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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**

**Authors**

**Stine, Peter (USDA Forest Service, PSW)**

**Hessburg, Paul (USDA Forest Service, PNW)**

**Spies, Thomas (USDA Forest Service, PNW)**

**Kramer, Marc (University of Florida)**

**Fettig, Chris (USDA Forest Service, PSW)**

**Hansen, Andy (Montana State University)**

**Lehmkuhl, John (USDA Forest Service, PNW. retired)**

**O’Hara, Kevin (UC Berkeley)**

**Polivka, Karl (USDA Forest Service, PNW)**

**Singleton, Peter (USDA Forest Service, PNW)**

**Charnley, Susan (USDA Forest Service, PNW)**

**Merschel, Andrew (Oregon State University)**

**** John Marshall

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**Executive Summary**

Millions of hectares of America’s forests have been negatively impacted by drought and insect and disease outbreaks, and are overloaded with fuel, priming them for unusually severe and large wildfires. In light of these trends, public support for forest restoration has grown. One of the top priorities of the USDA Forest Service is to restore resiliency to forest and range ecosystems, enabling them to cope with an array of disturbance factors. The underlying objective of forest management, and specifically forest restoration, is to promote continued delivery of desired ecosystem goods and services.

Natural resource managers and policy makers are already awash in information from the growing body of science, with little time to sort through it, let alone assimilate many different sources. Regional research and management executives have requested a succinct review of information on eastside moist mixed-conifer forests within the context of the broader forest landscape. 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 moist mixed-conifer 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. We therefore produced this science synthesis that compiles existing research, makes connections across disparate sources, and addresses multilayered natural resource issues. This is provided to land managers to assist in updating existing management plans and on-the-ground projects intended to promote resilience in moist mixed-conifer forests. We consider management flexibility at the local scale 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 social and policy concerns.

Key sections of the report include:

* A description of moist mixed-conifer forests and its context in the broader landscape
* Key concepts of restoration and the landscape perspective
* A comprehensive summary of pre-European settlement moist mixed-conifer forest ecology
* A description of the socio-economic context in the region
* A summary of human impacts to moist mixed-conifer forests
* Broad management implications
* A practical list summarizing management considerations for diagnosing restoration needs and designing landscape approaches

**Moist mixed-conifer forests**

Mixed-conifer forests are a component of the broader dry-to-wet conifer forest complex that is widely distributed across eastern Oregon and Washington. These forests are important for, among other factors, carbon sequestration, watershed protection, wildlife habitat, and outdoor recreation, and they provide economic opportunities through provisioning of a wide variety of forest products. Moist mixed-conifer forests cover a large area east of the crest of the Cascades in Oregon and Washington where grand fir, white fir, and Douglas-fir are the dominant late-successional tree species. The moist mixed-conifer forests can be considered intermediate between drier conifer forests where pine was 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 moist mixed-conifer 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.

Many decades of wildfire exclusion, domestic livestock grazing, and selective timber harvesting have interacted to alter the structure, composition and disturbance regimes of these forests. Moist mixed-conifer forests have become denser, have lost large individuals of fire-resistant tree species, and on many sites have become dominated by dense grouping of shade-tolerant species that are less resistant to fire and less resilient to drought. These vegetation changes and management activities have shifted fire regimes toward increasingly infrequent, larger and more severe fires, which tend to simplify the landscape into fewer, larger, and less diverse patches resulting in more homogenous landscapes. Many mixed-conifer forests currently are denser, 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 vary 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 supported by credible research that is relevant to practitioners and others interested in the management of moist mixed-conifer forests. Some of our findings include:

**The historical range of variation is useful as a guide but not a target.** Returning to it is no longer feasible or practical because of changing climate, land use, and altered forest structure and composition. The contemporary concept of restoration goes beyond the oft-stated goal of re-establishing 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 to achieve desired future ecosystem conditions for a landscape.

**Disturbance regimes are 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 moist mixed-conifer 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. These forests neither resemble nor function as they did 200 years ago.

**Mixed moist-conifer forests are more vulnerable to large, high-severity fire and insects outbreaks.** Widespread anthropogenic changes have created more homogenized conditions in this forest type, generally in the form of large, dense and multi-layered patches of fire-intolerant tree species. These changes have substantially altered the resilience mechanisms associated with moist mixed-conifer forests.

**Patterns of vegetation structure and composition in an eastside forest landscape shaped by intact disturbance regimes are diverse and vary 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. Despite the change and variability, topography, soils, and elevation constrain these vegetation patterns and provide a relatively simple 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 can capitalize on these tendencies.

**Several wildlife species of conservation concern require structural complexity typical of mature and old forests that are currently limited or at risk.** 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 under climate change. Restoration at a landscape scale will have challenges in retaining current patches of 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 national forest system lands 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.

**Next Steps**

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. Decision makers also must weigh trade-offs 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 the best 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 in specific management contexts. It is also the role of research to work alongside managers in the conduct of management and seek 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 local 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.

We draw the reader’s attention to the first subsection, 5.a, for a summary of the specific key findings. Here we synthesize the principle findings that have been gleaned from the body of scientific literature (summarized in Section 4) as they pertain to management of moist mixed-conifer forests. These constitute the “take home” messages that are intended to assist land managers**.**

The social agreement and institutional capacity for restoring moist mixed-conifer 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.

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.

Our hope is that this synthesis can serve as a comprehensive reference that provides a condensed and integrated understanding of the current state of knowledge regarding moist mixed-conifer forests, as well as an extensive list of published sources where readers can find further information. But 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 moist mixed-confer forests in eastern Oregon and Washington.

**SECTION 1 – INTRODUCTION**

**1.a Purpose and scope of this synthesis**

 The 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. 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 climatic warming. In the last few decades, many of these forests have also become further drought-stressed and increasingly vulnerable to high-severity fire (Westerling 2006) and insect outbreaks as a result of climate change (fig. 1).

Regional research and management executives have reported the need to update management guidance, scientific synthesis, and consensus among stakeholders regarding the challenge of restoring these ecosystems at risk. This synthesis responds to their need. In particular, they asked us to focus on eastside moist mixed-conifer (MMC) forests (see definition in section 2.a), 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 expertise, spanning a range of disciplines including landscape ecology, fire and forest ecology, wildlife biology, aquatic ecology and fish biology, disturbance ecology, and climate change.

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

• Vegetation, landscape, and disturbance ecology

• Wildlife habitats and populations; aquatic ecosystems and associated species

• 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 leads to consider the unique ecological conditions of each landscape, and to determine how this information guides specific land management needs. 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), one of several contemporary efforts that apply current science to planning and 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*) forest types, research and social license to begin restoration is relatively more advanced (e.g., see Jain et al. 2012, Franklin et al. 2013), and managers are proceeding with restoration treatments (e.g., see OWNF 2012, Hessburg et al. 2013). However, we include the pine and dry mixed-conifer types in our discussions because the dry and moist mixed-conifer types are intertwined spatially and ecologically.

**1.b Current management context and restoration mandate**

 Federal and state forest managers are charged with maintaining and restoring diverse, resilient (see the Glossary), and productive forests in these fire-prone landscapes. Restoration can improve the resilience of eastside MMC forests and avoid high ecological costs of uncharacteristic wildfire, fire suppression, and post-fire treatments, along with the ecological and socio-economic threats to adjoining state, private and tribal lands. But achieving these goals will be a challenge without 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 commercial products cannot cover the costs of noncommercial activities (e.g., pre-commercial thinning).

Developing and implementing sound management strategies requires up-to-date knowledge of the ecological, biological, and physical processes that naturally regulate forest ecosystem structure and function and how humans have modified them, 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 100-150 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 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 climate change. This is true of all mixed-conifer forests (including MMC), the subject of this synthesis.
* In many areas, 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. This has reduced fire and drought tolerance of these forests. Large increases in surface and canopy fuel loads are widespread, resulting in elevated 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.
* Climate change is in the process of transforming forests in eastern Oregon and Washington because it is linked to ongoing drought, insect, and disease mortality, and wildfires. This process of transformation is occurring at a brisk pace, and the window of opportunity for affecting 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), in order to conserve its dense, late-successional and old-forest nesting, roosting, and foraging habitats.
* Much of the impetus for maintaining large areas of habitat for the NSO on the east side is related to the clearcut harvest of most historical old forest habitat in the western Cascade and Coast Ranges. The current approach attempts to buy time for the owl on the east side as the west side regrows suitable habitat. Currently, large areas of eastside late-successional and old forest habitat are out of sync with the native fire regime, even as the climate warms; this is not true of the west side, where ½ to ¾ of the forested area was historically old forest at any one time.
* 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 will generally not work. Instead, considerations of local conditions, land use histories, and biophysical and land-use classifications can help address this variability.
* Managers report that definitions, classifications, and maps of forest types across the east side are not consistent. They 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 will not likely subside, and is driven by wide-ranging and often competing or conflicting societal values. 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.

 Current U.S. Forest Service priorities across the western United States include focusing efforts in three major areas: 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]](#footnote-1) implies re-enabling 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 re-establish 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.

 The concept of restoration includes, but is larger than the goal of re-establishing the historical range of variability, or the HRV. *A process-oriented approach supports restoring ranges of conditions that have occurred in the past (e.g., eradicating invasive species and reconnecting fragmented habitat 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 is to create resilient forests and rangelands that are adapted to a 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, ecological restoration must also be socially aware and socially connected. For some landscapes, it will simply be impossible and inadvisable to return them to prior conditions (Harris et al. 2006), while with 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 [ICBEMP] 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 socio-economic vantage points would be best considered in the context of regional and local landscape patterns and processes, because the natural system must support to the social system in the long term.

**1.c Structure of the report – where to find sections of interest**

Overview of contents

 Given the diversity of knowledge, interests, and experience of our readers, the report progresses from a review of the relevant science to a presentation of management considerations.

 Section 2 discusses how MMC types fit within a regional coniferous mosaic. In this section, the reader will understand our definition of MMC forests, what forest types are typically included within this classification, and generally where they are located. Section 3 presents ecological concepts that are foundational to landscape restoration.

 Section 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 scientific findings are also summarized in Section 4.d.

 Section 5 is the core of this report. This is where the reader will find 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.

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

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

**SECTION 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 Cascades in Oregon and Washington where grand fir (*Abies grandis*), white fir (*Abies concolor*), and Douglas-fir (*Pseudotsuga menziesii*) are the potential late-successional tree species (fig. 3). Depending upon the environment and local site climate, **patches of MMC forest typically occur where the *current* vegetation is mixture of shade-intolerant ponderosa pine (*Pinus ponderosa*), and/or western larch (*Larix occidentalis*) and shade-tolerant Douglas-fir, white or grand fir, and occasionally Englemann spruce (*Picea engelmannii*), as in the moist grand fir zone of the Blue Mountains (Powell 2007)**. In areas of complex, dissected topography, the MMC type intermingles with ponderosa pine and dry mixed-conifer types and wetter or cooler mixed-conifer types (Powell et al. 2007, Simpson 2007) (fig. 4). Other conifers occasionally associated with MMC type include lodgepole pine (*Pinus contorta*), western white pine (*Pinus monticola*), sugar pine (*Pinus lambertiana*), Shasta red fir (*Abies magnifica*), Pacific silver fir (*Abies amabilis*), and western hemlock (*Tsuga heterophylla*). Dry ponderosa pine and dry mixed-conifer conditions tend to occupy lower montane settings, ridgetops, and southern exposures, while MMC conditions typically occur in mid-to upper montane settings; on northerly and southerly aspects, especially in the upper elevations; in valley bottoms; and in lower headwall positions.

 A **defining element of MMC forest from a restoration perspective is the degree to which fire regimes and the composition and structure of the forest have been altered from their historical range of variability.** MMC forest historically experienced frequent to moderately frequent fires (<20-50 years), and fire severity was typically low or mixed, but patches of high-severity fire also occurred. In most parts of the MMC forest, this fire frequency has been suspended, and disturbance regimes altered through a combination of historical drivers including grazing, loss of Native American fire ignitions, and active fire suppression. Most MMC forests today contain a significant component of shade-tolerant species (e.g., white or grand fir or understory Douglas-fir). Under historical or more fire- and drought-resilient states these shade tolerants would have been absent or uncommon in the understory in many areas.

