.oregonstate.edu/xmlui/handle/1957/18399. (14 January 2014).

Sturrock, R.N.; Frankel, S.J.; Brown, A.V.; Hennon, P.E.; Kliejunas, J.T.; Lewis, K.J.; Worrall, J.J.; Woods, A.J. 2011. Climate change and forest diseases. Plant Pathology. 60: 133–149.

Sundstrom, S.; Nielsen-Pincus, M.; Moseley, C.; McCaffrey, S. 2012. Woody biomass use trends, barriers, and strategies: perspectives of US Forest Service managers. Journal of Forestry. 110: 16–24.

Suring, L.H.; Gaines, W.L.; Wales, B.C.; Mellen-McLean, K.; Begley, J.S.; Mohoric, S. 2011. Maintaining populations of terrestrial wildlife through land management planning: a case study. Journal of Wildlife Management. 75: 945–958.

Sutherland, S.; Nelson, C.R. 2010. Nonnative plant response to silvicultural treatments: a model based on disturbance, propagule pressure, and competitive abilities. Western Journal of Applied Forestry. 25: 27–33.

Swan, L.; Tokarczyk, J.; Kaetzel, B. 2012. Eastern Oregon primary wood products processing facilities and operations, Final Draft. Vallejo, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, State and Private Forestry. 21 p.

Swanson, M.E.; Franklin, J.F.; Beschta, R.L.; Crisafulli, C.M.; DellaSala, D.A.; Hutto, R.L.; Lindenmayer, D.B.; Swanson, F.J. 2011. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment. 9: 117–125.

Swenson, J.J.; Waring, R.H. 2006. Modeled photosynthesis predicts woody plant richness at three geographic scales across northwestern USA. Global Ecology and Biogeography. 15: 470–485.

Tague, C.; Grant, G. 2009. Groundwater dynamics mediate low-flow response to global warming in snow-dominated alpine regions. Water Resources Research. 45: W07421. 12 p.

Taylor, A.H.; Skinner, C.N. 1998. Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management. 111: 285–301.

Taylor, A.H.; Skinner, C.N. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications. 13: 704–719.

Theobald, D.M.; Crooks, K.R.; Norman, J.B. 2011. Assessing effects of land use on landscape connectivity: loss and fragmentation of western U.S. forests. Ecological Applications. 21: 2445–2458.

Theobald, D.M.; Reed, S.E.; Fields, K.; Soule, M. 2012. Connecting natural landscapes using a landscape permeability model to prioritize conservation activities in the United States. Conservation Letters. 5: 123–133.

Thomas, J.W. ed. 1979. Wildlife habitat in managed forests: the Blue Mountains of Oregon and Washington. Agric. Handb. 553. Washington, DC: U.S. Department of Agriculture, Forest Service. 512 p.

Thomas, J.W.; Anderson, R.G.; Maser, C.; Bull, E.L. 1979a. Snags. In: Thomas, J.W., ed. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agric. Handb. 553. Washington, DC: U.S. Department of Agriculture, Forest Service: 60–77.

Thomas, J.W.; Forsman, E.D.; Lint, J.B.; Meslow, E.C.; Noon, B.B.; Verner, J. 1990. A conservation strategy for the northern spotted owl. Interagency scientific committee to address the conservation of the northern spotted owl. Portland, OR: U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior Bureau of Land Management, Fish and Wildlife Service, and National Park Service. 427 p.

Thomas, J.W.; Maser, C.; Rodiek, J.E. 1979b. Edges. In: Thomas, J.W., ed. Wildlife habitats in managed forests: the Blue Mountains of Oregon and Washington. Agric. Handb. 553. Washington, DC: U.S. Department of Agriculture, Forest Service: 48–59.

Tiedemann, A.R.; Berndt, H.W. 1972. Vegetation and soils of a 30-year deer and elk exclosure in central Washington. Northwest Science. 46: 59–66.

Tiedemann, A.R.; Woodard, P.M. 2002. Multiresource effects of a stand-replacement prescribed fire in the *Pinus contorta-Abies lasiocarpa* vegetation zone of central Washington. Gen. Tech. Rep. PNW-GTR-535. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 26 p.

Tinnin, R.O. 1981. Interrelationships between *Arceuthobium* and its hosts. American Midland Naturalist. 106: 126–132.

Tinnin, R.O. 1984. The effect of dwarf mistletoe on community ecology. In: Hawksworth, F.G.; Scharpf, R.F., eds. Biology of dwarf mistletoes. Gen. Tech. Rep. RM-GTR-111. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 117–122.

Tinnin, R.O.; Hawksworth, F.G.; Knutson, D.M. 1982. Witches' broom formation in conifers infected by *Arceuthobium douglasii*: an example of parasitic impact on community dynamics. American Midland Naturalist. 107: 351–359.

Tinnin, R.O.; Knutson, D.M. 1980. Growth characteristics of the brooms on Douglas-fir caused by *Arceuthobium douglasii*. Forest Science. 26: 149–158.

Toman, E.; Stidham, M.; Schindler, B.; McCaffrey, S. 2011. Reducing fuels in the wildland-urban interface: community perceptions of agency fuels treatments. International Journal of Wildland Fire. 20: 340–349.

Topik, C.; Halverson, N.M.; T. High, T. 1988. Plant association and management guide for the ponderosa pine, Douglas-fir, and grand fir zones, Mount Hood National Forest. R6 Ecol. Tech. Pap. 004-88. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 132 p.

Torgersen, T.R. 2001. Defoliators in eastern Oregon and Washington. Northwest Science. 75: 11–20.

Torgersen, T.R.; Powell, D.C.; Hosnan, K.P.; Schmidt, F.H. 1995. No long-term impact of carbaryl treatment on western spruce budworm and host trees in the Malheur National Forest, Oregon. Forest Science. 41: 851−863.

Underwood, E.C.; Viers, J.H.; Quinn, J.F.; North, M.P. 2010. Using topography to meet wildlife and fuels treatment objectives in fire-suppressed landscapes. Journal of Environmental Management. 46: 809–819.

U.S. Department of Agriculture, Forest Service [USDA FS]. 1996. Status of the Interior Columbia River Basin: summary of scientific findings. Gen. Tech. Rep. PNW-GTR-385. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 151 p.

U.S. Department of Agriculture, Forest Service [USDA FS]. 2013. Ecological restoration policy. Federal Register Vol. 78, No. 177. RIN 0596-AC82. P. 56202.

U.S. Department of Agriculture [USDA]. 2010. Strategic plan: FY 2010–2015. Washington, DC: U.S. Department of Agriculture. 58 p.

U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 1997. Grizzly bear recovery plan supplement: North Cascades Ecosystem Recovery Plan. Missoula: MT: U.S. Fish and Wildlife Service. 64 p.

U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2011. Revised recovery plan for the northern spotted owl (*Strix occidentalis caurina*). Portland, OR: U.S. Fish and Wildlife Service. 277 p.

Van Pelt, R. 2008. Identifying old trees and forests in eastern Washington. Washington State Dept. of Natural Resources, Olympia WA. 166 p. http://www.dnr.wa.gov/ResearchScience/Topics/ForestResearch/Pages/lm_oldgrowth_guides.aspx. (14 January 2014).

Vavra, M.; Walburger, K.; Delcurto, T. 2005. Ungulate ecology of ponderosa pine ecosystems in the Northwest. In: Ritchie, M.; Maguire, D.; Youngblood, A., eds. Proceedings of the symposium on ponderosa pine: issues, trends, and management. Gen. Tech.Rep. PSW-GTR-198. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 145–158.

Verschuyl, J.P.; Hansen, A.J.; McWethy, D.B.; Sallabanks, R.; Hutto, R.L. 2008. Is the effect of forest structure on bird diversity modified by forest productivity? Ecological Applications. 18: 1155–1170

Vinyeta, K.; Lynn, K. 2013. Exploring the role of traditional ecological knowledge in climate change initiatives. Gen. Tech. Rep. PNW-GTR-879. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 37 p.

Voggesser, G.; Lynn, K.; Daigle, J.; Lake, F.K.; Ranco, D. 2013. Cultural impacts to tribes from climate change influences on forests. Climatic Change. [online]. doi: 10.1007/s10584-013-0733-4.

Volney, W.J.A.; Fleming, R.A. 2007. Spruce budworm (*Choristoneaura* spp.) biotype reactions to forest and climate characteristics. Global Change Biology. 13: 1630−1643.

Wagener, W.W. 1965. Dwarf mistletoe removal and reinvasion in Jeffrey and ponderosa pine, northeastern California. Res. Note PSW-RN-73. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 8 p.

Wales, B.C.; Mellen-McLean, K.; Gaines, W.L.; Suring, L. 2011. Focal species assessment of current condition and the proposed action (alternative B) for the Blue Mountains forest plan revisions (draft Oct. 2011). Unpublished paper. On file with: U.S. Department of Agriculture, Forest Service, Wallowa-Whitman National Forest, Blue Mountain Forest Plan Revision Team, Baker City, OR 97814.

Walker, B.; Holling, C.S.; Carpenter, S.R.; Kinzig, A. 2004. Resilience, adaptability and transformability in social–ecological systems. Ecology and Society. 9: 5. http://www.ecologyandsociety.org/vol9/iss2/art5/. (17 January 2014).

Walker, B.; Salt, D. 2006. Resilience thinking: Sustaining ecosystems and people in a changing world. Washington, DC: Island Press. 174 p.

Wallin, D.O.; Swanson, F.J.; Marks, B. 1994. Landscape pattern response to changes in pattern generation rules: land-use legacies in forestry. Ecological Applications. 4: 569–580.

Walther G.; Post, E.; Convey, P.; Menzel, A.; Parmesan, C.; Beebee, T.J.C.; Fromentin, J.; Hoegh-Guldberg, O.; Bairlein, F. 2002. Ecological responses to recent climate change. Nature. 416: 389−395.

Waring, R.H.; Coops, N.C.; Running, S.W. 2011. Predicting satellite-derived patterns of large-scale disturbances in forests of the Pacific Northwest Region in response to recent climatic variation. Remote Sensing of Environment. 115: 3554−3566.

Washington State Department of Natural Resources [WA DNR]. 2007. The future of Washington forests. Olympia, WA: DNR Communications Group. 76 p.

Washington Wildlife Habitat Connectivity Working Group [WHCWG]. 2010. Washington connected landscapes project: statewide analysis. Olympia, WA: Washington Departments of Fish and Wildlife, and Transportation. 223 p.

Wasserman, T.N.; Cushman, S.A.; Shirk, A.S.; Landguth, E.L.; Littell, J.S. 2012. Simulating the effects of climate change on population connectivity of American marten (*Martes americana*) in the northern Rocky Mountains, USA. Landscape Ecology. 27: 211–225.

Watson, D.M. 2001. Mistletoe—a keystone resource in forests and woodlands worldwide. Annual Review of Ecology and Systematics. 32: 219–249.

Weaver, H. 1974. Effects of fire on temperate forests: western United States. In: Kozlowski, T.T.; Ahlgren, C.E., eds. Fire and ecosystems. New York: Academic Press: 279–320.

Weins, J.A.; Haywood, G.D.; Safford, H.D.; Catherine, C.M. 2012. Historical environmental variation in conservation and natural resource management. Hoboken, NJ: John Wiley & Sons. 337 p.

Wenger, S.J.; Isaak, D.J.; Luce, C.H.; Neville, H.M.; Fausch, K.D.; Dunham, J.B.; Dauwalter, D.C.; Young, M.K.; Elsner, M.M.; Rieman, B.E.; Hamlet, A.F.; Williams, J.E. 2011. Flow regime, temperature, and biotic interactions drive differential declines of trout species under climate change. Proceedings of the National Academy of Sciences of the United States of America. 108: 14175–14180.

Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; Swetnam, T.W. 2006. Warming and earlier spring increase western US forest wildfire activity. Science. 313: 940–943.

Westman, W.E. 1978. Measuring the inertia and resilience of ecosystems. Bioscience 28: 705–710.

White, R. 1983. The altered landscape: social change and land in the Pacific Northwest. In: Robbins, W.G., Frank, R.J., Ross, R.E., eds. Regionalism and the Pacific Northwest. Corvallis, OR: Oregon State University Press. 109–128.

White, R. 1991. It's your misfortune and none of my own: a new history of the American West. Norman, OK: University of Oklahoma Press. 644 p.

White, R. 1992. Discovering nature in North America. Journal of American History. 79: 874–891.

White, R. 1999. Indian land use and environmental change. In: Boyd, R., ed. Indians, fire, and the land in the Pacific Northwest. Corvallis, OR: Oregon State University Press: 36–49.

Whitlock, C.; Knox, M.A. 2002. Prehistoric burning in the Pacific Northwest: Humans versus climate influences. In: Vale, T., ed. Fire, native peoples, and the natural landscape. Washington, DC: Island Press: 195–232.

Whitlock, C.; Shafer, S.; Marlon, J. 2003. The role of climate and vegetation change in shaping past and future fire regimes in the northwestern USA and implications for ecosystem management. Forest Ecology and Management. 178: 5–21.

Whitlock, M.C. 1996. The red queen beats the jack-of-all-trades: the limitations on the evolution of phenotypic plasticity and niche breadth. The American Naturalist. 148: S65–S77.

Whyte, K.P. 2013. Justice forward: tribes, climate adaptation and responsibility. Climatic Change. 120: 517–530.

Wickman, B.E. 1992. Forest health in the Blue Mountains: the influence of insects and diseases. Gen. Tech. Rep. PNW-GTR-295. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 15 p.

Wienk, C.L.; Sieg, C.H.; McPherson, G.R. 2004. Evaluating the role of cutting treatments, fire and soil seed banks in an experimental framework in ponderosa pine forests of the Black Hills, South Dakota. Forest Ecology and Management. 192: 375–393.

Wiens, J.A.; Hayward, G.D; Holthausen, R.S.; Wisdom, M.J. 2008. Using surrogate species and groups for conservation planning and management. BioScience. 58: 241–252.

Williams, A.P.; Allen, C.D.; Macalady, A.K.; Griffin, D.; Woodhouse, C.A.; Meko, D.M.; Swetnam, T.W.; Rauscher, S.A.; Seager, R.; Grissino-Mayer, H.D.; Dean, J.S.; Cook, E.R.; Gangodagamage, C.; Cai, M.; McDowell, N.G. 2012. Temperature as a potent driver of regional forest drought stress and tree mortality. Nature Climate Change. doi:10.1038/nclimate1693.

Williams, C.K.; Kelley, B.F.; Smith, B.G.; Lillybridge, T.R. 1995. Forested plant associations of the Colville National Forest. Gen.Tech. Rep. PNW-GTR-360. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 370 p.

Williams, C.K.; Lillybridge, T.R. 1983. Forested plant associations of the Okanogan NF. R6 Ecol 132-1983. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 116 p.

Williams, D.W.; Liebhold, A.M. 1995. Herbivorous insects and global change: potential changes in the spatial distribution of forest defoliator outbreaks. Journal of Biogeography. 22: 665–671.

Williams, M.A.; Baker, W.L. 2012. Comparison of the higher-severity fire regime in historical (A.D. 1800) and modern (A.D. 1984–2009) montane forests across 624,156 ha of the Colorado Front Range. Ecosystems. 15: 832–847.

Wimberly, M.C.; Spies, T.A. 2001. Modeling landscape patterns of understory tree regeneration in the Pacific Northwest, USA. Applied Vegetation Science. 4: 277–286.

Winter, G.; Vogt, C.; Fried, J.S. 2002. Fuel treatments at the wildland-urban interface- common concerns in diverse regions. Journal of Forestry. 100: 15–21.

Winthers, E.; Fallon, D.; Haglund, J.; DeMeo, T.; Nowacki, G.; Tart, D.; Ferwerda, M.; Robertson, G.; Gallegos, A.; Rorick, A.; Cleland, D.T.; Robbie, W. 2005. Terrestrial Ecological Unit Inventory technical guide. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office, Ecosystem Management Coordination Staff. 245 p.

Wipfli, M.S.; Richardson, J.S.; Naiman, R.J. 2007. Ecological linkages between headwaters and downstream ecosystems: transport of organic matter, invertebrates, and wood down headwater channels. Journal of the American Water Resources Association. 43: 72–85.

Wisdom M.J.; Holthausen, R.S.; Wales, B.C. 2000. Source habitats for terrestrial vertebrates of focus in the Interior Columbia River Basin: broad-scale trends and management implications. Vol. 1—Overview. Gen. Tech. Rep. PNW-GTR-485. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 156 p.

Wissmar, R.C.; Smith, J.E.; McIntosh, B.A.; Li, H.W.; Reeves, G.H.; Sedell, J.R. 1994a. A history of resource use and disturbance in riverine basins of eastern Oregon and Washington (early 1880s–1990s). Northwest Science. 68: 1–35.

Wissmar, R.C.; Smith, J.E.; McIntosh, B.A.; Li, H.W.; Reeves, G.H.; Sedell, J.R. 1994b. Ecological health of river basins in forested regions of eastern Oregon and Washington. In: Hessburg, P.F., ed. Eastside Forest Ecosystem Health Assessment. Gen. Tech. Rep. PNW-GTR-326. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 65 p.

Wondzell, S.M.; King, J.G. 2003. Postfire erosional processes in the Pacific Northwest Rocky Mountain regions. Forest Ecology and Management. 78: 75–87.

Woodard, P.M. 1977. Effects of prescribed burning on two different-aged high elevation plant communities in eastern Washington. Seattle, WA: University of Washington. 228 p. Ph.D. dissertation.

Wright, C.S.; Agee, J.K. 2004. Fire and vegetation history in the eastern Cascade Mountains, Washington. Ecological Applications. 14: 443–459.

Wu, J.; Loucks, O.L. 1995. From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. The Quarterly Review Biology. 70: 439–466.

Wu, J.; David, J.L. 2002. A spatially explicit hierarchical approach to modeling complex ecological systems: theory and applications. Ecological Modelling. 153: 7–26.

Appendix 1—List of Practical Considerations for Landscape Evaluation and Restoration Planning

This list of management considerations is intended to provide an example of the kinds of key steps that could be taken in achieving restoration and resilience goals. The list does not establish a rigid process nor required steps in project planning; rather, these are reminders of the types of planning actions that can enable managers to achieve a broad range of landscape restoration objectives. Their application to a specific project is at the discretion of the responsible official, depending on the purpose of the action. The six major elements of this example process are presented first, followed by a more detailed discussion of the kinds of activities that can contribute knowledge through this process:

- **Local landscapes can be prioritized within ecoregions for restoration.** Local landscapes might be individual or groups of subwatersheds or watersheds. Local landscape domains should represent land units that are useful to terrestrial and aquatic system evaluations. Broad-scale assessment methodology can be used to base prioritization on the degree of vegetation departure from historical and future climate change reference conditions (including exotic plant invasions); vulnerability to large-scale, high-severity fire, drought, insect, and pathogen-caused tree mortality events; and priorities for terrestrial and aquatic habitat restoration and regional linkages, especially as they relate to listed and focal species. Other criteria may also be important.
- **Consider engaging in stakeholder processes early and as often as is practical.**
- **Characterization of the current vegetation conditions of high-priority local landscapes is recommended.** Using a meso-scale assessment methodology, estimate and spatially allocate the departure of current conditions from historical and climate change reference conditions; landscape vulnerability to large-scale, high-severity fire, drought, insect, and pathogen-caused tree mortality events; noxious and nonnative plant and animal invasions; and priorities for listed and focal species (terrestrial and aquatic) habitat restoration and improved connectivity.
- **For local landscapes, we recommend developing target frequency and patch size histograms (distributions) of desired vegetation conditions,** including physiognomic types, cover types, structural classes, canopy cover classes, density and age classes, and other related conditions, considering the potential vegetation type, the topographic setting, local soils, historical

disturbance regimes and range of variability, expected climate change and future range of variability, special legacy wildlife concerns, and other ecological, operational, or socioeconomic factors as needed.