Dry mixed-conifer and ponderosa pine sites typically experienced frequent fire (<10-25 years), but may have had a relatively open forest mosaic structure that was similar to MMC sites. Wetter or cooler mixed-conifer sites experienced longer fire return intervals (>50-150 years), and greater frequency of higher severity fire, and would have had a component of older shade-tolerant trees in the overstory, with dense patches of multi-storied forest. Because few detailed fire history studies exist for this type, we use potential vegetation types (PVTs, see discussion below) as a surrogate for the fire regime and the degree to which MMC forests are departed in terms of composition and structure. These PVTs can be a first approximation of the environments and locations of MMC forest (table 1). The PVTs used by agency managers—often the best available source of information—are only an approximate surrogate for the fire regime of a site or local landscape. 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, small, moist 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 appeared in other contexts. For MMC and other types, context matters, and it is critical to considering the native fire regime. Final determination of MMC forest for restoration purposes should be based on landscape context and local environment and disturbance history as evidenced in the composition, age-structure, stumps, and fire scars.

The MMC type is widely distributed across the east side (fig. 5) 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 5 is a provisional map intended to give a general picture of the distribution of MMC forest. No standard, peer-reviewed maps of MMC PVT exist for the entire region. See section 4.a for more details on source of regional information and details on vegetation in general.

**SECTION 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.

 MMC forests are hierarchically structured systems with complex interactions between spatial scales. At a meso-scale (local landscapes, e.g., watersheds and subwatersheds), patterns of forest structure and composition emerge, which are primarily maintained by interactions between environments, topography, weather, soils, geomorphology, and disturbances. But other broader and finer scale patterns also exist, and these are also influential to maintaining and even changing meso-scale vegetation patterns over space and through time.

 Understanding the relevance of landscape ecology to land management involves not only an appreciation of landscape patterns and their interactions with processes but also interactions among the elements of the observed patterns, and how patterns and interactions change over time. For example, forest ecosystem responses to disturbances or weather changes can sometimes be non-linear, 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 (Peterson 2002, Malamud et al 1998, Moritz et al. 2013). The challenge for scientists and managers seeking to develop and implement restoration and/or resilience strategies is to simplify complexity without missing key pieces—something that is easier said than done.

 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.

**3.a The concept of resilience**

 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 HRV. However, resilience is not an easily defined concept in a practical sense. Furthermore, operational metrics of resilience have received little attention (Carpenter et al. 2001). To become more operational, resilience must be defined in terms of specific system attributes and in terms of “resilience of what to what?” 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) has identified three conceptual domains of resilience: 1) engineering resilience, which focuses on recover or return time to stable equilibrium (e.g., return to particular forest structure or composition); 2) ecological resilience, which focuses on maintaining function and persistence with multiple equilibria (e.g., “historical range of variability”); and 3) socio-ecological resilience which focuses on reorganization, adaptive capacity and multi-scale interactions among the many community members, stakeholders, and responsible government organizations which 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 socio-economic system that interacts with the ecological systems.

 Alteration of an ecosystem or “ecosystem inertia” is critical to understand before we can consider if and how the system can be restored (fig. 6). Westman (1978) suggests five characteristics that depict the potential resilience of a system.

1. **Inertia**: The resistance of a system to disturbance.
2. **Elasticity**: The speed with which a system returns after disturbance.
3. **Amplitude**: A measure of how far a system can be moved from a previous state and still return.
4. **Hysteresis** The lagging of an effect behind its cause, such as delayed response of the system to a disturbance.
5. **Malleability**: The difference between the pre- and post- disturbance conditions. The greater/lesser the malleability, the lesser/greater the system’s resilience.

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 the need for employing an adaptive management approach with both research and monitoring where we can assess the results of management efforts and follow disturbances and recovery efforts over a long term. It also suggests the value of reliable ecosystem models in the measurement of both inertia and resilience. 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 non-native vegetation can also be resilient in its own way; for example, landscapes dominated by cheatgrass (*Bromus tectorum*), which is generally considered undesirable, can be resilient in the face of restoration efforts by land managers. In this example, managers have experienced significant challenges of attempting to restore sagebrush steppe ecosystems invaded by cheatgrass (D’Antonio et al. 2009, Chambers et al. 2009).

Landscapes often have inertia to change because of alterations of pattern and process from more than a century of human activity. Wallin et al. (1994) found that landscape patterns generated by past forest management or disturbance can take many decades or centuries to restore (see also Heinselman 1973). Hysteresis (in a large dose) can operate in altered landscapes to create undesirable resilience or at least protracted delay of desired response to management. For example, the widespread accumulation of Douglas-fir (*Pseudotsuga menziesii*) and, to a lesser degree, grand fir (*Abies grandis*) across many 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 complicated ecological relationships that influence the relative resiliency of an ecosystem create significant challenges in executing successful restoration. Although much of this may be beyond the control of management, it is useful to understand the limitations and constraints of restoration strategies.

**3.b 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) vegetation, climate, topography, and disturbance interact to control system behavior at multiple spatial and temporal scales; (2) Euro-American activities have altered these ecological interactions and reduced landscape resilience; and (3) increasing resilience requires management actions that restore processes and patterns across scales. Without this broad geographic and ecosystem perspective that includes the past, present, and future role of humans, 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* landscapes. We define local landscapes as variably-sized areas, typically ranging in size from one to several subwatersheds (HUC 6 or 12-digit watersheds Sieber et al. 1987, see also the National Hydrography Dataset at <http://nhd.usgs.gov/>), which reside in a single ecological subregion (*sensu* Hessburg et al. 2000b), and each exhibiting characteristic mosaics of successional stages and topography consistent with the climate and disturbance regimes of that subregion. Subwatersheds (i.e., HUC level 6) typically range in size from about 4,000 to 16,000 hectares (10,000 to 40,000 acres), but larger and smaller subwatersheds also occur. We define a regional landscape as the collection of all local landscapes that comprise an ecological subregion.

 We define (local or regional) landscape resilience (see Resilience section for more details) as the capacity of the ecosystems to absorb disturbance and climatic 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 section, we outline seven key ecological principles that are foundational to restoring eastside forests, including MMC forests. We expand on these in later sections throughout the document; however, our aim here is to highlight key ideas that motivate our thinking.

1. **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 climatic 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, the NRV, also referred to as the historical range of variation (HRV). As the climate shifted, so did the NRV. A static NRV is a common misperception and provides a misleading objective for land managers. Prolonged periods of warming or cooling, wetting or drying, or combinations of these have occurred repeatedly over time. Whenever these changes happen, it pushes 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 (e.g., live and dead legacies) encoded in the existing vegetation structure, composition and pattern variation. This is the quality of a natural system that we represent as landscape resilience. When extreme climatic changes occurred in the past, it pushed systems outside of the previous NRV into a new NRV with all of the corresponding shifts in vegetation that have been observed at various times in the biogeoclimatic record.
2. **Vegetation dynamics and fire regimes of MMC forests were/are variable.** In the historical forest, each forest type exhibited low-, mixed- or high-severity fires; the amount of each severity type varied by forest type, with the climatic regime, and with differences in the physical geography. Patterns of vegetation structure and composition and fire frequency and severity would have 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 (10-30 years), in others, fire frequency was more variable (25-75 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*) and dry mixed-conifer forests were in-filled largely by ponderosa pine and shade intolerant Douglas-fir, respectively, but in the MMC forests were in-filled by grand or white fir (*Abies concolor*) 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 were eliminated by subsequent frequent low-severity fires. During longer intervals between fires, significant fuel ladders may have developed and mixed-severity fire effect would have been typical. On wet mixed-conifer forest sites, shade-tolerant species were persistent and fire intervals were long enough to allow development of old shade-tolerant trees and larger patches of dense multi-layers forests within stands and landscapes.

1. **Time and space have profound effects on ecosystem function and pattern.** Eastside forest patterns shift over space and time, but topography, soils, and elevation constrains patterns and provides a relatively simple template for understanding and managing landscape patterns. As a first approximation, the topographic and edaphic patterns of landscapes provide a natural template for pattern modification. 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, experienced more severe fire behavior than southing 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.
2. **Ecosystems and their component parts are organized in an interactive, hierarchical arrangement.** No forest type exists in isolation: processes at regional and local landscape scales control patch dynamics, and ultimately, fine-scale patterns and processes. The overall mosaic of disturbance regimes and forest types of the larger landscape influences patterns and processes at the scale of a patch. 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 that would otherwise be expected from knowledge of their site conditions alone. In landscapes with more southing aspects and ridges, corresponding northing aspects typically also see more frequent fires and lower than typical severity. Context and scale can help guide patch-level decisions.
3. **Post settlement human activities have resulted in 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 multi-layered patches of fire-intolerant species. These changes have increased the area and frequency of large, high-severity fire patches, and extent and frequency of other biotic disturbances (e.g., bark beetle and budworm outbreaks). These changes have substantially altered the resilience mechanisms associated with the native forests; namely, mixed-severity dominated wildfire regimes that created fire-resilient vegetation patterns characterized by a variable mosaic of early, mid, and late-successional patches, whose range of pattern conditions can be estimated. 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 long enough time for coarse and fine-scale patterns and species composition to be in some degree of synchrony with their physical environments and the climatic system. Much of this synchrony has been lost through the cumulative effects of human activities on the landscape. Current and future fire regimes and landscape patterns are on new trajectories trending away from resilience. To restore this coupling between patterns and processes (wildfire, insect, pathogen, 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 climatic regime.
4. **Rare ecological events have a disproportionate effect on ecosystems. Rare, large-scale events (disturbance, climatic, biotic, geologic) can have disproportionate effects on future landscape dynamics, especially if their frequency, size, or severity are unprecedented for the climatic and environmental conditions.** Large wildfires, dramatic climatic extremes, rapid changes in plant and animal species distributions, and large insect outbreaks are examples of natural or 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 many 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, multi-layered patches. Changing this situation will require the reduction in the prevalence, complex layering, and density of these host species over a very large area to match the scale of the vulnerability to this disturbance.
5. **Resilience is dependent 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 25-35 percent of the forest had been recently burned by high-severity fire, and a significant area was in an early seral (grass, shrub, or seedling/sapling) or recently burned and recovering condition. 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 were resilient, but not all stands or patches were.
6. **Completely natural or historical landscape patterns cannot be the goal in the current and future climates and landscape conditions. However, the past (e.g., HRV) is an important guide to creating resilient forests**. Knowledge of how forests and landscapes changed in response to disturbances and climate variation in the past can provide models and clues for providing future forests and landscapes that have desired ecological patterns and process. 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 futures.

 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 to 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 a modicum of skill. Developing this characterization of a forest, though new to forest management, is not difficult and will enable a much more effective treatment strategy at the stand level where managers typically do their on-the-ground work.

**[Sidebar: 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., Taylor and Skinner 2003, Hessburg et al. 2005, Hessburg et al. 2007, North et al. 2009) have established that patterns of forest condition and fire behavior are strongly affected by topographic and physiographic features. Variability in soils also contributes to the 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 northing aspects and valleys. Wildfire conditions in summer were typically drier on southerly aspects and 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 aspects and ridgelines.** In application, southerly aspects and ridges can be managed to support fire-tolerant species in clumped and gapped distributions through the following: (1) favoring very large, large, and medium tree sizes so that they occupy at least 40-50 percent of the tree cover in the majority of these patches, and represent at least 50-60 percent of their total area; (2) stocking to support a dominance of surface fire behavior stemming from low- and mixed-severity fires, and densities that support endemic but not epidemic bark beetle populations; (3) maintaining species composition that strongly discourages the spread and intensification of root diseases, while allowing their presence; (4) maintaining stocking on south slopes and ridges by low and free thinning and similar methods and especially by prescribed burning at regular intervals. Prescribed burning and thinning activities should 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-facing aspects and valley bottoms tend to support a mix of fire-tolerant and fire-intolerant species in relatively dense, often multi-layered 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 method, 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; consequently, variations emerge that increase landscape complexity, look for them. In landscapes with more southing 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 a simple topographically-driven landscape template.

Figure 7 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. Due to fire exclusion the forests have filled in as shown by the 2010 photo.] John Marshall photo credit

End of sidebar

**SECTION 4 SCIENTIFIC FOUNDATIONS**

**4.a.1 Ecological composition, patterns, and processes prior to Euro-American settlement (< ~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). Moist mixed-conifer (MMC) forest patches frequently occur within a broader continuum of mixed conifer forest types (fig. 10) and within topographic locations juxtaposed with mixed conifer or ponderosa pine vegetation types, or grassland and shrubland patches (fig. 11) (Hessburg et al. 1999b, 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 function as “conveyor belts,” readily spreading wildfire to adjacent patches. Likewise, dry mixed-conifer patches that experience frequent surface fires often influence any adjacent moist mixed-conifer patches. However, when moist mixed-conifer 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.

Where is the MMC forest located?

 No published, peer-reviewed maps of MMC PVT exist 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 PNW Research Station and the Institute of Natural Resources at Oregon State University) for defining MMC forest distribution and abundance (fig. 5, table 3). This map is intended solely for the purpose of regional scale planning and analysis. Local classifications and maps that are spatially accurate at the stand scale will better define the distribution of MMC forest for project level planning and management.

 Other classification strategies also apparently exist (e.g., Simpson 2007) for vegetation zones that have resulted in another alternative depiction of this forest type across the region (see fig. 12 for a sample comparison). Confounding this issue, national forests often have their own local maps of PVT at some scale that are used for management. These may not correspond to the regional maps in this publication. We suggest that any future region-wide strategy for MMC forest be underpinned with a single map that uses a regionally coherent classification standard and mapping protocol. The ideal map would accurately identify the major areas that support the MMCF 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.

**4.a.2 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. The standard system of vegetation classification and mapping used by the Forest Service and other federal agencies is based on PVTs, thus we use this scheme to characterize mixed-conifer types. A PVT is the native, late-successional (or “climax”) plant community 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, PVT types 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 climax conditions were typically never realized due to myriad interacting and multi-scale disturbance processes, especially fire, that both advance and retard succession. Nevertheless, PVTs can be useful for identifying the productive potential and approximate fire regime of biophysical environments.

 A plant association is a vegetation classification unit defined on the basis of a characteristic range of species composition, diagnostic species occurrence, habitat conditions, and physiognomy (USDA Forest Service 2005). A number of plant associations are included in the MMCF type and some grade into the wetter mixed-conifer types. See the preliminary list in table 3, which may need to be refined for local applications based on expert opinion and localized forest history information.

Tree species of the MMC forest are characterized as either early- or late-successional (or seral), and as either shade-intolerant or tolerant, respectively. The most common early-seral forest tree species are western larch (*Larix occidentalis*), ponderosa pine (*Pinus ponderosa*), lodgepole pine (*Pinus contorta*), western white pine (*Pinus monticola*), trembling (quaking) aspen (*Populus tremuloides*; especially where a seasonally high water table exists), Douglas-fir (*Pseudotsuga menziesii*), 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*), white fir (*Abies concolor*; in central and southern Oregon), and subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*) 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 MMC forest would have displayed highly variable structure and composition regionally and have been among the most diverse and complex of all forest types. The MMC forest landscape included a wide range of cover type and structural class combinations (table 4) that would derive from a mixture of low- to mixed-severity fires with lesser amounts high-severity fire in the mix (e.g., see Hessburg et al. 2007, Perry et al. 2011). Recent studies in MMCF in 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 sites (Hagmann et al. 2013, Merschel 2013).

 The potential natural vegetation typically is not the same as the current or actual vegetation, which may exist in an early-, mid-, or late-successional state or in a non-native and human-altered state (e.g., agricultural patch, urban or rural development area, pasture, or road right-of-way). The difference between the actual 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 climatic 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 climatic niches may be maladapted, resulting in compromised productivity and increased vulnerability to disturbance. In some locations the future will not resemble the past, 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 Forest, using regionally standardized protocols for PVT definitions and mapping.

 While 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, Heyerdahl et al. 2001, Hessburg et al. 2007, 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 moist mixed-conifer, 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).

 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) argued “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) corroborates this finding. In dry mixed-conifer, they found that low- and mixed-severity fires dominated by surface fire effects affected most of the area of this type. In the moist mixed-conifer 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 13 and 14 illustrate the inherent heterogeneity that typically exists on these complex landscapes. Figure 13 shows the complex spatial configuration that result simply from basic topographic position in the southern end of the Umatilla National Forest. Figure 14, illustrates the more complex patchwork of landscape units that result from a combination of topography, and simplified categories of precipitation, and soils in the Wenaha watershed on theUmatilla National Forest. This is a simple example of the concept of 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.

**Sidebar**

**Mixed-conifer plant association classification**

 Plant associations of the grand fir, Douglas-fir, and Shasta red fir (*Abies magnifica*) series were classified as dry; moist; moist-wet; or wet high-elevation / warm, wet, maritime mixed-conifer forest. Associations in the subalpine fir and Shasta red fir series were included if ponderosa pine is an overstory component in the series and/or if plant association guides mentioned low-severity fires were part of the historical fire regime. We assigned plant associations to a mixed-conifer type based on their environmental setting and species composition (table 3). Associations in dry environments where ponderosa pine was common in both the overstory and understory we classify as dry mixed-conifer. Associations in intermediate climatic settings where Douglas-fir or grand fir are usually the most common understory species and are often co-dominant in the overstory with ponderosa pine, or are predicted dominant in the overstory or understory community by the primary indicator plant species, we classified as MMC forest. Associations in the relatively wet environments where shade-tolerant fire-intolerant species were common in the overstory and early-seral species were rare and often even aged were classified as wet mixed-conifer forest. Ponderosa pine was usually absent in all strata in these latter types and early-seral species were lodgepole pine, western larch and Douglas-fir. Associations that could not be classified as moist or wet because of broad variability in species composition, which we expect is related to variation in historical disturbance regimes, were classified as moist-wet. Associations in this classification exhibit structural characteristics of both moist and wet mixed-conifer forest. The wet high-elevation / warm, wet, maritime classification was given to all associations with species composition and an infrequent high-severity fire regime that was inconsistent with our definition of eastside mixed-conifer forest.

Historical fire regimes are based on descriptions in plant association guides or inferred from species composition and environment if author descriptions were not provided. Note that associations in the wet high-elevation / warm, wet, maritime classification do not fit this description. Because the disturbance regime in the moist-wet classification is broad and poorly described sites in these associations may or may not fit our intended focus of mixed-conifer forest with a historical mixed-severity fire.

Description of table columns

**National Forest / Region** - Describes the subregion of the plant association.

**Extent** – Describes the geographic distribution and approximate area of a plant association relative to other associations.

**Species composition** – All tree species recorded in each association all listed. Species are ordered based on importance within an association.

**Historical fire regimes** – Describes the historical fire regime of an association. This was described based on descriptions in plant association guides or inferred from environmental setting and species composition of each plant association.

**1** - Fire return interval is less than 25 years and fires are low-severity with respect to mature trees. Crowning rarely occurs in small patches.

**2** - Fire return interval is more variable between fires and fire free intervals may exceed 40 years. Fires are predominantly low-severity, but larger patches of high-severity fire kill large old trees on longer intervals >40 years.

**3** – Low-severity fires rarely occur and fire return intervals usually exceed 75 years. Fires are usually moderate to high-severity with respect to mature trees.

**Source** – Citation for plant association description. (Evers 2002)

Comparison of agency regional fire regime classification with classification used to characterize fire regimes of plant associations as described in plant association guides:

|  |  |
| --- | --- |
| Fire Regime Group Classification (see Appendix A)  | Corresponding class from 3 category classification |
| I | 1 |
| III | 2, 3 |
| IV | 3 |

**End of sidebar**

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 (Omernik 1987 fig. 15 and Bailey 1995), and ecological subregions (Hessburg et al. 2000a) that reside within them.

 The location of the region on the windward Pacific Coast results in a predominantly marine-type climate west of the Cascade Mountains, while east of the Cascades, the climate possesses both continental and marine characteristics (Western Regional Climate Center, [www.wrcc.dri.edu](file:///C%3A%5CDocuments%20and%20Settings%5Cpstine%5CLocal%20Settings%5CTemporary%20Internet%20Files%5CContent.Outlook%5CIE0N115T%5Cwww.wrcc.dri.edu)). In fall and winter, a low pressure system in the North Pacific brings moist and mild (i.e., pacific) westerly airflow across the region that results in a wet season beginning in mid to late October, reaching a peak in winter (January to March), then gradually decreasing later in the 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.

 The orographic effects of the deeply dissected mountain ranges in the region result in occasionally heavy precipitation on Cascade west slope, and at higher elevations, and relatively dry conditions on the east slope, and in lower elevations (fig. 16). Temperatures vary from mild maritime conditions west of the Cascade Crest to continental and temperate or Mediterranean conditions east of the Cascades.

 Plant lifeforms and community types vary predictably across these landforms and climatic gradients. The Douglas-fir/western hemlock (*Tsuga heterophylla*) habitat type dominates west of the Cascade Crest, except in the Willamette, Umpqua, and Rogue River valleys in the rain shadow of the Coast Mountains and the 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 Cascade Crest east to the west slope of the Rockies, 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 Plains. Ponderosa pine and juniper habitats make up the lower forest and woodland ecotone. Douglas-fir occupies intermediate elevations, western larch and grand fir occupy still higher elevations, and Englemann spruce and subalpine fir are found near upper tree line. Lodgepole pine inhabits subalpine environments and other low and mid-montane 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 and Great Basins (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 et al 1998, Peterson 2002). 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. 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 myriad 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. 17). 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 varied 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 dry and MMC forests. Given the generally sparse network of dendroecological and landscape reconstruction studies (especially 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 moist mixed-conifer 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 of Oregon, where the disturbance mix may have had less high-severity fire, moist mixed-conifer 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, 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 moist mixed-conifer forest would have had long intervals (>75-100+ years) between fires. It is on these sites that dense, multi-layered 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 et al. 1997, Camp 1999). 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 mal-adapted 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 moist mixed-conifer were typically low and fell within a relatively narrow range (Table 5). Of 15 estimates of historical small, medium, and large tree stand densities from different dry and moist mixed-conifer types and environments (based on seven total studies), 12 estimates fell within the range of 40 to 170 trees per ha (16 to 70 tpa) and most of these estimates probably represent the minimum because most trees from cohorts that were regenerated in 18th and 19th 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-fir, appear to fall within a relatively narrow range of low values from around 20 to 40 trees per hectare (8 to 16 tpa) (Table 5).

**4.a.3 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 Loucks 1995, Wu and David 2002). 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 18 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.

 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. 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) 5600 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 broader 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” where disturbance regimes and environment favor the development of forest containing old, fire-intolerant tree species (Camp et al. 1997). 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).

 At all spatial and temporal scales of the hierarchy, landscapes exhibit transient patch dynamics and non-equilibrium behavior, resulting in ecosystem patterns and processes that may/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 (Wu and Loucks 1995).

 Landscape patterns drive processes at local and regional landscape scales; and at either scale, no two landscapes exhibit the same patterns, either across space or over 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 19 and 20). 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 20). Patterns and spatial arrangements of combined cover type and structural stage patch sizes (syn. 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, Malamud et al. 2005, Moritz et al. 2010, Perry et al. 2011). This constraint probably appeared to be relatively stationary over multi-decadal timeframes, but varied over multi-century 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 varied in the long term, the processes and patterns they reflected also varied over time. In a warming climate, for example, 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 21). 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 22). 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.