- **Consider developing a method to spatially allocate and distribute these patch size distributions to local landscapes.** Fine tune target distributions based on landscape context and special legacy considerations.
- **Consider developing Proposed Landscape Treatment Areas (PLTAs) in local landscapes based on ecological and socioeconomic criteria**, including stakeholder input.

The following more detailed steps may be considered for local landscape evaluations where appropriate to the purpose of the project or planning action:

- **Characterizing and mapping the current vegetation conditions of entire local landscapes; include all potential vegetation types (PVTs) is recommended.** Motivating questions are:
  - What are the current patterns of physiognomic types, cover types, structural stages or successional classes, stand density, and basal area classes?
  - What is the density and distribution of large, old, fire-tolerant/intolerant trees, and the density and basal area of shade-tolerant trees?
  - What are current surface and canopy fuel conditions?
  - What is the vulnerability of the current vegetation to major insect and pathogen disturbances?
  - What is the vulnerability of the current vegetation and fuelbeds to wildfires?
- **For each ecoregion, we suggest characterizing historical reference conditions (ca. 1880–1900) for all PVTs.** Motivating questions are:
  - What were the historical patterns of physiognomic types, cover types, structural stages or successional classes, stand density, and basal area classes?
  - What was the historical pattern and distribution of large, old, fire-tolerant trees?
  - What was the historical pattern and distribution of shade-tolerant trees? How did their basal area and density vary? How were these patterns distributed with respect to topography, PVTs, and dominant processes?
  - What is a set of guiding reference conditions (potential restoration and resilience targets) based on historical patterns of structure and composition?

  - What is a desired set of class and landscape pattern metrics to characterize these reference conditions?

  Develop reference conditions that enable a direct comparison of the current conditions with the historical reference conditions.

- **Consider comparing current vegetation with reference conditions** (structure, composition, disturbance regimes). Motivating questions are:
  - Where within the local landscapes have disturbance regimes and vegetation structure and composition have been significantly altered by Euro-American activities?
  - How does this departure vary by topography, soils, and PVT setting, and by disturbance regime?
- **To strengthen understanding of sustainable landscape-scale ecological condition, future climate change reference conditions (ca. 2010–2050) for all PVTs can be characterized for each ecoregion.** Motivating questions are:
  - What do climate change models predict to be the most likely climate change scenarios for each ecoregion? Identify one or two most likely climate change scenarios for each ecoregion; use existing information from models or expert opinion.
  - What are predicted future patterns of physiognomic types, cover types, structural stages or successional classes, stand density, and basal area classes?
  - What is a set of guiding reference conditions (potential restoration and resilience targets) based on future patterns of structure and composition?
  - What is a desired set of class and landscape pattern metrics to characterize these reference conditions?

  Develop reference conditions that enable a direct comparison of the current conditions with the future climate change reference conditions.

- **Assessment of the risk of high-severity fire under current fuel conditions is suggested.**
  - Tools such as FlamMap, a fire behavior mapping and analysis program, are currently being used for this.
  - Terrain-routed wind flow projections can be obtained from Wind Ninja, Wind Wizard, or similar simulation tools.
  - To initialize wind speeds and directions, use meteorological data from nearby weather stations where available, or expert advice as needed.

    - Determine the mapped distribution of burn probability, probable flame length, fireline intensity, rate of spread, crown fire ignition, and spread potential.
    - Note the hotspots on the landscape for high-probability flame length and fireline intensity.
    - Note the corridors on the landscape that most strongly tend to spread fire.
- **Assessment of the departures from historical and climate change reference conditions in patch-scale fire behavior attributes.** Motivating questions are:
    - How have spatial patterns of surface and canopy fuels changed with respect to these references?
    - How will changes translate to changes in expected fire behavior?
    - What are the key changes by PVT setting?
    - How are changes in expected fire behavior spatially distributed?
- **Evaluating the wildlife habitat conditions (distribution, abundance, and connectivity of different habitat types) and assessing current conditions. The following steps have been used for regional-scale wildlife habitat assessment (Suring et al. 2011), but can be adjusted to address planning at other scales:**
    - Identify species of conservation concern.
    - Describe habitats and other important factors (e.g., sensitivity to activities associated with roads or residential developments).
    - Group species according to habitat associations and other threats.
        - Select species for assessment. Appropriate assessment target species could include threatened, endangered, or sensitive species, or focal species selected based on their association with specific habitat conditions and particular threats associated with human activities.
        - Develop assessment models for each focal species.
        - Develop conservation strategies that address management of habitat and other risks.
        - Design monitoring and adaptive management plans.
- **A terrestrial landscape diagnosis of habitat retention and vegetation restoration needs can be developed by considering items above.**
    - Evaluation of the road network associated with the current landscape is recommended.
    - Identify all roads crossing streams, culverts, and other fish passage barriers.
    - Identify road segments that confine stream channels, reduce access to off-channel features, and contribute the most sediment.

  - Identify portions of the road network that most reduce subsurface water flow and accelerate runoff.
  - Identify the portions of the road network that are most essential for access.
- **Evaluation of the stream network of the current landscape is recommended.**
  - Identify and map existing anadromous and cold water fish habitats, especially for listed or focal species.
  - Identify and map habitats with inherent potential for these same species.
  - Identify and map the cold water upwelling areas within the stream network and the extent of their influence on water temperature.
- **Evaluation of the surface erosion and mass failure potential within the landscape is recommended.**
  - Identify anadromous and cold water fish habitats that are prone to erosion, especially those of regionally listed or focal species.
  - Identify habitats with inherent potential for these same species that are especially prone to erosion effects.
- **An aquatic landscape diagnosis of habitat restoration needs can be derived by considering items above.**
- **Terrestrial and aquatic landscape and road restoration needs can be diagnosed using considerations items above.**
- **Using integrated risk and vulnerability assessments and socioeconomic opportunity factors can be effective in spatially allocating treatment priorities and identifying proposed landscape treatment areas (PLTAs)** (fig. 47). Spatial decision support tools can help users jointly consider the spatial allocation of treatment priorities across a multitude of factors.
  - Assess and map current land use and land designations and whatever constraints, challenges, or opportunities this suggests.
  - Develop and identify PLTAs that restore key concerns identified in analysis.
  - Among the PLTAs, vary the emphasis among the concerns to provide a wide variety of management alternatives to consider.
  - Array and compare the social and economic benefits and costs of each alternative.
  - Identify transitioning issues: How do we transition from a tenuous current condition to a more desirable and resilient future? How do we provide specific attention to species at risk and the elements of the landscape that need conservation in the near term while transitioning to the future?

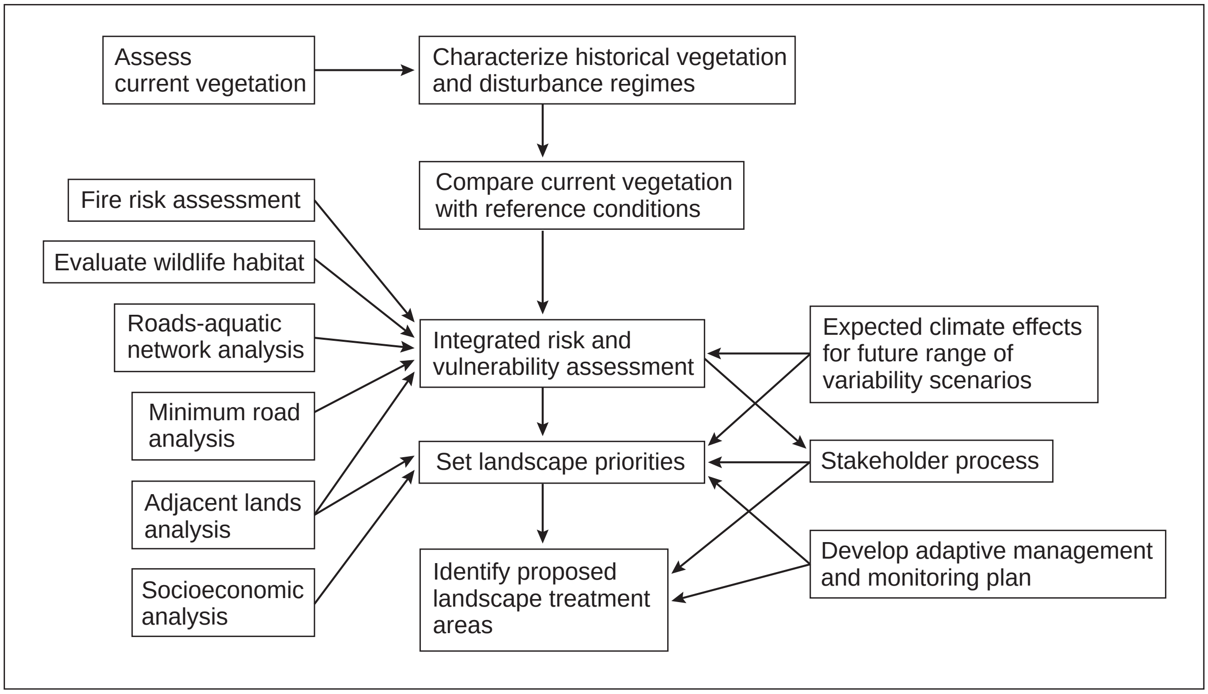

Figure 47—Schematic diagram of a suggested landscape evaluation workflow.