 Changes that are small in amplitude and short-term relative to the tolerances of dominant organisms (often multi-annual to multi-decadal) will do little to reshape the envelope because of the strength of system memory (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 centuries or longer) reshape pattern envelopes by imprinting those landscapes with new disturbance and recovery regimes over large areas. For example, changes to the climate of the southwestern U.S. 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).

 Given the frequency of large and small changes in the forest ecosystem, 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 climatic 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. Multi-scale 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.

**4.a.4 Natural disturbance factors**

 In this section, we highlight key disturbance processes in the MMC forest. Disturbance regimes vary across climatic 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 details.

**4.a.4.i 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 are complete. Tree rings are one source of empirical data that can lead to multi-century 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 while many such reconstructions predate most logging, most represent conditions that occurred 30 to 60 years after fires were excluded. So our inferences are constrained by data limitations, especially when we need to apply them to particular potential vegetation types or ecoregions.

**4.a.4.ii Historical fire regimes across the region**

 In this region, several spatially extensive studies have reconstructed multi-century histories of low-severity fire regimes from fire scars in dry and MMC forests and inferred their climatic, topographic, and land-use drivers at landscape and regional scales (Everett et al. 2000, Heyerdahl et al. 2001, Hessl et al. 2004, Wright and Agee 2004, Kellogg et al. 2008). Few studies have focused specifically on MMCFs as we define them (Woodard 1977, Heyerdahl et al. 2001, Tiedemann and Woodard 2002, Perry et al. 2011, Wright and Agee 2004, Hessburg et al. 1999a, 1999b, 1999d, 2007). 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 historically 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 one hundred 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) provides fire severity patch size distributions relating to dry and moist mixed-conifer 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 (Heyerdahl et al. 2001, Gedalof et al. 2005, 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 (Heyerdahl et al. 2001, Hessburg et al. 2007). 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 in 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.

**4.a.4.iii Fire dynamics: resulting spatial patterns and tree mortality rates**

 Fire is a significant driver of the mixed-conifer forest ecosystem. The periodic influences of fire largely control the rates and patterns of succession and determine forest structure and composition. 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 moist mixed-conifer forests limits the ability of managers to effectively identify appropriate steps towards ecosystem restoration. We do know that natural fire regimes in moist mixed-conifer forest 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, Fig. 19).

* **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 10-30 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- to 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, James Johnston personal communication). Elevation and aspect influences on plant available soil moisture played important roles. Figure 23 illustrates a forest subject to a typical low-severity fire regime.
* **Mixed-severity fires** (killing 25-75 percent of the overstory) were common in MMC forest because much of the area in these productive systems was visited by wildfires on a relatively infrequent basis (i.e., about every 25-75 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. 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 24 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*], Shasta red fir, and pacific silver fir [*Abies amabilis*]) and did not appear to dominate the fire regime of MMC forests. Figure 25 illustrates a forest subject to a high-severity fire regime.

**4.a.4.iv 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 (1-50 ha [~2-123 acres]) to medium-sized (100-5,000 ha [~247 – 12,355 acres]) fire events were most numerous, and pock-marked the landscape in great abundance (Malamud et al. 1998). But although they represented 85-95 percent of fire events, they only accounted for 5-15 percent of all of the landscape area burned by historical wildfires (Malamud et al. 1998, Malamud et al. 2005; fig 26). 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 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 acres]) were much rarer, but they typically burned most of the landscape area that was subject to fire (75-95 percent).

 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 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 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 climatic events, wildfires could become very large, and fire behavior was often unrelated to the initial 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.

**4.a.5 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 non-native defoliating species and one non-native sap-sucking species of importance (table 6). The impacts 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 via girdling in a short period of time. 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 tree vigor and thus forest susceptibility. Factors such as stand density, basal area or stand density index, tree diameter, and host density are consistently identified as primary attributes positively correlated with bark beetle infestations (Furniss and Carolin 1977, Fettig et al. 2007). The effects of density are primarily mediated through three factors that affect host finding and colonization success by bark beetles: microclimate, tree spacing, and changes in host vigor.

 Defoliators consume, mine, and/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 6, Fig. 27 and 28), 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). In the 1950-1980s, large-scale efforts were implemented to control Douglas-fir tussock moth and western spruce budworm outbreaks using aerially-applied chemical and biological (bacteria and viruses) insecticides, but were thought to have little effect on population dynamics or levels of tree mortality (Torgersen et al. 2005, Torgersen 2011). A successful example of classical biological control has significantly limited the impact of larch casebearer (*Coleophora laricella*) in the region (Ryan 1997).

 While 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, but also create structure and food sources that have significant value to wildlife communities. Section 4.a.6 provides more 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 mountain pine beetle (*Dendroctonus ponderosae*)-affected lodgepole pine forests (fig. 29). 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. 2013), most studies predict increases in surface fire rates of spread and fireline intensities due to increases in fine fuel loadings and reductions in sheltering (e.g., Page and Jenkins 2007, Klutsch et al. 2011, Hicke et al. 2012). 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. Defoliation, largely the result of infestation of these two insects, often predisposes trees to subsequent mass attack by other insects, specifically bark beetles.

**4.a.6 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 readers to delve into this literature if it is of particular interest. 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: caused by 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 (<5-10 percent of patches affected) but not dominant in most pre-settlement 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, 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 in turn-of-the-century forests 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, 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-group annosum root disease** centers were relatively uncommon in pre-settlement 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 (Schmitt et al. 1984, 1991; Filip et al. 1992). Most surprising in the MMC forest is the rate of increase in S-group annosum root disease in grand and white fir. These forests contain large increases in S-group 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-group 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-group annosum is often high in Douglas-fir and grand fir forests, but mortality in ponderosa pine is uncommon. With prolonged warming, however, P-group 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-group 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 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 were particularly threatening to the long survival of their host species (Alexander and Hawksworth 1975, Parmeter 1978, Tinnin 1981). Douglas-fir dwarf mistletoe (*Arceuthobium douglasii*) and western larch dwarf mistletoe (*Arceuthobium 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, where fires appeared with moderate frequency. Mistletoes were probably most common on south 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 meters (10 to 15 feet) 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.

 Young Douglas-fir, conversely, 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, for example 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 (for example, 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 those managing for wood production (i.e., foresters who would be inclined to remove diseased trees) and those managing to maintain or enhance wildlife habitat (wildlife specialists).

 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 which 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).

**4.a.7 Fish and wildlife component of the moist mixed-conifer ecosystem**

 A variety of terrestrial wildlife species are associated with the moist mixed-conifer forests of the Pacific Northwest, including three federally listed threatened species (Northern spotted owl [*Strix occidentalis caurina*], grizzly bear [*Ursus arctos*], and lynx [*Lynx canadensis*]), three federally listed candidate species (Oregon spotted frog [*Rana pretiosa*], fisher [*Martes pennanti*], and wolverine [*Gulo gulo*]), and one recently de-listed endangered species (gray wolf [*Canis lupus*]). 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, northern spotted owls (NSO) are found in moist mixed-conifer forest 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, much of which has been altered by both logging and more recently large fires in the last 50-100 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 in wet sites through much of the Cascades in Oregon and Washington (McAllister and Leonard 1997, Cushman and Pearl 2007). More recently, populations have been documented in the Washington southeast Cascades (near Trout Lake and Conboy, Klickitat County, Washington) and Oregon south-central Cascades. These frogs are highly aquatic, occupying emergent wetlands, lakes, and marshes within forested landscapes. The historic range of fisher encompasses mesic interior forests in the Oregon and Washington Cascades, northeast Washington, and Blue Mountains (Lewis and Stinson 1998). Fisher were extirpated from most of that range as a result of fur trapping. A small population is present in the Oregon southern Cascades but at present they are largely absent from interior forests in Oregon and Washington.

We do not have the space for an exhaustive review of the substantial literature on spotted owls in the eastern Cascade Range, however we will highlight a few key aspects of NSO biology directly related to moist mixed-conifer forests. 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, Courtney et al. 2008), 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 spotted owl nesting, roosting, and foraging habitat is “a multilayered, multispecies canopy dominated by large (>76.2 centimeters [30 inches] dbh ) conifer overstory trees, and an understory of shade-tolerant conifers or hardwoods; a moderate to high (60-80 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). While spotted owls 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 historic conditions (Everett et al 1997, Lehmkuhl et al. 1994, Hessburg et al. 2005), and provide fuel characteristics and spatial patterns that are conducive to uncharacteristically high-intensity, wide-spread wildfire events (Agee 2003).

Spatial and structural heterogeneity is also important for spotted owl prey. Bushy-tailed woodrats (*Neotoma cinerea*) and northern flying squirrels (*Glacomys sabrinus*) 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 (*Strix varia*) are an important factor contributing to recent spotted owl population declines (Forsman et al. 2011). Spotted owl populations in Washington and northern Oregon declined by approximately 40-60 percent from 1989 to 2008 (Forsman et al. 2011). Many experts expect that spotted owl 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 mid-slope settings as early as the 1990s when most spotted owl activity centers were documented, resulting in relatively few historical spotted owl sites being documented in valley-bottom MMC settings that otherwise appear to be suitable spotted owl habitat (Singleton 2013). Persistence of the spotted owl population in mixed-conifer forest landscapes may be dependent on ability of the local spotted owl population to adapt to the presence of barred owls (Gutteriez et al. 2007). Substantial habitat loss due to 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 (Forsman et al. 2011, Dugger et al. 2011).

Wildlife habitat features at multiple spatial scales

 Morrison et al. (1998, p. 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 interior mesic 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. 30). **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, 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 habitat feature is therefore based on multi-scale habitat selection (Johnson 1980).

 **Broad-scale terrestrial habitat features**

**Wildlife in moist mixed-conifer forests in Oregon and Washington**

Using the wildlife habitat relationships database of Johnson and O’Neal (2001), Lehmkuhl (2005) concluded that bird and mammal communities in eastside interior mixed-conifer 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; fig. 41). 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).

 At the broadest scale, the distribution of wildlife and fish species is determined by regional to sub-continental 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 are highest in the Okanogan Highland and Siskiyou ecoregions (Swenson and Waring 2006, Hansen et al. 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. Genetic diversity, metapopulation dynamics, and population viability for some 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 within eastside mesic forests (fig. 31) 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, WHCWG 2010, Theobald et al. 2012).

 **Meso-scale terrestrial habitat features**

 Meso-scale landscape patchiness determines the arrangement of habitat resources. A critical feature of wildlife habitat in mixed-conifer landscapes in eastern Washington and Oregon is the multi-scale (landscape and stand) diversity and juxtaposition of patch types of differing composition and structure (Perry et al. 2011).

Forest patches in different stages of development typically have different habitat characteristics and provide different sorts of habitat features (Thomas 1979, Johnson and O’Neal 2001). Recently disturbed patches tend to support high plant productivity, and often exhibit a particularly high diversity of plant and animal species, including many wildlife specialist species (Swanson et al. 2011, Betts et al. 2010, fig. 32). They are also important for ungulate summer forage (Thomas 1979). Even-age young forest patches are generally less diverse, particularly for sites dominated by dense, closed-canopy conditions with little structural or understory diversity. Older forests with a diversity of tree ages and sizes (including larger trees) also support several specialist species and provide hiding cover for ungulates. Across the variety of forest structural conditions found in MMC forests, wildlife species and richness are favored by structural complexity in the form of varied tree sizes, abundant snags, and coarse woody debris (McComb 2001). Complex burning patterns have many potential impacts on wildlife depending on species' life history and population structure (Smith 2000). These processes result in a variety of forest structure and composition patterns that can support different terrestrial wildlife communities.

 The patchiness of mixed-forest landscapes might be misconstrued to mean that habitats are highly "fragmented" in the sense that habitat patches do not occupy a large fraction of the landscape or are not well connected. However, an important and often ignored distinction needs to be made between inherent, or natural, patchiness of landscapes (often characteristic of mixed-severity fire regimes) and fragmentation of habitats induced by human activity (Sallabanks et al. 1999, Bunnell 1995). One of the impacts of altered disturbance regimes, specifically wildfire, has been the establishment of relatively large, homogeneous patches of young and mid-aged forest (Hessburg et al. 2000a). This landscape simplification (the opposite of fragmentation) has favored some types of habitat over others.