- Develop a landscape diagnosis and prescription for the best subset of alternatives based on stakeholder inputs and the condition of the resources.
- Assess potential effects of proposed work on adjoining lands and possible effects of ongoing activities on adjacent lands.

- **Develop a plan for monitoring implementation and effectiveness of the proposed treatments.** Monitoring programs take many different forms depending on specific objectives and available resources. Careful planning of a monitoring strategy is important to enable scientifically effective and reliable data collection and analysis for the intended purposes. The Forest Service planning rule provides detailed guidance for Forest Plan monitoring (36 CFR 219.12) including both a broad and local-scale approach. This plan also includes the option for jointly developed (i.e., more than one forest) plans that will effectively address broader-scale needs.

- **Regularly assess new scientific findings and adjust plans accordingly.**

Disclaimer: As stated above, the preceding elements are provided for the consideration of managers in conducting landscape evaluations to achieve sustainable land management objectives. These specific elements are not meant to suggest strict requirements; rather they are examples of the kinds of considerations that enable managers to obtain the best available scientific information for project planning.

Appendix 2—Regional-Scale Fire Regimes Group Classification for Oregon and Washington[1]

Most Moist Mixed-Conifer (MMC) Fire Regimes Fall Within Types I and III a and b.

0 to 35 years, low severity—

Typical climax plant communities include ponderosa pine, eastside/dry Douglas-fir, pine-oak woodlands, Jeffrey pine on serpentine soils, oak woodlands, and very dry white fir. Large stand-replacing fire can occur under certain weather conditions, but are rare events (i.e., every 200+ years).

0 to 35 years, mixed and high severity—

Includes true grasslands (Columbia basin, Palouse, etc.) and savannahs with typical return intervals of less than 10 years; mesic sagebrush communities with typical return intervals of 25 to 35 years and occasionally up to 50 years, and mountain shrub communities (bitterbrush, snowberry, ninebark, ceanothus, Oregon chaparral, etc.) with typical return intervals of 10 to 25 years. Certain specific communities include mountain big sagebrush and low sagebrush-fescue communities. Grasslands and mountain shrub communities are not completely killed, but usually only top-killed and resprout.

35 to 100+ years, mixed severity—

This regime usually results in heterogeneous landscapes. Large, high-severity fires may occur but are usually rare events. Such high-severity fires may "reset" large areas (10,000 to 100,000 ac [4050 to 40 500 ha]) but subsequent mixed-severity fires are important for creating landscape heterogeneity. Within these landscapes a mix of stand ages and size classes are important characteristics; generally, the landscape is not dominated by one or two age classes. In southeastern Oregon, this regime also includes aspen, riparian communities, most meadows, and wetlands.

<50 years, mixed severity—

Typical potential plant communities include mixed conifer, very dry westside Douglas-fir, and dry grand fir. Lower severity fire tends to predominate in many events.

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[1] Evers, L. 2002. Fire regimes of Oregon and Washington. Unpublished report. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 3 p.

50 to 100 years, mixed severity—
Typical climax plant communities include well drained western hemlock; warm, mesic grand fir, particularly east of the Cascade crest; and eastside western redcedar. The relative amounts of lower and higher severity patches within a given event are intermediate between IIIa and IIIc.

100 to 200 years, mixed severity—
Typical potential plant communities include western hemlock, Pacific silver fir, and whitebark pine at or below 45° N latitude and cool, mesic grand fir and Douglas-fir. Higher-severity fire tends to dominate in many events.

35 to 100+ years, high severity—
Seral forest communities that arise from or are maintained by high-severity fires, such as lodgepole pine, aspen, western larch, and western white pine, often are important components in this fire regime. Dry sagebrush and mountain mahogany communities also fall within this fire regime. Natural ignitions within this regime that result in large fires may be relatively rare, particularly in the Cascades north of 45° N latitude.

35 to 100+ years, high severity, juxtaposed—
Typified by what would normally be considered a long-interval regime that lies immediately above a shorter interval or lower severity fire regime. Most often the fire originates lower on the slope and burns uphill into regime IVa. In southeastern Oregon, this subregime includes Wyoming big sagebrush communities on deeper soils below 5,000 ft elevation. Forest examples include lodgepole pine immediately above ponderosa pine in the eastside Washington Cascades and aspen imbedded within dry grand fir in the Blue Mountains. This regime is often found in lower elevations or drier sites than is considered typical for regime IV.

100+ years, high severity, patchy arrangement—
Typical potential forest communities include subalpine fir and mountain hemlock parkland and whitebark pine north of 45° N latitude.

Other community types include mixed Wyoming big sagebrush and low sagebrush on low productivity sites such as scablands, stiff sagebrush, and true old growth juniper savannah (<10 percent canopy closure). Some forbs are present, such as Sandberg's bluegrass, and the availability of many of these areas for burning depends on wet years that result in much greater grass production than is typical. Typical fire return interval in these communities is 100 to 150 years.

100 to 200 years, high severity—

Typical forest plant communities include subalpine mixed conifer (spruce-fir), western larch, and western white pine. Important potential forest plant communities include mountain hemlock in the Cascades and Pacific silver fir north of 45° N latitude.

Other plant communities include the intergrade between Wyoming big sagebrush and greasewood, shadscale on non-alkali soils, spiny hopsage, and alpine grasslands and heath in southeastern Oregon.

> 200 years, high severity—

This fire regime occurs at the environmental extremes where natural ignitions are very rare or virtually nonexistent, or environmental conditions rarely result in large fires. Sites tend to be very cold, very hot, very wet, very dry, or some combination of these conditions.

Typical plant communities include black sagebrush, salt desert scrub, greasewood on dunes, true old-growth juniper with at least 10 percent canopy closure and mountain-mahogany in rocky areas, and alpine communities and subalpine heath in the Blue Mountains and Cascades. Most species tend to be small and low-growing. Bare ground is common.

200 to 400 years, high severity—

Forest plant communities are at least somewhat fire-adapted. Typical plant communities include Douglas-fir, noble fir, and mountain hemlock on drier sites in parts of western Washington.

400+ years, high severity—

Forest plant communities are weakly fire-adapted or not fire-adapted. Typical plant communities include Douglas-fir, Pacific silver fir, western hemlock, western redcedar, and mountain hemlock on moister sites in western Washington.

No fire—

This regime includes forest plant communities with no evidence of fire for 500 years or more. Stands often have extremely deep duff layers on poorly developed soils. Typical plant communities include Sitka spruce and Pacific silver fir along the Oregon and Washington coast and very wet western redcedar sites.

Appendix 3—Reconstructed Historical Vegetation Conditions Within Sampled Areas of the Blue Mountains Province

This appendix is provided to highlight vegetation conditions that existed early in the 20th century in the Blue Mountains. The data are sorted by potential vegetation type (PVT); this sorting gives the reader the capacity to contrast overstory and understory size class, canopy cover, cover type, and structural class conditions for the major potential vegetation types. These insights are a companion to the body of this synthesis and are meant to amplify our observations about disturbance regimes and the sorts of patterns resulting from them. The data are adapted from Hessburg et al. (1999b, 2000a) and are available upon request. We provide these data to help the reader develop a mental picture of how the Blue Mountains landscape looked and likely functioned before forest management was in full swing (fig. 48). The vegetation conditions do not reflect a pristine condition and instead reflect some influence of early livestock grazing, timber harvest, and wildfire exclusion. Nonetheless, they are the best data we have and are based on the earliest available black-and-white, stereo-aerial photography, which was usually taken in anticipation of major future harvest activities. We found the earliest aerial photography to be an extensive reconnaissance of mostly nonharvested forest reserves. From characterizations like these, it is possible to visualize the primary directions and magnitudes of changes that occurred to the present day as a consequence of the first century of forest management.