 Habitat heterogeneity on pre-settlement landscape was largely a result of mixed-severity fire. 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. These mosaics were quite different than the contemporary managed landscapes that are often expressed as largely homogeneous patches of different sizes with sharply defined boundaries (e.g., see Chen et al. 1993, 1995, Halpern et al. 2005).

**Fine-scale terrestrial habitat features (structural elements)**

 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. This extremely valuable structure is specifically found in cavities (live and dead standing trees), large branches, broken-top trees, brooms, and other features that provide cover or a platform for these daily living requirements. Much of that structural diversity is associated with big trees, both living and dead (Marcot et al. 2002). Even within-stand structure is influenced by disturbance processes. Big trees and logs are often a legacy of previous stand conditions that have been retained after some disturbance event, like fire or harvest (“biological legacies;” Franklin et al. 2000). The presence of a few large trees within a stand can make a big difference. For example, snags and down logs provide denning and resting sites for American marten (*Martes americana*). Long-legged myotis bats (*Myotis volans*) roost in snags and bark crevices of old, large trees. Stout lateral epicormic branches can provide nest sites for northern goshawks (*Accipiter gentilis*) and other species (Daw and DeStefano 2001).

Cavities (in snags or living trees) are a keystone structure 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 (Parks et al. 1999, Bednarz et al. 2004). The security, thermal, and moisture characteristics found beneath and within large down logs can be particularly important for small mammals, amphibians, reptiles, and invertebrates. 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).

 Within stand structural diversity is generally much greater in stands with a variety of tree age or size classes. Multi-layer canopies contribute to unique thermal characteristics within stands and provide structural complexity important for arboreal mammals and their avian predators. Goshawks and spotted owls use multi-layered 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 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. 2014); and fruit-bearing shrubs provide important food resources for black bears (*Ursus americanus*) and other species (Lyons et al. 2003).

**4.a.8 Aquatic habitats (adapted and condensed from Bisson et al. 2003)**

 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).

**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, 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 dozer lines. Post-fire riparian erosion results in larger pulsed sediment and wood delivery to streams, and in some circumstances, channels can be re-organized, affecting aquatic habitats (Reeves et al. 1995; Benda et al., 2003; Miller et al., 2003, Minshall 2003). Water retention properties of the soils and reduced evapotranspiration from vegetation loss in the surrounding landscape affect runoff dynamics. In time, coarse sediment and wood are depleted by decay, and fluvial processes, woods gradually breaks up and is transported with rock and sediment downstream, until the system is replenished by other subsequent post-fire 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 (Burton 2005) to the natural fire regime in burned reaches, depending on connectivity of stream networks. 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 processes needed for interconnected mosaics of diverse and changing habitats and communities.

Streams draining burned areas of the Entiat Experimental Forest (EEF) in eastern Washington had peak flows 120 percent higher than pre-fire 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 pre-fire hydrologic data are essential to quantifying the post-fire response.

 **Climate change affects forest landscapes, wildfire regimes, aquatic habitats, management options, and interactions among these.** The effect of climate variability on stream hydrology (Jain and Lall, 2001; Poff et al., 2002) and related physical processes (Schumm and Hadley, 1957; Bull, 1991; Meyer et al., 1992; Pederson et al., 2001) is likely quite large. Changes in hydrology can happen rather abruptly (i.e., 10–100 years), and decadal- to multi-decadal-scale climate regime shifts can influence stream flows more than the management practices we focus most attention on (Jain and Lall, 2001). April 1 snowpack has declined in mountainous regions across the western U.S. (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 across the eastside as temperatures rise throughout the region. Changes in the timing and magnitude of precipitation are expected due to 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 in the upper elevations (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. 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 impact 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).

 Riparian “microclimate” can also depend on the ecological state of the streamside vegetation. Vegetation density can affect temperature and soil moisture gradients immediately adjacent to streams, and the magnitude of temperature increase and soil moisture decrease depends on the width of the area in which vegetation is reduced (Anderson et al. 2007). If riparian density is disturbance-mediated, then these microclimate consequences can be important for post-disturbance recovery. Climate change, fire, and overall land management (e.g., logging) practices are all “disturbances” that can have effects that are transmitted to aquatic ecosystems.

 **Fish and other aquatic biota are likely to be affected by wildfire and climate change in moist mixed-conifer forests.** Aquatic ecosystems consist of interacting species at all trophic levels. Productivity in each level can depend on input 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., Nakano et al. 1999, Baxter et al. 2004). 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, Wipfli et al. 2007, Binckley et al. 2010). 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., 2013). 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/rearing streams and that respond to disturbances such as wildfire. Less is known about how fire will affect macroinvertebrate communities that serve 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).

 Post-fire riparian conditions can have long-term effects on aquatic biota due 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 re-organized by erosion and sedimentation following the fire (Rosenberger et al., in preparation). This study ranged from lower-elevation dry forest, to elevations containing MMC and sub-alpine 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 and bull trout are likely the most temperature-sensitive species likely to be present in mixed-conifer zones. Bull trout (*Salvelinus confluentus*) 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 (Rieman et al. 2007, Isaak et al. 2010, 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/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 (*Onchorhynchus clarki*) 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 opening 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 (Figures 33 and 34; Bennett and Polivka, in preparation). 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.

**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 restoring habitat structure and the processes that support it, and life history complexity of native species, to the best practical extent (Gresswell 1999) where necessary. Restoring degraded aquatic ecosystems requires a similar perspective. To conserve or promote resiliency in aquatic (as in terrestrial) systems, land managers need 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 (Ebersole et al., 1997; Frissell et al., 1997; Gresswell, 1999; Naiman et al., 2000; Benda et al., 2003; Minshall 2003; Rieman et al., 2003). This means that management attempts to restore: (1) more natural patterns in the timing and amount of stream flows (Poff et al., 1997); (2) more natural production and delivery of coarse sediment and large wood to streams (Reeves et al., 1995; Beechie and Bolton, 1999; Meyer and Pierce, 2003); (3) riparian communities that function again as sources of organic material, shade, and stream buffering (Gregory et al., 1991); (4) streams, floodplains, 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 (Healey and Prince, 1995; Gresswell et al., 1994; Rieman and Dunham, 2000; Poole et al., 2001; Roghair et al., 2002; Dunham et al., 2003; Rieman et al., 2003). 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 (NRC, 1996; Lee et al., 1997) are all problems that merit focused 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 such as fire (Burton 2005). Lacking this outcome, 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 where the threat of large 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 (Kruse et al., 2001; Dunham et al., 2003). 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 impact of potential fires or fire suppression activities could be an important short-term strategy (Brown et al., 2001; Rieman et al., 2003).

Priority 2. Watersheds where 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 (Rieman et al., 2000; Hessburg and Agee, 2003). Existing road systems can initially facilitate understory vegetation treatments and fuel reduction, and subsequently removed, moved, or improved to re-establish 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 where 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, 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 (for example, see Miller et al. 2003, Hamlet et al. 2005). 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 MMCF ecosystems throughout the interior northwest (Rieman et al. 2003).

There will be key uncertainties to consider in the management of aquatic habitat in MMCF landscapes as well. Variable incidences of fire, variable need for fuels treatment, variable current quality of aquatic habitat, and management trade-offs 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 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 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 Camp et al. (1997), Camp (1999), Olson and Agee (2005), Agee (1988, 1994), 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), Skinner (1997), Taylor and Skinner (1998, 2003), and Wright and Agee (2008). With the exception of low gradient stream reaches (for example, ≤ 5 degrees of in-stream slope angle), it seems apparent that the fire regime of riparian zones tracked fairly well with that of the adjacent upslope. In contrast, the disturbance regime of low gradient and often fish-bearing reaches tracked with the hydrologic regime. Low gradient reaches were often depositional zones during flood events, with intact floodplains, 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 did burn through this vegetation.

Gaining a better understanding of the historical disturbance regimes and vegetation patterns of riparian zones is a fertile area for additional research. The existing datasets of the Interior Columbia River basin mid-scale assessment (Hessburg et al. 1999b, 2000a) provide extensive reconstructions of now nearly 400 subwatersheds whose riparian zones could be re-evaluated to provide further insights on successional patterns with riparian zones, the distribution of 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 support 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, 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 not 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).

 Post-fire riparian conditions can have long-term effects on aquatic biota due 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 (*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., in preparation). This study ranged from lower-elevation dry forest, to elevations containing MMC and sub-alpine forest in the Boise River basin. These observations are thus relevant to management strategies in mixed-conifer forests at higher elevations.

 Riparian “microclimate” can also depend on the ecological state of the streamside vegetation. Vegetation density can affect temperature and soil moisture gradients immediately adjacent to streams, and the magnitude of temperature increase and soil moisture decrease depends on the width of the area in which vegetation is reduced (Anderson et al. 2007). If riparian density is disturbance mediated, then these microclimate consequences can be important for post-disturbance recovery. Climate change, fire, and overall land management (e.g., logging) practices are all “disturbances” that can have effects that are transmitted to aquatic ecosystems.

**4.b Human impacts to moist mixed-conifer systems: influences of the last ~ 100-150 years**

 Humans have inhabited parts of eastern Oregon and Washington for over 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 less knowledge of in the impacts of their management of moist and wetter 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:

1. 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. 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.
2. After regeneration harvests a more homogenized forest structure emerged, where treatments drove stand structure and species composition towards a single cohort and commercially desired species. After selection cutting, stand structure moved toward multiple cohorts, due to continuous regeneration and release (some stands were entered several times), and species composition moved towards domination by shade-tolerant species, in dense multilayered arrangements.
3. A simplified vegetation mosaic was created by removal of large, fire-tolerant trees over repeated harvest entries, which typically left many smaller and few large trees. The result was ever increasing density and a surplus (in comparison with the native disturbance regimes) of young and intermediate aged forests.
4. 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*], and regenerated grand fir, white fir, Douglas-fir, and subalpine fir dominated stands).
5. An increased vulnerability to large and severe fires, insect outbreaks, and disease pandemics.
6. Fewer grasslands and shrublands.

 Management activities have also had a profound effect on the spread of non-native species, especially in the dry rangeland and forest ecosystems of eastern Oregon and Washington, which have not experienced the degree of invasion of non-native plants and animals, as in many other parts of the world, but current levels of invasion, and ongoing spread remain a significant concern. As the climate changes, conditions for continued spread of some non-native (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 the coming years, unless significant steps are taken to slow the advance of alien plants and animals.

**4.b.1 Timber harvest and associated activities**

 Timber harvest and road development over the last 100 years have had a significant impact on MMC forests in eastern Oregon and Washington (fig. 35). 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 predominantly even-aged where regeneration harvesting was practiced and uneven-aged where selection cutting was more common, with 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, Larsen and Churchill 2008, 2012, Hessburg et al. 2003, 2005, Merschel 2012).

 Historical replanting at many sites used maladapted seed stock from locations sometimes many hundreds of kilometers away, 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 primitive understanding of the relationship of seed provenance 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, Huff et al. 1995, Hann et al. 1997, Hessburg et al. 1999b). 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 entire region contain large (>~53 centimeter [21 inch] dbh) young trees with species compositions and stand structures that may succumb to bark beetles, or fire, and may profit directly from density and species management.

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

**4.b.2 Fire exclusion and natural fire dynamics**

Fire frequency in the region declined due to 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, Hessburg et al. 1999a, 1999b, 2000a, 2005, 2007; Heyerdahl et al. 2001, 2008, Hessl et al. 2004, Wright and Agee 2004, Haugo et al. 2010), resulting in further changes to natural forest composition and structure. In the absence of fire, forest density increased, the role of shade-tolerant tree species expanded, and fuel loads have increased (Everett et al. 1994, Huff et al. 1995, Lehmkuhl et al. 1994, Hessburg et al. 2000a, 2003, 2005, Perry et al. 2011, Baker 2012, Merschel 2012). Not coincidentally, human population densities have increased rapidly in these same settings. Many people prefer to live at the lower forest ecotone, and 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 (Perry et al. 2011, Moritz 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. 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 is 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).