Key to abbreviations in the tables that follow:

Potential vegetation types (PVT):

PIPO = ponderosa pine
WD PSME/ABGR = warm/dry—Douglas-fir/grand fir
CM PSME/ABGR = cool/moist—Douglas-fir/grand fir
WD ABLA2/PIEN = warm/dry—subalpine fir/Engelmann spruce
CM ALBA2/PIEN = cool/moist—subalpine fir/Engelmann spruce

Cover types:

PIPO = ponderosa pine
PSME = interior Douglas-fir
LAOC = western larch
ABGR = grand fir or white fir
Other = forest cover types not in PIPO, PSME, LAOC, or ABGR
NF = herbland, shrubland, and nonforest/nonrange types

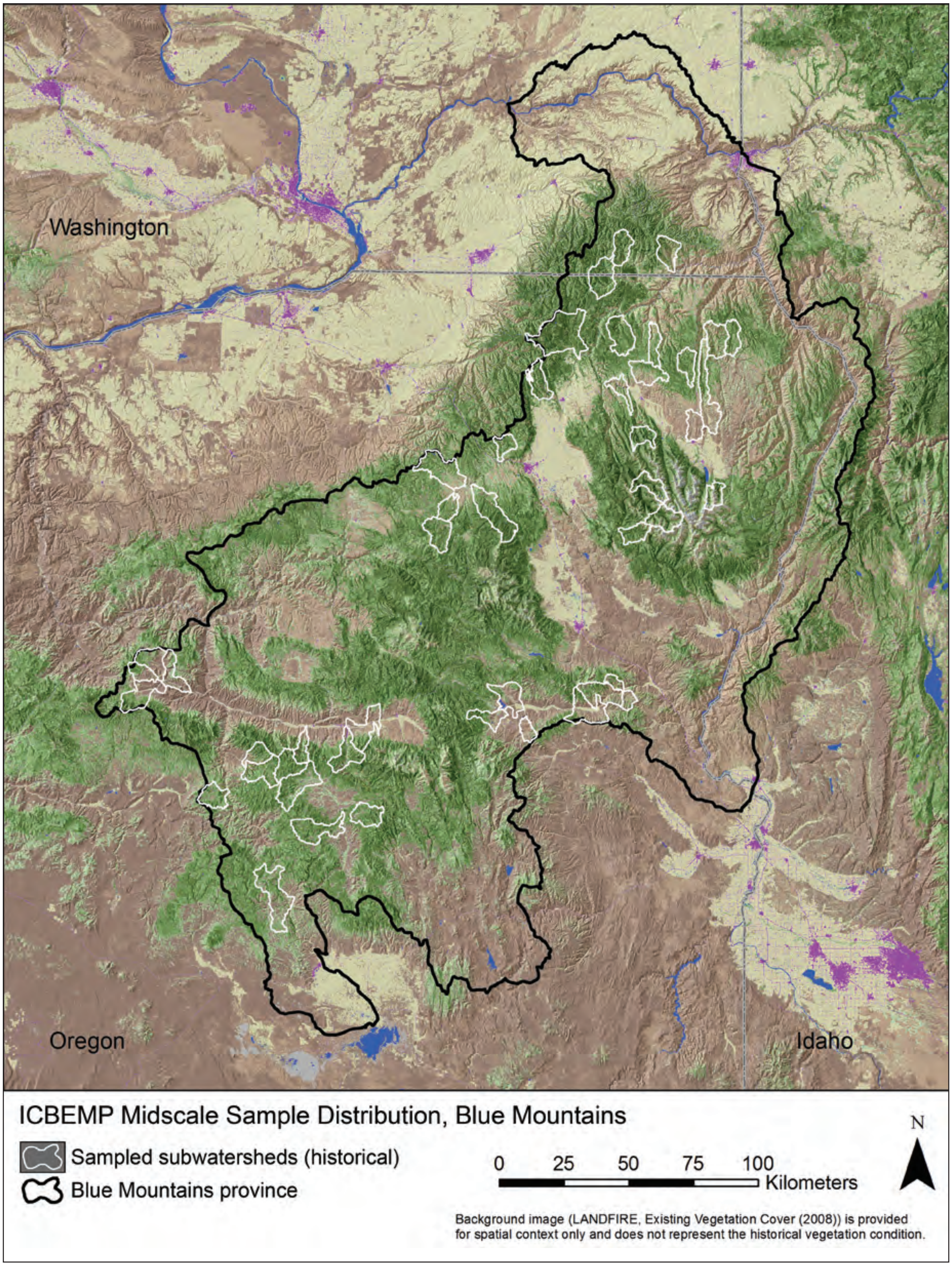

Figure 48—Subwatersheds sampled in the Blue Mountains ecological reporting unit during Interior Columbia Basin Ecosystem Management Project (ICBEMP) (Hann et al. 1997; Hessburg et al. 1999a, 2000c). The black outline delimits the sampled Blue Mountains area; the white lines delineate the reconstructed subwatersheds.

Structure class:

si = stand initiation
seoc = stem exclusion, open canopy
secc = stem exclusion, closed canopy
ur = understory reinitiation
yfms = young forest, multistory
ofms = old forest, multistory
ofss = old forest, single story
wood = woodland structures (all woodland structural classes combined)
herb = herbland (all herbland structural classes combined)
shrub = shrubland (all shrubland structural classes combined)
other = nonforest/nonrange and other anthropomorphic types

Tree size (overstory "OS," understory "US," and "Size"):

SS = seedlings and saplings (< 5.0 in diameter at breast height [d.b.h.])
PT = pole-size trees (5.1 to 8.9 in d.b.h.)
SmT = small trees (9.0 to 15.9 in d.b.h.)
MedT = medium trees (16.0 to 25.0 in d.b.h.)
LgT = large trees (>25.0 in d.b.h.)
Single story (no US) = forested patches without a visible understory
Nonforest = herbland, shrubland, and nonforest/nonrange types

Canopy cover (CC):

10–100 = percentage forest canopy cover class: 10–30, 40–60, 70–100
Single story (no US) = forested patches without a visible understory ("US_CC" tables only)
Nonforest = herbland, shrubland, and nonforest/nonrange types

Table 8 shows a total sampled area of about 415 000 ha, or more than 1 million ac. In the Blue Mountains, about 4 percent of that area resided in the dry PIPO PVT, about 10 percent in the dry mixed conifer (WD Douglas-fir/grand fir), 46 percent in the moist mixed conifer (CM Douglas-fir/grand fir) PVT, 4 percent in the dry spruce-fir PVT (WD subalpine fir/Engelmann spruce), and 8 percent in the moist and cool to cold spruce-fir PVT (CM subalpine fir/Engelmann spruce). The remainder of the area (28 percent) was occupied by early-seral forest conditions (grass, shrub, bare ground), woodlands, and other forested PVTs.

Table 8—Percentage of sampled area in the Blue Mountains ecological reporting unit (ERU) by potential vegetation type

Potential vegetation type	Area	PCT_ERU
	Hectares	*Percent*
Ponderosa pine	15 624.56	3.77
Warm/dry Douglas-fir/grand fir	40 208.19	9.70
Cool/moist Douglas-fir/grand fir	191 632.00	46.22
Warm/dry subalpine fir/Engelmann spruce	16 903.80	4.08
Cool/moist subalpine fir/Engelmann spruce	33 475.22	8.07
Total	297 843.76	71.83

Table 9 examines the distribution of overstory tree size classes (OS) combined with canopy cover class (CC) for each of the five major PVTs. The canopy cover classes are 0 (nonforest), 10 to 30 percent, 40 to 60 percent, and 70 to 100 percent. Canopy cover was originally interpreted in decile classes but reclassified here for simplicity. The size classes are given in the key that precedes these tables. Note the amount of each PVT that was occupied by early-seral nonforest conditions. One quarter of the area in the PIPO PVT was not forested, but in grass, shrub, or bare ground condition at any one time. This was also true of about 10 to 20 percent of the area of the other PVTs. Also observe that there was a relatively small area in SS in any CC class in the PIPO PVT, but this area increased among the other PVTs. When considered along with the nonforest area by PVT, this gives some insight into the amount of stand-replacing disturbance (in grass, shrub, or forested patches that was going on at any time. Note also that regardless of PVT setting, most of the area in any OS size class falls within the lower canopy cover classes. Note as well that the area with MedT or LgT in the overstory consistently ranges from 40 to 60 percent of the PVT area (PIPO = 42.7 percent, WD PSME/ABGR = 55.6 percent, CM PSME/ABGR = 59.75 percent, WD ABLA2/PIEN = 38.2 percent, and CM ABLA2/PIEN = 44.42 percent), and most of that area is in the lower CC classes. Finally, note that each PVT shows a considerable area in PT and SmT trees (PIPO = 31.5 percent, WD PSME/ABGR = 36.1 percent, CM PSME/ABGR = 31.1 percent, WD ABLA2/PIEN = 37.1 percent, and CM ABLA2/PIEN = 46.3 percent), suggesting that mixed- and high-severity fire occurrence has to be a part of the fire history story.

Table 9—Percentage of the sampled area in the Blue Mountains province by historical potential vegetation type, overstory size class, and canopy cover class combination

	Potential vegetation type				
Overstory size × CC	PIPO	WD PSME/ABGR	CM PSME/ABGR	WD ABLA2/PIEN	CM ABLA2/PIEN
			Percent		
SS_10–30	0.09	0.51	0.17	0.22	
SS_40–60		0.01	0.03	0.23	0.22
SS_70–100		0.07	0.10	5.43	1.43
PT_10–30	4.07	3.46	2.15	3.38	3.47
PT_40–60	0.15	0.59	0.82	2.50	2.56
PT_70–100	0.12	0.61	1.65	1.50	3.42
SmT_10–30	25.91	24.04	13.42	12.52	15.19
SmT_40–60	0.69	5.14	7.10	10.10	13.96
SmT_70–100	0.52	2.21	5.95	7.09	7.73
MedT_10–30	35.89	32.30	29.57	12.32	11.80
MedT_40–60	3.34	9.14	14.16	10.95	12.85
MedT_70–100	0.12	1.59	3.66	3.96	6.30
LgT_10–30	3.36	4.29	8.49	6.39	3.28
LgT_40–60		6.37	3.02	3.29	8.06
LgT_70–100		1.97	0.85	1.27	2.14
Nonforest	25.75	7.69	8.85	18.85	7.59
Total	100.00	100.00	100.00	100.00	100.00

CC = canopy cover; PIPO = ponderosa pine; WD PSME/ABGR = warm/dry Douglas-fir/grand fir; CM PSME/ABGR = cool/moist Douglas-fir/grand fir; WD ABLA2/PIEN = warm/dry subalpine fir/Engelmann spruce; CM ALBA2/PIEN = cool/moist subalpine fir/Engelmann spruce; SS = seedlings and saplings; PT = pole-size trees; SmT = small trees; MedT = medium trees; LgT = large trees.