**4.b.3 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 in a weakened condition. Large cattle herds also grazed adjacent dry forests and nearby grassy riparian zones. Since cattle require lots 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 1–4 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 removed by livestock (Roath and Krueger, 1982).

 Because sheep require less water and can graze more successfully on rangelands than cattle, introductions of sheep boomed in the mid- to late 1880s with the decline in cattle numbers. Eventually sheep numbers outstripped cattle and often violent conflicts 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 8000–10,000 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 that 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 (Wissmar et al. 1994a,b; Langston 1995; Hann et al. 1997).

 Grazing permits and fees were required on national forest lands beginning in 1906, although grazing intensity increased until the 1920s (Wilkinson 1992). In the 1930s, 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 dropped. 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 non-native species introductions, some changes in native plant community structure and productivity are likely permanent (Lehmkuhl et al. 2014). 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). While public land managers have made steady strides with reducing cattle impacts to riparian vegetation and sediment load in small streams and creek bottoms, this is likely an area for continuous improvement.

**4.b.4 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, mesocarnivores (medium-sized predators) and diseases, and ultimately, cause local extirpation of some native species (Pickett et al. 2001, McKinny 2002, Hansen et al. 2005). The presence of homes in the WUI also strongly impacts 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 fire-fighting costs.

 Rural development and land use change is expected to continue to influence the distribution, configuration, and abundance of the forested landscape in the Pacific Northwest. Theobald et al. (2011) evaluated change in abundance and connectivity of “high-quality” forest patches due to land use change in the western U.S. through 2000, and projected land use change to 2030. They estimated that land uses associated with residential development, roads, and highway traffic have caused roughly a 4.5 percent loss in area (20,000 km2 [7722 mi2]) of large, unmanaged forested patches, and continued expansion of residential land will likely reduce forested area by another 1.2 percent by 2030. The projected loss/change of forest types for 2000-2030 was particularly high in Washington and Oregon (fig. 36). 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 Mountain (Theobald et al. 2011). This 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.

**4.b.5 Climate change**

 Climate change is affecting MMC forests in eastern Oregon and Washington through changes that include warmer temperatures (Mote and Salath 2010) and snowpack decline (Mote 2003b, Mote et al. 2005). Over the next 100 years, because CO2 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 (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 increases forest mortality in the eastern Cascades (Choat et al. 2012, Williams et al. 2012).

 In addition to the general changes in temperature and precipitation predicted by various regional models there is the likelihood that change at the local scale could be quite variable due to high variability in physiographic environments where MMC forests are found. 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. This fine-scale variation becomes another component of future uncertainty, especially at fine and meso-scales.

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, and population viability. Impacts may be particularly severe for currently rare or endemic species with restricted distributions (citation here). Anticipated changes to wildlife include (1) the susceptibility of high-elevation habitats and species dependent on snowpack (e.g., wolverine for the Blue Mountains), (2) impacts on wetlands and associated species, especially those sensitive to water temperature (e.g., tailed frog), and (3) phenological mismatches for migratory birds and other species (citation here). Changes are expected to happen more quickly than species’ potential to adapt (citation here). 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, Nunez 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 1945-1976 and in a warm phase from 1977-1998 (Waring et al. 2011, Mantua et al. 1997, Mantua and Hare 2002, Mote et al. 2003). Minimum temperatures have increased most rapidly in the semi-arid Columbia and Great Basins, and in semi-arid portions of the Okanagan Highlands, but have changed relatively little west of the Cascade Crest and in the Blue Mountains (Fig. 37). Mean average temperature has been observed to have increased by 0.8° C (1.50 °F) since 1900 (Mote and Salathe 2010). Climate 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 1970-1999.

 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 13-38 percent since 1900 but has shown substantial inter-annual and inter-decadal 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 (1-2 percent in 2030-2059, and 2-4 percent in 2070-2099; 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 with a conspicuously dry and warm summer season and most precipitation coming in winter months. 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 U.S. is a direct result of these changes in climate (Westerling et al. 2006).

 April 1 snowpack has declined in mountainous regions across the western U.S. (Mote 2003b, Mote et al. 2003). Changes are largely attributed to elevated winter and spring temperatures (Hamlet et al. 2005, Stewart et al. 2005). Snowpack decline is expected to continue across the eastside as temperatures rise throughout the region. Changes in the timing and magnitude of precipitation are expected due to interactions between rising air temperatures, snowfall, and rainfall across complex local terrain. Warming is expected to result in more precipitation falling as rain than snow, and earlier snowmelt timing in the upper elevations (e.g., Hamlet et al. 2005).

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 1950-75 and 1995-2005 based on climate effects on simulated leaf area index (LAI). While 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 American 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. 38). Furthermore, they found that regions exhibiting reduced competitiveness under climate change scenarios had experienced higher rates of disturbance by insects, fire, and other factors.

 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). We acknowledge that limitations to the use of climate envelop approaches for modeling potential species distribution changes are well known. 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 (fig. 39). 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 forecast to decline substantially: 41 percent of Rocky Mountain Montane Conifer, which currently occurs in the Okanogan Highlands and the Blue Mountains, is forecast to remain in 2060. Importantly, by 2090 only 12-57 percent of the current area is forecast 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 ecophysiologial 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 CO2 concentrations through increased water use efficiency. However, forest growth response to rising CO2 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 (for example, Scolytid 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 continue to 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 (Polivka et al. *in preparation*). 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.

**4.b.6 Changes in vegetation**

 It is well documented that the composition of forests has changed since Euro-American settlement (Hessburg and Agee 2003). Forests are now typically several times denser, in most locations, than they would have been under native fire regimes (Camp 1999, Perry et al. 2004, Merschel 2012, Hagmann et al. 2013) (table 2). However, one recent study (Baker 2012) has indicated that these forests were “generally dense” (100 tpa [275 tph]) in the late 1800s and suggests that the amount of change in the structure of these forests and the need for density reduction has been overstated. It is unclear how the density estimates from the Baker study apply to the MMC forests, which are a subset of the ponderosa, dry mixed-conifer and lodgepole pine stands that he sampled. In the Baker study, less than 1 percent of the trees were identified as white or grand fir, suggesting that either his records did not sample much MMC forest or that there 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 densities for the same areas. Munger (1917) reported a density of 55 large tpa for ponderosa pine forest, and 61 large tpa 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.

 While total stand densities have increased several fold, the densities of large trees (> 50 centimeters [~20 inches]) has 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 5). The decline has been observed in both Washington and Oregon 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 density of large trees (Ohlson and Schellhaas 1999).

 Heyerdahl has shown that fire occurrence in several pine and mixed-conifer forest patches in central Oregon strongly declines after 1880, and grand fir establishment starts to increase in the 1880s and 1890s (Heyerdahl et al. 2001 and unpublished data, Merschel 2012). If we assume that the pre-settlement density of trees in mixed-conifer patches was 20 to 40 trees per hectare (50 to 100 tpa), then current densities for mixed-conifer forests may be 2 to 6 times the historical densities of some patches. For example, several recent studies have estimated current mixed conifer densities at ~80 to over 120 trees per hectare (200 to over 300 tpa; Camp 1999, Perry et al. 2004, Merschel 2012). Under the recent disturbance regime of fire suppression and logging old trees, MMC forests often consist of scattered remnant of medium to large-sized shade-intolerant dominant trees, with medium-sized shade-tolerant co-dominant trees, often several lower 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 (fig. 17).

 Seed availability, either absence of seeds of a species or presence of nearby seed sources (e.g., a kind of “mass effects” in which a high rate of propagule input maintains species on sites that are not well-suited for reproduction or long-term survival; Shmida and Wilson 1985), which can have a strong influence on community development, 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. 1999b, 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 side and east side 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 eastside 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 20-35 percent (Hessburg et al. 1999b, 2000a). Current landscapes have much more of this type of fire-infrequent old-growth.

 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, 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 is quite different at the bottom of a drainage where soil moisture and shade enables mixed species of denser, multi-layered patches compared with patches on a mid, south-facing slope that 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 like westside old growth. 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 for long periods (like they do on the west side). The transience of old forest conditions in interior MMC forest is a consequence of the unique insect, disease, and fire disturbance processes found on these drier landscapes. The dense, multi-layered 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 micro-climate are suitable (e.g., sufficient soil moisture). By contrast, the sparser, single layered old forests, dominated by large pines, are capable of persisting for very long periods of time (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 large, old, fire-resistant ponderosa pine, Douglas-fir, western larch, or combinations of these species 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) and mixed-severity fire 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 include longer return intervals (>50 years) and old growth stands would have larger components of fir in the understories and sometimes in the overstories as scattered old shade-tolerant trees.

 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 various kinds of mixed-conifer forests, have been heavily impacted by fire exclusion due to their relatively short life expectancy (stands begin to deteriorate at 55-60 years of age and are pathologically old and decadent at 90-110 years; Bartos 2000). Wildfires normally revitalized aspen clones, with some patches showing 10,000-20,000 stems per hectare (24,710-49,421 stems per acre) in early life stages after fires. Ungulate browsing, both wild and domestic, and a host of stem cankers, foliar diseases, and defoliators naturally thin aspen clones to a few hundred stems per hectare after several decades.

**4.b.7 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 6), 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 (e.g., Douglas-fir tussock moth and western spruce budworm). Wickman (1992) noted that historically the impacts of insects on mixed-conifer forests were typically of shorter duration and lower severity than 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), and while 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 (Lehmkuhl et al. 1994, Hessburg et al. 1994). For a recent review of the expected effects of climatic change on forest pathogens, we refer the reader to Sturrock et al. (2011) and Dale et al. (2001).

**4.b.8 Effects of natural disturbance, land use, and climate change on wildlife and fish populations and habitats**

 There have been significant changes to fish and wildlife populations in this region during the last 150 years. The array of human disturbances and resulting changes in vegetation, discussed above, has had a significant impact on fish and wildlife habitats across this entire area. Additionally, the nearly complete removal of grizzly bear and gray wolf from the region by the 1920s and intensive predator control on coyotes (*Canis latrans*) and cougars (*Puma concolor*) through the 1960s, would have had significant effects of allowing elk and deer populations to flourish in higher numbers than prior to Euro-American settlement.

 Regional landscape assessments (Everett et al. 1994, USDA 1996, Lehmkuhl et al. 1994, Hann et al. 1997, Hessburg et al. 1999b, 2000a) over the past 20 years have documented the profound effects of historical human use and forest management on the mixed-conifer, and other, wildlife habitats of eastern Oregon and Washington. The area and condition of grasslands, shrublands, and old forest multi-story and late-seral single-story forests, and the forest backbone of large residual trees have declined markedly (Hessburg and Agee 2003, and Hessburg et al. 2005) (fig. 40). 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.

 Those changes have been tracked by similar changes in wildlife “source” habitats for species of conservation concern. Habitat for species associated with low-elevation old forest, which includes single-story ponderosa pine forest and single- and multi-story mixed-conifer forest, has declined in most of eastern Oregon and Washington (table 8) (Wisdom et al. 2000). The habitat trend for species associated with old forest (i.e., forests with multi-story stands and large, old trees), which spans the elevation gradient from low to high, varies across the region from markedly declining in the northeastern Cascades 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.

 The habitat trends reported by Wisdom et al. (2000) 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 is neutral in two of the four provinces; but, within those neutral provinces this habitat decreased in 47 percent of the subwatersheds in the Blue Mountains and 37 percent of the subwatersheds in the southeastern Cascades.

 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 northern spotted owl recovery are complex. Habitat conditions for NSO in this region are significantly different today than 150 years ago. While 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?

 A recent NSO recovery plan (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 recent Recovery Plan (USFWS 2011) describes the difficulties in defining conservation objectives for habitat of northern spotted owls in these drier forests of the eastside:

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 complimentary – 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.