Table 10 examines the distribution of understory tree size classes (US) combined with canopy cover class (CC) for each of the five major PVTs. It is a companion to the previous table and provides insights about the understories of the forested PVTs, when that is the focal point. The canopy cover classes are 0 (nonforest), 10 to 30 percent, 40 to 60 percent, and 70 to 90 percent; obviously 100 percent US CC cannot exist for it to remain a US rather than an OS layer. Canopy cover was originally interpreted in decile classes but reclassified here for simplicity. The size classes are given in the key that precedes these tables. Note that as in table 9, regardless of PVT most of the area showing any US size class or CC occurs in the lowest 10 to 30 percent CC class (PIPO = 46.9 percent, WD PSME/ABGR = 51.9 percent, CM PSME/ABGR = 50.6 percent, WD ABLA2/PIEN = 38.5 percent, and CM ABLA2/PIEN = 49.1 percent). A much smaller sized area existed in any PVT in the 40 to 60 or 70 to 90 percent CC classes (PIPO = 7.6 percent, WD PSME/ABGR = 18.6 percent, CM PSME/ABGR = 24.3 percent, WD ABLA2/PIEN = 13.6 percent, and CM ABLA2/PIEN = 17.1 percent). Note also that in all PVTs, 16 to 29 percent of the patch area shows no evidence of US trees of any size class. Taken together, these results suggest that, for each of the five PVTs, most patch area did not have dense understories, suggesting a strong influence of fire, perhaps in overlapping reburns, and from adjacent drier PVTs (e.g., those surrounding the WD and CM ABLA2/PIEN PVTs) that typically displayed more frequent fire return (the PIPO and WD and CM PSME/ABGR PVTs).

Table 10—Percentage of the sampled area in the Blue Mountains ecological reporting unit (ERU) by historical potential vegetation type, understory size class, and canopy cover class combination

Understory size × CC	Potential vegetation type				
	PIPO	WD PSME/ABGR	CM PSME/ABGR	WD ABLA2/PIEN	CM ABLA2/PIEN
			Percent		
SS_10–30	21.69	18.49	7.25	2.32	6.76
SS_40–60	2.35	6.53	1.55	2.01	2.45
SS_70–90		1.30	0.38	0.24	1.03
PT_10–30	23.45	24.61	26.22	24.94	25.15
PT_40–60	3.96	7.61	10.14	6.84	5.89
PT_70–90	1.20	0.49	1.30	1.56	1.52
SmT_10–30	1.73	8.56	15.47	11.14	17.09
SmT_40–60	0.04	2.14	8.44	2.62	5.50
SmT_70–90		0.47	0.98	0.18	0.69
MedT_10–30		0.21	1.62	0.06	0.09
MedT_40–60		0.05	1.26	0.12	
MedT_70–90			0.20		
Without any understory	19.83	21.85	16.34	29.10	26.24
Nonforest	25.75	7.69	8.85	18.85	7.59
Total	100.00	100.00	100.00	100.00	100.00

CC = canopy cover; PIPO = ponderosa pine; WD PSME/ABGR = warm/dry Douglas-fir/grand fir; CM PSME/ABGR = cool/moist Douglas-fir/grand fir; WD ABLA2/PIEN = warm/dry subalpine fir/Engelmann spruce; CM ALBA2/PIEN = cool/moist subalpine fir/Engelmann spruce; SS = seedlings and saplings; PT = pole-size trees; SmT = small trees; MedT = medium trees; LgT = large trees.

Table 11 examines the areal distribution of cover types (CT) for each of the five major PVTs. For an expanded description of the cover types see Hessburg et al. 1999a (table 7: 48–49). Note that the PIPO cover type existed in all PVTs. It was dominant in the PIPO (65.5 percent), WD PSME/ABGR (79.42 percent), and CM PSME/ABGR (44.08 percent) PVTs, and occurred in trace amounts in the ABLA2/PIEN types. Surprisingly, more area in the PIPO CT occurred in the WD PSME/ABGR than in the PIPO PVT. This was driven by the large area (about 26 percent) in patches in a nonforest physiognomy (grass and shrub cover) in the PIPO PVT. Looking across all PVTs, it appears that grass and shrub vegetation patches were integral to the forest patchwork; many patches occurred in all forested PVT settings that were otherwise capable of supporting the growth of forest cover. Undoubtedly, some of these patches were alternative stable states that experienced relatively frequent fires and experienced difficulty establishing a dominant cover of trees. In contrast with the PIPO and WD PSME/ABGR PVTs, CT conditions in the CM PSME/ABGR PVT are increasingly complex, with 47 percent of the patch area in PSME, LAOC, ABGR, and other forest cover types. Nearly one-quarter of the CM PSME/ABGR PVT area was dominated by the ABGR (ABCO) CT, second only to the PIPO CT. A similar area in the ABGR CT occurred in the WD and CM ABLA2/PIEN PVTs, but the greatest difference was that the other forest CTs dominated the latter PVTs. These observations coupled with those from the preceding three tables suggest that several findings are plausible: (1) surface fires associated with low- and mixed-severity fires likely dominated the fire regime in the PIPO and WD PSME/ABGR PVTs; however, replacement-severity fires also occurred and they provided a nontrivial contribution; (2) low-, mixed- and high-severity fires likely occurred in the CM PSME/ABGR PVT owing to variability in fire frequency and topo-edaphic conditions, and surface fire effects from low- and mixed-severity fires were likely on par with crown fire effects from mixed- and high-severity fires; and (3) crown fire effects from mixed- and high-severity fires likely dominated in the ABLA2/PIEN PVTs.

Table 11—Percentage of the sampled area in the Blue Mountains ecological reporting unit (ERU) by historical potential vegetation type and cover type

	Potential vegetation type				
Cover type	**PIPO**	**WD PSME/ABGR**	**CM PSME/ABGR**	**WD ABLA2/PIEN**	**CM ABLA2/PIEN**
			Percent		
PIPO	65.48	79.42	44.08	1.48	0.45
PSME	0.80	4.17	15.26	10.09	4.98
LAOC	0.52	0.96	4.93	9.59	6.57
ABGR	0.06	1.45	22.67	17.82	26.07
Other forest cover types	7.39	6.30	4.21	42.17	54.33
Nonforest	25.75	7.69	8.85	18.85	7.59
Total	100.00	100.00	100.00	100.00	100.00

PIPO = ponderosa pine; WD PSME/ABGR = warm/dry Douglas-fir/grand fir; CM PSME/ABGR = cool/moist Douglas-fir/grand fir; WD ABLA2/PIEN = warm/dry subalpine fir/Engelmann spruce; CM ALBA2/PIEN = cool/moist subalpine fir/Engelmann spruce; PSME = Douglas-fir; LAOC = western larch; ABGR = grand fir.

Table 12 layers together in one location the relations existing in three of the four previous tables, excluding table 10. We integrate them all in table 13. From table 11 we saw that 65.5 percent of the PIPO PVT area supported the PIPO CT, and 26 percent of the area was nonforested. Of that 65.5 percent, 86 percent (56.3 percent of the PVT area) existed in two classes: the SmT and MedT_10 to 30 percent CC classes. Two aspects are noteworthy: canopy cover is low, in many cases less than full potential site occupancy and size class is dominated by small- and medium-sized, not large, and very large trees. In the photointerpretation, large trees were in evidence in a great many patches. To be identified as an overstory layer, LgT CC had to meet or exceed 10 percent of the CC, the lowest CC class. Where LgT cover did not meet the 10 percent CC threshold, the LgT cover was considered a remnant of a former patch and its CC was pooled with the dominant overstory size class. If, as the literature suggests, surface fires primarily thin from below but contribute <20 percent overstory mortality, then surface fire dominated regimes continuously regenerate patches affected by them, and they manifest at any point in time as multicohort, multi-aged, multisized patches. Put alternatively, if fires are occurring every 5 to 15 years in the PIPO and WD PSME/ABGR PVTs and they conservatively kill on average just 5 percent of the CC at each instance of fire, it is difficult to imagine finding many patches of PIPO CT with a high canopy cover of mostly LgT, and with many trees older than about 350 years of age. That is what is shown here. Notice the strong similarities between the PIPO and WD PSME/ABGR PVTs—but there are differences as well. In both PVTS one can see the powerful influence of surface fires in maintaining low canopy cover conditions; however, in the WD PSME/ABGR PVT, the story is less tidy; nearly 28 percent of the area is comprised of patches with a broader array of size classes, intermediate and higher canopy cover classes, and a broader range of cover types. It is apparent that fire frequency and fire severity are more variable in the WD PSME/ABGR than the PIPO PVT. This pattern continues into the CM PSME/ABGR PVT, where there is greater evidence of CT, size, CC, and hence fire regime variability.