 In general, we recommend that dynamic, disturbance-prone forests of the eastern Cascades, California Cascades and Klamath Provinces 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 forests through decades of selective cutting and sanitation treatments (USFWS 2011). Managers are well 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. NSO 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 (Forsman et al. 2011, Dugger 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.

 Past forest management practices often focused on reducing pathogens like mistletoe and root-rot 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 and provide nesting and roosting opportunities for a variety of species, including spotted owls and their prey (Parks et al. 1999, Sovern et al. 2011, Watson 2001, fig. 41).

 Rural development, roads and forest management practices also influence connectivity for fish and wildlife. These broad-scale linkage patterns are determined by major landform features and associated ecotonal boundaries, in combination with human development patterns (fig. 31). For animals that make long distance dispersal movements (including some large carnivore species), the absence of barriers to movement is often more important than the physical connectedness of specific habitat conditions (e.g., Shirk et al. 2010). Major natural barriers for species associated with moist mixed-conifer forests in the Pacific Northwest include the shrub-steppe landscapes of central Washington and Oregon, and dramatic landforms like the Columbia Gorge, Lake Chelan, Okanogan Valley (U.S.), and Hells Canyon (Singleton et al. 2002, WHCWG 2010). Major anthropogenic barriers include high volume highways (for example, Interstate-90, Interstate-84, portions of US 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 Okanagan Valley and portions of the Highway 3 corridor between Osoyoos to Castelgar (Singleton et al. 2002, WHCWG 2010). These linkage patterns are particularly important for species associated with moist mixed-conifer forests in Washington because populations in Washington are often southern extensions of populations centered in Canada (Singleton et al. 2002, WHCWG 2010).

Climate change and fish and wildlife

 Climate change has the potential to strongly influence wildlife habitats. At the broadest level, changes in the area and location of major habitat types influence wildlife. The habitat types 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 of the USFS sensitive species are associated with subalpine habitats including wolverine, fisher, grizzly bear, lynx, 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 U.S. under projected future climates. Based on a downscaled ensemble model, they projected 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 which 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. These changes will have significant ramifications for the entire biological community (Walther et al. 2002, Rosenzweig at al. 2008).

 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, Nunez 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 northwest U.S.

 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 due to change 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 Agee2003, Hessburg et al. 2000a, 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 impact 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. Cutthroat (*Oncorhynchus clarkia*) and bull trout (*Salvelinus confluentus*) are likely 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 (Rieman et al. 2007, Isaak et al. 2010, Wenger et al. 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 (figures 33 and 34; Bennett and Polivka, in preparation). 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.

**4.c Current socioeconomic context**

When this science synthesis was initiated the decision was made to focus on the biophysical science associated with managing moist-mixed conifer 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.[[2]](#footnote-2) In addition, some concepts of resilience include interactions between social and ecological subsystems (see section 3a). 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. Two important issues – collaboration and the social acceptability of forest restoration treatments – are addressed in Sections 6a and 6b. The key issues of focus are wood products infrastructure and associated business capacity, biomass utilization, and recreation. Also highlighted are some of the tribal concerns associated with forest restoration in the region.

 Literature addressing 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, published literature); and much of it consists of assessment work rather than scholarly research per se. What follows addresses broader socioeconomic considerations associated with forest restoration in the region, along with germane management implications.

 Oregon and Washington east of the Cascades are dominated by rural counties where natural resources have historically played an important role in contributing to local economies through agriculture, ranching, and 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). The literature on community and socioecological resilience (e.g., Berkes and Ross 2013, Davidson 2010, Magis 2010, Walker and Salt 2006) suggests that if communities have access to, and can take advantage of, job opportunities associated with national forest management they can become more resilient to social, economic, and ecological change. This occurs through (1) diversifying their employment base; (2) innovations to invest in new business and employment opportunities; and (3) enhancing future job opportunities.

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. The economic benefits associated with forest restoration are not limited to local job retention and creation, however. Other economic benefits resulting from fire hazard reduction through 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 on federal lands 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). Restoration of moist mixed-conifer forests can potentially contribute to all of these goals. 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).

Wood processing infrastructure and business capacity for forest restoration

 Decreased timber harvesting 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 while the greatest need for hazardous fuels reduction is on the east side (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 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 3 closed sawmills, 2 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 employ 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 – thereby enhancing the economic resilience of local businesses – 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 work force (OFRI 2012, Swan et al. 2012).

With regard to production from moist mixed-conifer 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 sub-alpine 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 species. Size preferences typically range from 12” to 16” for large logs (though some mills take up to 22” logs), and 4”-6” to 10”-12” 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 non-federal 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 non-federal (almost all private) lands. Moreover, the nature of the supply from Forest Service lands has shifted significantly since the 1980s from predominantly sawlogs greater than 12” in diameter, to mostly non-saw logs (biomass) (Swan et al. 2012).

To significantly increase forest restoration treatments in east side 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 dependable supply of raw material over a ten-year time horizon at a minimum (OFRI 2012). In eastern Oregon, assessments have found that private lands will not be able to support pre-economic recession harvest levels in a sustainable way, implying that increased wood supplies will need to come largely from federal lands. Harvests from federal lands in eastern Oregon are currently estimated at 7 percent of annual growth (OFRI 2012). State of Oregon (2012) provides 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 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 has been growing in eastern Oregon over the past decade, however (e.g., businesses that carry out activities in support of timber production, firefighting, reforestation) (Davis et al. 2010). Retaining and increasing the local business capacity (infrastructure, workforce, equipment, etc.) 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 100 miles in eastern Oregon, increasing log haul distances and associated transportation costs (Swan et al. 2012). 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). This is because reduced competition for logs – which drives down stumpage prices – and higher transportation costs to get logs to more distant mills 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 further away, and that there was a threshold distance for this effect in Oregon and Washington of 40 minutes.

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 increase proximity 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 200 miles 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 5” to 10” (40 percent) and 10” to 20” (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 lands to achieve this goal. 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 (Keegan et al. 2006). In order 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). Again, a reliable supply of wood is needed to justify such investments. 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 supply guarantee. 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. The program aims to not only encourage national forests to commit to carrying out restoration projects, 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 issue.

Biomass utilization

 The development of 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 co-generation 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).

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, and 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 in order 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 was the single greatest cost associated with biomass utilization, and that decreasing the travel distance between markets and harvest sites was the only strategy that offset 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 shares 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 by-product 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). Nevertheless, the abundance of hydropower in the Pacific Northwest limits market opportunities for biomass electricity, and the supply of biomass for manufacturing value-added products is greater than market demand (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 for heating (Swan et al. 2012). Key to increasing biomass utilization east of the Cascades is stimulating market demand and opportunities.

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). More 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), on-site 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.

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 moist-mixed conifer 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, though 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). A study from Oregon’s Mount Jefferson Wilderness found that the B&B Fires of 2003 did not dramatically affect recreation there (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 effects of crown fires on recreation (Hesseln et al. 2004). However, 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 impacts. It is important to note, however, that some wildfires – for example, fires that take place in areas dominated by shrubs – may have positive effects on recreation by creating more open forest conditions (Kline 2004). Because 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), the best way forward for identifying 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). Such an approach will also make it easier to identify how different types of recreation activities may be affected by restoration treatments.

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 tribal members in forest restoration; management to address the potential impacts of climate change on forest resources that are important to tribes; and incorporating tribal members and their traditional ecological knowledge into forest restoration decision making and implementation. These concerns are addressed here, in turn.

 The 2004 Tribal Forest Protection Act was passed in order 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 tribal members in fuels reduction and post-fire rehabilitation, only a handful of projects have been implemented on national forest lands to date. 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 of the TPFA and encourage its use (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 needed 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 (Lynn et al. 2011). In eastern Oregon and Washington, climate change impacts on forests that may affect tribes are associated with 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, minerals) under climate change (Voggesser et al. 2013). As a result, climate impacts on forests may threaten tribal subsistence, culture, economies, and sovereignty. The traditional ecological knowledge held by tribal members can play an important role in helping federal forest managers identify species having social, cultural, and economic importance to tribes in the moist mixed-conifer 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 towards more 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 so are reviewed in Charnley et al. (2007), Donoghue et al. (2010), Vinyeta and Lynn (2013), and Voggesser et al. (2013).

Again, this brief overview of socioeconomic issues associated with forest restoration in eastern Oregon and Washington is not intended to be all-inclusive. Rather, it aims to touch on some of the key socioeconomic issues managers may face in making decisions about the restoration of moist mixed-conifer forests. It 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.

**4.d 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 (over 200 citations) provided.

Our key scientific findings and concepts are:

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 humandisturbances, 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. Patterns at each of the three scales are not likely to be stationary in time or space. Any given landscape today is on a trajectory to a future condition that will be a result of these combined forces.
* For landscape planning and management purposes**, regional landscapes can be viewed as a multi-level hierarchy of mosaics**; 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 divide, the most significant effect is caused by the high divide, which creates a rain shadow effect. **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 can be 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 (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 re-establish ponderosa pine or larch. The presence of abundant seed from shade-tolerant species 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 severity.
* The pattern and processes of forest ecosystems in a landscape is a function of the range of variability in disturbances, climate, and species movements that has occurred over a long period of time. **The historical range of variability that created pre-Euro-American forest landscapes has been changing and returning to it is no longer feasible or practical.** The range of variability in these processes in the future will determine how the current pattern and process of ecosystems and species will develop. Managers can influence the future range of variability to achieve desired future ecosystem conditions for a landscape. The historical range of variation 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 lifeforms 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 variation, but the degree of difference will vary with topography and ecoregion.

Disturbances

* Today, **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, did the bulk of structuring and composing the historical landscape**, providing sustainable patterns of terrestrial and aquatic habitats, and supporting processes. Several forest insects (e.g., western spruce budworm) and pathogens (e.g., root diseases) are notable disturbances in this region.
* Historical fire regimes in dry and MMC forests varied across a broad range of spatial and temporal scales in response to local and regional variation in climate and weather, topography, soils, and fuels. **The disturbance ecology and resulting vegetation of the mixed-conifer type is influenced by the dominant disturbance regimes of the local landscape**. Where dry mixed-conifer forests are a dominant feature within a landscape, the moist mixed-conifer forest often is influenced by more frequent fire. The converse is also true.
* **Small and medium sized fires (<1000 ha in size [2471 ac]) were most numerous**, comprising 85-95 percent of the fires, but representing no more than 5-15 percent of the total area burned. **Larger fires (>2000 ha in size [4942 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 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, past silvicultural practices, livestock grazing, and introductions of alien plant species** have significantly influenced the structure and composition of many forests.
* Dry and **MMC forests represent a broad range of potential vegetation types** and include thegrand fir, white fire 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 red cedar [*Thuja plicata*], 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 multi-cohort 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.
* **Medium (~41 to 64 centimeters [16 inches to 24.9 inches]) and large (> ~64 centimeters [25 inches]) fire- and drought-tolerant trees 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 historic range of variability, despite their ecological importance.
* **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 for quite some 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 as a surrogate for 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. This has occurred 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

* **Animals use habitats and sense habitat patterns across a range of scales: “habitats within habitats.”** At the **finest scale,** animals use specific habitat features associated with specific forest structure attributes (e.g., cavities for nesting, or specific food items). At the **meso-scale,** the arrangement of microhabitats is important to meet their life-history requirements (e.g., the right combination of food availability and security from predators). At the broadest scale**,** connectivity of appropriate habitats facilitates finding mates, dispersal to new areas, prevention of genetic isolation, and meta-population dynamics.
* **Dry and MMC forests in the region may serve an important role in enhancing connectivity for many wildlife species, particularly larger mammals, between the Cascade and Rocky Mountains.** Rocky Mountain and Cascade subalpine systems are forecast to decline most rapidly in response to a warming climate. Several USFS sensitive species are associated with subalpine habitats, including wolverine, grizzly bear, lynx, and American marten.
* The **lower forest ecotone across the region is especially challenging for management. This zone is high in wildlife species diversity and habitat heterogeneity, but is experiencing rapid exurban development, increased fuel loads, and increased risk of fire**. Future climate change is expected to exacerbate these tensions. In landscape and forest planning, attention should be paid to the dry forests of the lower forest ecotone and interactions with the neighboring moist forests.
* **Trees acquire distinctive physiological, pathological, and structural features as they age and are influenced by abiotic and biotic agents** (e.g., wind, stand development processes, weather events, insect attacks, and chronic forest diseases, like stem decay, dwarf mistletoe infection and root disease). These features of old trees (often including their larger size) make them **structural cornerstones in forests, contributing to ecosystem services, such as wildlife habitat**, resistance to fire and drought, and genetic reservoirs. They require centuries to replace and perform a great service as live and dead trees.
* The endangered **northern spotted owl and** **several other species require the elements of stand structural complexity typical of mature and old forests that are currently limited in some locations, and at risk in most areas where they currently exist**. Densification of forests on portions of these landscapes in recent decades, due to fire exclusion, has created some new suitable habitat for owls. However, many of these forest stands are transient and will not likely persist, due to the increasing likelihood that fire or insect infestation will enter these stands in the near future. These disturbances are projected to increase under climate change. Retention and recruitment of NSO habitat remains important despite NSO population declines that appear to be a consequence of competitive interactions with barred owls. Substantial reductions in available NSO habitat would be likely to exacerbate negative effects of competitive interactions with barred owls and limit opportunities for the NSO population to adapt to the presence of barred owls.