Table 12—Percentage of the sampled area in the Blue Mountains ecological reporting unit (ERU) by historical potential vegetation type, cover type, size class, and canopy cover class

Cover type	PIPO	WD PSME/ABGR	CM PSME/ABGR	WD ABLA2/PIEN	CM ABLA2/PIEN
	Percent				
PIPO:					
SS_10-–30		0.46	0.16		
SS_40–60		0.01			
SS_70–100		0.04	0.02		
PT_10–30	2.05	1.56	1.24	0.53	
PT_40–60	0.06	0.42	0.34		
PT_70–100	0.12	0.51	0.14		0.09
SmT_10–30	23.71	20.66	8.60	0.17	0.26
SmT_40–60	0.29	3.47	3.00	0.18	0.02
SmT_70–100		1.69	1.25		
MedT_10–30	32.56	28.75	17.93	0.51	0.08
MedT_40–60	3.26	8.22	6.56		
MedT_70–100	0.12	1.44	0.35		
LgT_10–30	3.31	3.91	3.73	0.09	
LgT_40–60		6.32	0.66		
LgT_70–100		1.94	0.09		
PSME:					
SS_10–30			0.01	0.12	
SS_40–60			0.02	0.23	
SS_70–100			0.04		
PT_10–30	0.16	0.46	0.32		0.06
PT_40–60	0.09	0.07	0.12	0.63	
PT_70–100			0.10		0.06
SmT_10–30	0.32	0.90	1.55	1.65	0.36
SmT_40–60	0.04	0.72	1.48	1.55	1.21
SmT_70–100		0.11	1.14	1.24	0.25
MedT_10–30	0.06	1.06	4.74	1.94	2.06
MedT_40–60	0.07	0.62	1.82	1.38	0.52
MedT_70–100		0.15	0.66	0.25	0.04
LgT_10–30	0.05	0.04	2.69	0.96	0.41
LgT_40–60		0.05	0.51	0.16	
LgT_70–100			0.06		
LAOC:					
SS_10–30					
SS_40–60			0.01		0.07
SS_70–100		0.02	0.01	3.31	0.28
PT_10–30		0.03	0.11		0.12
PT_40–60		0.03	0.16		0.19
PT_70–100		0.04	0.27	0.26	0.16
SmT_10–30		0.16	0.55	1.38	1.41
SmT_40–60		0.11	0.24	0.11	1.29
SmT_70–100	0.52	0.08	0.47	0.11	0.36
MedT_10–30		0.47	1.72	0.86	1.54
MedT_40–60		0.01	0.59	2.70	0.59
MedT_70–100			0.24	0.05	0.10

Table 12—Percentage of the sampled area in the Blue Mountains ecological reporting unit (ERU) by historical potential vegetation type, cover type, size class, and canopy cover class (continued)

	Potential vegetation type				
Cover	PIPO	WD PSME/ABGR	CM PSME/ABGR	WD ABLA2/PIEN	CM ABLA2/PIEN
	Percent				
LgT_10–30			0.20	0.80	0.46
LgT_40–60			0.36		
LgT_70–100					
ABGR:					
SS_10–30					
SS_40–60					
SS_70–100					0.08
PT_10–30			0.28	0.25	0.36
PT_40–60			0.02	0.14	0.69
PT_70–100			0.24	0.36	0.07
SmT_10–30	0.03	0.01	2.26	1.77	2.04
SmT_40–60		0.67	2.25	3.03	4.70
SmT_70–100		0.14	2.54	1.51	2.84
MedT_10–30	0.03	0.17	3.82	2.30	2.40
MedT_40–60		0.23	5.12	2.48	4.16
MedT_70–100			2.40	2.02	3.21
LgT_10–30		0.20	1.58	1.27	0.27
LgT_40–60			1.48	2.24	3.93
LgT_70–100		0.03	0.68	0.45	1.35
Other forest cover types:					
SS_10–30	0.09	0.05		0.10	
SS_40–60					0.15
SS_70–100			0.03	2.11	1.07
PT_10–30	1.86	1.42	0.21	2.61	2.93
PT_40–60		0.07	0.18	1.72	1.67
PT_70–100		0.06	0.90	0.87	3.05
SmT_10–30	1.85	2.31	0.46	7.56	11.12
SmT_40–60	0.36	0.16	0.12	5.23	6.74
SmT_70–100		0.18	0.55	4.23	4.28
MedT_10–30	3.23	1.86	1.36	6.72	5.72
MedT_40–60		0.06	0.07	4.40	7.58
MedT_70–100			0.01	1.65	2.95
LgT_10–30		0.13	0.28	3.27	2.15
LgT_40–60				0.89	4.14
LgT_70–100			0.02	0.82	0.79
Nonforest	25.75	7.69	8.85	18.85	7.59
Grand total	100.00	100.00	100.00	100.00	100.00

CC = canopy cover; PIPO = ponderosa pine; WD PSME/ABGR = warm/dry Douglas-fir/grand fir; CM PSME/ABGR = cool/moist Douglas-fir/grand fir; WD ABLA2/PIEN = warm/dry subalpine fir/Engelmann spruce; CM ALBA2/PIEN = cool/moist subalpine fir/Engelmann spruce; PSME = Douglas-fir; LAOC = western larch; ABGR = grand fir; SS = seedlings and saplings; PT = pole-size trees; SmT = small trees; MedT = medium trees; LgT = large trees.

In table 13, we layer the information from all five of the preceding tables together using structural classes of O'Hara et al. (1996). Structural classes are used because they provide a sense of the relative abundance of components of each PVT landscape resulting from the combined influences of forest succession, stand dynamics, and disturbance processes. The structural classes take the continuum of conditions and bin them into key mileposts in succession and disturbance regimes. Toggling between tables 12 and 13 is useful for visualizing this point. In the PIPO PVT, we see that stem exclusion open canopy structures (seoc, 31.4 percent) dominated, occupying nearly one half of all PIPO cover (31.4/65.5). This was followed by stand initiation structure (si, 21.2 percent), grass and herb patches (herbland, 15.62 percent), shrubland patches (9.8 percent) woodland patches where tree cover (PIPO and JUOC, western juniper) was <30 percent CC (woodland), followed by young multistory forest (yfms), and understory reinitiation (ur, 3.32 percent) structures. It is plausible that PIPO_si cover occurred in patches after severe fires or that PIPO forest was reinvading a much larger area of grass, shrub, and woodland patches under the influence of livestock grazing, fire suppression, and fire exclusion. At the time of the photointerpretation, only 2.5 percent of the total area showed any evidence of prior timber harvest, and most of this was light selection cutting. Conditions were similar to the PIPO PVT in the WD PSME/ABGR PVT; however, more closed canopy conditions are present and structural conditions are spread across a broader range of CTs. From table 11, we note that nearly 80 percent of the area in the WD PSME/ABGR PVT displayed the PIPO CT. This was distributed across all structural classes, but not evenly so; 59 percent was comprised on PIPO_si, PIPO_seoc, and PIPO_yfms conditions, PIPO_ur contributed another 9 percent, and old single and multistory forests contributed another 10 percent. At the time of the photointerpretation, one third of the WD PSME/ABGR PVT area had experienced some noticeable selection cutting. We notice a much broader distribution of CT and structural class combinations when we observe the CM PSME/ABGR PVT. Here, 33 and 23 percent of the area is comprised of yfms and seoc structures, respectively, across all cover types. At the time of the photointerpretation, 15 percent of this PVT area had experienced light selection cutting, and there was little evidence of regeneration harvesting. In both the WD and CM ABLA2/PIEN PVTs, there was trace evidence of selection and regeneration harvests. In most cases, regardless of PVT, the selection cutting dominated the observed logging in the early photos, it was obvious and recent, and the largest and oldest remnant large and very large trees were the target of opportunity.

Table 13—Percentage of the sampled area in the Blue Mountains ecological reporting unit (ERU) by historical potential vegetation type, cover type, and structural class

	Potential vegetation type				
Cover × structure	PIPO	WD PSME/ABGR	CM PSME/ABGR	WD ABLA2/PIEN	CM ABLA2/PIEN
			Percent		
PIPO:					
si	21.23	16.59	3.06	0.06	
seoc	31.35	27.60	11.95	0.78	0.14
secc	0.04	2.48	1.56	0.18	0.23
ur	3.32	8.72	7.70	0.19	0.08
yfms	9.44	14.51	17.64	0.27	
ofms	0.11	5.60	1.40		
ofss		3.91	0.78		
PSME:					
si		0.49	0.68	1.11	0.11
seoc	0.29	2.09	2.63	2.72	1.32
secc		0.15	1.81	1.98	0.55
ur	0.44	0.54	3.19	1.68	1.57
yfms	0.07	0.85	6.15	2.32	1.44
ofms		0.03	0.28	0.27	
ofss		0.02	0.53		
LAOC:					
si		0.02	0.45	3.85	1.07
seoc		0.13	0.14	0.11	0.27
secc	0.52	0.13	0.93	1.59	1.51
ur		0.53	1.54	2.94	2.67
yfms		0.15	1.40	0.41	1.05
ofms			0.46	0.68	
ofss			0.01		
ABGR:					
si			0.11		0.23
seoc	0.06	0.18	7.20	4.22	4.07
secc		0.50	2.65	3.15	4.74
ur		0.31	3.98	3.82	5.78
yfms		0.23	6.11	3.58	5.89
ofms		0.20	0.59	0.56	1.28
ofss		0.03	2.03	2.48	4.08
Other forest cover types:					
si	0.39	0.77	0.13	3.12	4.00
seoc		0.11	0.71	16.18	15.61
secc		0.25	1.14	8.23	7.97
ur		0.49	0.80	6.79	9.77
yfms		0.71	1.28	5.64	10.63
ofms				0.42	2.09
ofss			0.02	1.77	4.27
Woodland	7.00	3.99	0.14	0.01	
Nonforest					
Herbland	15.62	3.27	7.45	13.74	4.56
Shrubland	9.80	4.16	0.79	1.39	1.15
Nonforest/nonrange	0.34	0.27	0.61	3.73	1.89
Total	100.00	100.00	100.00	100.00	100.00

PIPO = ponderosa pine; WD PSME/ABGR = warm/dry Douglas-fir/grand fir; CM PSME/ABGR = cool/moist Douglas-fir/grand fir; WD ABLA2/PIEN = warm/dry subalpine fir/Engelmann spruce; CM ALBA2/PIEN = cool/moist subalpine fir/Engelmann spruce; PSME = Douglas-fir; LAOC = western larch; ABGR = grand fir; si = stand initiation; seoc = stem exclusion, open canopy; secc = stem exclusion, closed canopy; ur = understory reinitiation; yfms = young forest, multistory; ofms = old forest, multistory; ofss = old forest, single story.

Glossary

Adaptive management—A system of management practices based on clearly identified outcomes and monitoring to determine if management actions are meeting desired outcomes, and if not, to facilitate management changes that will best ensure that outcomes are met or reevaluated. Adaptive management stems from the recognition that knowledge about natural resource systems is sometimes uncertain (36 CFR 219.16; FSM 1905).

Composition—The biological elements within the different levels of biological organizations, from genes and species to communities and ecosystems (FSM 2020).

Connectivity (or habitat connectivity)—The degree to which intervening landscape characteristics impede or facilitate the movement of organisms or ecological processes between patches. The concept of connectivity implies that some feature or patch in the landscape is spatially related to other, similar features, and that intervening landscape characteristics influence the ecological relationship between those features. Although somewhat counterintuitively, it is important to note that a landscape can be highly fragmented or patchy, as is commonly the case in landscapes with mixed-severity fire regimes, and still be highly connected for a variety of ecological processes.

Disturbance—A force that causes significant change to the structure, composition, or function of an ecosystem through natural events such as earthquake, fire, flood, insect and disease outbreaks, weather, or wind; or by human activities such as the harvest of forest products. Many or most disturbances of interest are integral parts of and important to ecosystem function and health.

Ecophysiological—Relating to the interrelationships between an organism's function and its physical environment.

Ecosystem services—Benefits people obtain from ecosystems (FSM 2020), including:

1. **Provisioning services** such as clean air and fresh water, as well as energy, fuel, forage, fiber, and minerals;
2. **Regulating services** such as long-term storage of carbon; climate regulation; water filtration, purification, and storage; soil stabilization; flood control; and disease regulation;
3. **Supporting services** such as pollination, seed dispersal, soil formation, and nutrient cycling;
4. **Cultural services** such as educational, aesthetic, spiritual and cultural heritage values, as well as recreational experiences and tourism opportunities.

Elasticity—The speed with which a system returns after disturbance.

ENSO—El Niño/Southern Oscillation, a band of anomalously warm ocean water temperatures that occasionally develops off the western coast of South America and can cause climatic changes across the Pacific Ocean. The extremes of this climate pattern's oscillations cause extreme weather (such as floods and droughts) in many regions of the world.

Epicormic—Literally, "of a shoot or branch," this term implies growing from a previously dormant bud on the trunk or a limb of a tree.

Fire intensity—The amount of energy or heat release during fire.

Fire regime—The characteristics of fire in a given ecosystem, such as the frequency, severity, extent, and seasonality of fire.

Fire severity—The scale at which vegetation and a site are altered or disrupted by fire, from low to high severity. It is a combination of the degree of fire effects on vegetation (amount of fire caused vegetation mortality) and on soil properties.

Fragmentation—A term that has various meanings depending on the context of its use. Here we define it in two related ways (also see "connectivity"):

> **Landscape fragmentation**—The breaking up of continuous habitats into patches, thereby generating habitat loss, isolation, and edge effects.
>
> **Wildlife habitat fragmentation**—The set of mechanisms leading to the discontinuity in the spatial distribution of resources and conditions present in an area at a given scale that affects occupancy, reproduction, and survival in a particular species.

Frequency distribution—A depiction, often appearing in the form of a curve or graph, of the frequency with which possible values of a variable have occurred. In this report, we speak of the frequency of wildfire patches of various sizes.

Function—Ecological processes, such as energy flow; nutrient cycling and retention; soil development and retention; predation and herbivory; and natural disturbances such as wind, fire, and floods that sustain composition and structure (FSM 2020).

Future range of variation (FRV)—The natural fluctuation of pattern components of healthy ecosystems that may occur in the future, primarily affected by climate change, human infrastructure, and invasive species.

Heterogeneity—Any factor that induces variation in individual demographic rates.

Hierarchy theory—Ecological hierarchy theory presupposes that nature is working at multiple scales and has different levels of organization which are part of a rate-structured, nested hierarchy.

Historical range of variation (HRV)—The natural fluctuation in pattern of components of ecosystems over time.

Inertia—The resistance of a system to disturbance.

Integrity—An ecosystem has integrity if it retains its complexity, intact biotic and abiotic processes, capacity for self-organization, and sufficient diversity, within its structures and functions, to maintain the ecosystem's self-organizing complexity through time.

Interdigitate—The interlocking of (in this case) different components of the landscape like the fingers of two clasped hands. This creates an intertwining or intermingling mosaic of different vegetation or habitat conditions.

Landscape—A large land area composed of interacting ecosystems that are repeated in terms of factors such as geology, soils, climate, and human impacts. Landscapes are often used for coarse grain analysis.

Landscape hierarchy—Landscapes are systems that can be divided or decomposed into a hierarchy of nested geographic (i.e., different sized) units. This organization provides a guide for defining the functional components of a system and defines ways components at different scales are related to one another.

One common way this is conceptualized is the following organization: region = millions of hectares; landscapes = tens of thousands of hectares; watersheds = hundreds to thousands of hectares; stands (patches) = one to tens of hectares; gaps = 0.01 to 0.1 ha.

Mosaic—The contiguous spatial arrangement of elements within an area. For regions this is typically the upland vegetation patches, large urban areas, large bodies of water, and large areas of barren ground or rock. However, regional mosaics can also be land ownership, habitat patches, land use patches, or other elements. For landscapes, this is typically the spatial arrangement.

Multi-aged stands—Stands having two or more age classes and including stands resulting from variable-retention systems or other traditionally even-aged systems that leave residual or reserve trees.

Nested hierarchy—The name given to the hierarchical structure of groups within groups or branches from a trunk used to classify organisms.

Orographic—The lift of an air mass when it is forced from a low elevation to a higher elevation as it is moves over rising terrain. As the air mass gains altitude, it cools, its relative humidity increases, and clouds and sometimes precipitation result.

Patch—An area of similar vegetation, in structure and composition. Patches may range in size from a portion of a hectare to thousands of hectares.

PDO—The Pacific Decadal Oscillation is often described as a long-lived El Niño-like pattern of Pacific climate variability. As seen with the better-known El Niño/Southern Oscillation (ENSO), extremes in the PDO pattern are marked by widespread variations in Pacific Basin and North American climate.

Plant association—Plant associations are a finer level of classification in the potential vegetation hierarchy. They are defined in terms of a climax dominant overstory tree species and an understory herb or shrub species that is typical of the environmental conditions of a distinctive community of plants.

Plant association group (PAG)—A group of potential vegetation types that have similar environmental conditions and are dominated by similar types of plants (e.g., the dry shrub PVG). They are often grouped by similar types of life forms.

Potential vegetation type (PVT)—A potential vegetation type is a kind of physical and biological environment that produces a kind of vegetation, such as the dry Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) type. Potential vegetation types are identified by indicator species of similar environmental conditions. For example, Douglas-fir indicates a cooler and moister environment than ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson). Because of growth, mortality, and disturbance of the vegetation, many other kinds of vegetation will occur on this type through time. In many cases the indicator species will not be present, due to disturbance. Douglas-fir is simply an indicator, and name, for the kind of physical and biological environment stratification that is used for prediction of response.

Resilience—The capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks (FSM Chapter 2020). Resilience can be further defined to mean the amount of disturbance that an ecosystem can withstand without changing functional states. In this context, while dominant floristics may vary, a forest remains a forest as exemplified by maintenance of certain characteristic biological composition and the ecological goods and services it produces. Resiliency is the inherent capacity of a landscape or ecosystem to maintain its basic structure, function, and organization in the face of disturbances, both common and rare.

Restoration—The Forest Service defines restoration (National Forest System Land Management Planning, 36 CFR 219.19) as the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. Ecological restoration focuses on reestablishing the composition, structure, pattern, and ecological processes necessary to facilitate terrestrial and aquatic ecosystem sustainability, resilience, and health under current and future conditions.

Stand—A descriptor of a land management unit consisting of a contiguous group of trees sufficiently uniform in age-class distribution, composition and structure, and growing on a site of sufficiently uniform quality, to be a distinguishable unit.

Structure—The organization and physical arrangement of biological elements such as snags and down woody debris, vertical and horizontal distribution of vegetation, stream habitat complexity, landscape pattern, and connectivity (FSM 2020).

Sustainability—Meeting needs of the present generation without compromising the ability of future generations to meet their needs. Sustainability is composed of desirable social, economic, and ecological conditions or trends interacting at varying spatial and temporal scales, embodying the principles of multiple use and sustained yield (FSM 1905). Conditions that support native species, ecosystem services, and ecological processes are sustainable when influences on them have not resulted in significant depletion or permanent damage.

Topo-edaphic—Related to or caused by particular soil conditions, as of texture or drainage, rather than by physiographic or climatic factors within a defined region or area.

Variable density thinning—The method of thinning some sub-stand units to a different density than other sub-stand units.

Vegetation series (plant community)—An assembly of different species of plants growing together in a particular habitat; the floral component of an ecosystem. According to Powell et. al. (2007 page 12), a series is the highest level of a potential vegetation hierarchy and is defined by the dominant climax plant species.

Vegetation type—A plant community with distinguishable characteristics.

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