* M**any early-seral forest dependent wildlife species in the region use habitat that results from periodic low to mixed-severity disturbance**. Some wildlife species are adapted to use of early-seral forests, characterized by young trees and a significant shrub component. Most landscapes currently have an over-abundance of small to mid-size trees (~25 – 64 centimeters [10 –25 inches]) at the expense of the early-seral forest/shrubland and the older forests dominated by a few large trees. A combination of natural disturbance, prescribed disturbance, and silvicultural manipulations can be used to restore an appropriate range of seral stages across a broad range of patch sizes.
* **Native fish and aquatic ecosystems would be favored by restoring the natural fire regimes of dry and MMC forests.** To do this will require the re-coupling of vegetation patterns and patch sizes of seral stages and fuelbeds that support these regimes, which would recouple physical processes at scales and periodicities that generally support the species of interest. However, because a number of native fish species, life histories, and phenotypes are now listed as threatened, endangered or sensitive, it will be necessary to mitigate some of the effects of this re-coupling as work proceeds.

Management applications in the field

* “Ecological restoration” has become a principle objective that drives current management on national forest system lands. 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 socio-ecological interactions (Hobbs et al. 2011). In using this term restoration, our intention 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 socio-ecological resilience focuses on the adaptive capacity of species, ecosystems, and institutions rather than restoring or recovering a condition. In this paper, 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 still may be similar to their HRV. However, in others **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 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).

**SECTION 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. Decision makers also must weigh trade-offs 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 the best 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 research to work alongside managers in the conduct of management and seek to learn from 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.

We draw the reader’s attention to the first subsection, 5.a, immediately below. Here we synthesize the principle scientific findings that have been gleaned from the body of scientific literature (summarized in Section 4 above) as it pertains to management of MMC forests. These constitute the “take home” messages that are intended to assist land managers.

**5.a Key concepts for management**

* **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.
* **Manage for habitats of all native species.** When managing for species at risk (e.g., northern spotted owl [*Strix occidentalis caurina*]), there is a natural tendency to focus on developing and maintaining habitats for these species, to the detriment of others. Instead, manage regional patterns of vegetation and habitats with all species in mind. This does not mean that managers must cultivate every habitat type on every landscape. Applying the HRV and FRV reference conditions to local landscapes should provide managers with insights about the typical ranges of habitat conditions and types that would normally be supported by the biogeoclimatic settings and the dominant disturbance regimes.
* **Manage regional landscapes for well-connected patterns of habitats for all native species.** Regional goals for landscape habitat management are 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 (for example 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.
* To the extent possible**, local landscape patterns should reflect the overlap in the HRV and the FRV ranges**. If landscapes reflect these overlapping envelopes of an ecoregion, 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.
* **When restoring within-patch heterogeneity, no two patches should exhibit the same tree clump and gap size distributions.** If they do, patch heterogeneity in local landscapes will have been simplified. Vary tree clump and gap sizes. 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.
	+ **Focus significant restorative management on second-growth 60 to 100 year old forests.** Emphasize 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.
	+ **Restructure 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 change) and fuel accumulation. Wildfires and insect outbreaks are resetting conditions on the landscape, spatial allocation is driven by fire weather and fuels, changes are occurring suddenly, and the window of treatment opportunity for restoration is growing shorter. Identify patches in geographic locations that because of soils, aspect, elevation, and site climate are not likely to sustain these dense, drought and disturbance intolerant conditions. Re-establish the inherent landscape heterogeneity, using topography as the underlying template. Where early-seral species would naturally dominate (ponderosa pine [*Pinus ponderosa*], western larch [*Larix occidentalis*], and western white pine [*Pinus monticola*]) and natural regeneration is improbable (i.e., grand fir [*Abies grandis*], white fir [*Abies concolor*], and Douglas-fir [*Pseudotsuga menziesii*] will likely dominate for a long time), regenerate them, as is appropriate to the topo-edaphic conditions.
* **To the extent practicable, re-establish the inherent resilience capacities of ecological systems.** This is done by restoring ecological processes as the foundation of ecosystem resilience. A keystone priority is to re-establish the central role of the natural or inherent fire regime as the source of re-initiation, 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.
* **Expect surprises and maintain a broad range of management options. Expect rare events; they have always influenced the majority of the landscape area.** 21st 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 we should learn 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.
* **Provide a full complement of vegetation cover types and successional stages within mixed-conifer landscapes.** To the extent practicable, the patterns and variability of cover types and successional stages should be consistent with the inherent fire regimes. 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. Disturbance regimes and habitat conditions of neighboring types are interconnected and inter-related, and are best addressed by working together with adjacent landowners.
* **Maintain 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, re-grow them. This is one key to restoring climate and wildfire tolerance and re-establishing many core wildlife habitat values. Favor ponderosa pine in the dry mixed-conifer forests, and western larch, western white pine, sugar pine (*Pinus lambertiana*), and Douglas-fir in moist mixed-conifer, consistent with local conditions.
* **Maintain a backbone of large standing dead and down trees, and live deformed trees** (those with broken tops and limbs, cavities, structure for nesting and denning sites) to provide structure for wildlife use (Lehmkuhl et al. 2003). As a result of past timber harvest and “sanitation” treatments (Bull et al. 1997, Rose et al. 2001, Hessburg et al. 2005), these fine-scale structures are in short supply in most areas. Typically it takes years to decades for a once healthy tree to develop the proper characteristics of such a “wildlife” tree. Manage these structures (i.e., remove if necessary) as needed in developed recreation areas, such as campgrounds and trailheads, where they could pose a threat to human safety. Information for both snag and down wood management in green and dead stands can be found in the Decayed Wood Advisor (DecAid) ([www.fs.fed.us/r6/nr/wildlife/decaid/](http://www.fs.fed.us/r6/nr/wildlife/decaid/)).
* **Retain existing old forest refugia** (*sensu* Camp et al. 1997, Olson and Agee 2005) **where possible in suitable locations.** Dense, multi-layered 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) and routinely avoid the most damaging effects of wildfires. Likewise, focus restoration efforts on developing future old forest patches in these areas, especially where old forest exists in short supply.
* **Reduce density and layering in the surrounding neighborhood**, **and favor fire-tolerant species to help reduce the risk of loss of multi-layered old forest patches to high-severity fires** (Camp et al. 1997, Ager et al. 2007, Gaines et al. 2010). Loss of some existing complex structural patches is an inevitablebut reasonable trade-off for maintaining old forests in fire-prone environments.
* **Restore the natural variability of wildfire regimes as a key to restoring the physical processes that create 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.
* **Encourage 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 floodplain depositional areas whose riparian vegetation is dominated by hardwood tree and shrub vegetation, while the riparian vegetation of higher gradient streams is remarkably similar to the upslope vegetation.
* **Focus 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.** 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 humans have been learned and been convinced through structured trial and experience. The new Forest Service Planning Rule offers yet another opportunity to implement adaptive management. Flexible management approaches employing evidence-based, scientifically-credible methods promote learning.
* **Manage forests in the context of current and potential climate change.** Rates of climate change in the past century and projected for the coming 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 forest mortality. Such changes due to climate are likely to expand to larger portions of the region in coming decades. Identify rates of change in climate and ecological response over the past century and those projected for future decades and set management goals and treatments accordingly.

**[Sidebar: Example of landscape management – Okanogan-Wenatchee landscape evaluation decision support tool**

In eastern Washington, scientists at the PNW Research Station-Wenatchee and on the Okanogan and Wenatchee National Forests 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 these national forests 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, Forest-wide restoration strategy (USDA-FS 2012b). Under the strategy, the objectives of landscape evaluations are to:

1. transparently display how projects move landscapes towards resilience to drought, wildfire, and climate ;
2. define and spatially allocate desired ecological outcomes (e.g., adequate habitat networks for listed and focal wildlife species [Gaines et al. 2010] and disturbance regimes consistent with major vegetation types);
3. logically and transparently identify potential landscape treatment areas (PLTAs), treatment patches, and the associated rationale; and
4. 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: (i) current patterns and departures of vegetation structure and composition from historical and climate change reference conditions; (ii) spread potential for wildfires, insect outbreaks, and disease pandemics across stands and landscapes given local weather, existing fuel and host conditions; (iii) damaging interactions among road, trail, and stream networks; (iv) wildlife habitat abundance, distribution, and sustainability; and (v) minimum roads analysis (i.e*.*, which of the existing system roads are essential and affordable, and which are not) (fig. 9).

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. 1999b, 2000a). Since that project, these reconstructed conditions have been summarized for each of the major ecological subregions (Hessburg et al. 2000b) 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 typical for the biota, climate, geology, and disturbance regimes of a physiographic region (Hessburg et al. 2000b), 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, and 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. 2010), and for patch scale fire behavior attributes under a 90th percentile wildfire burn scenario (Hessburg et al. 1999b). 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 1999b, 1999c, 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 floodplain, separate existing channels from off-channel features, and contribute the most to in-stream sediment. For additional details about the entire landscape evaluation process, we refer you to Gaines et al. (2010) and Hessburg et al. (2013).



Figure 9. Workflow diagram of the Okanogan-Wenatchee National Forest DST. The DST was implemented in EMDS, the Ecosystem Management Decision Support software (Reynolds et al. 2003, Reynolds and Hessburg 2005) 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 HRV and the FRV reference conditions. *FlamMap* and *Randig* (Agar 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.

**End of sidebar**

**5.b 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 ecosystems. Silviculture represents the operational process and offers a matrix of options to achieve these broad-scale objectives.

 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 due 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 dry and moist mixed-conifer forested landscapes of eastern Oregon and Washington had a relatively open stand structure under the influence of active fire (see Section 2, Section 4.a.1, and table 2 for more information). 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 towards the species composition and diversity in age class structure, and complexity in vertical and horizontal stand structure that is 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 (Hessburg et al. 1994, Fettig et al. 2007, Franklin et al. 2013, O’Hara and Ramage 2013). For many landscapes, development of a multi-scale 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, occurrence, in clumps of varying size, 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 towards 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 towards more open structures with a subtle but distinctive spatial, compositional, and age heterogeneity. Landscape-level planning will provide the guidance for these decisions.

 Silviculture employs both passive and active means. 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 towards these structures should also be variable. Implementing forest 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. At certain times during stand development 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.

 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 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*) 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 supplementing 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 by-product 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-level diagnosis and prescription, and tailored to resolve many concurrent issues that emerge from landscape analysis. 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 (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 is appropriate. Variable retention systems that leave residual trees after harvest (e.g., Mitchell and Beese 2002, Beese et al. 2003) 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. The challenge will be finding ways to systematically integrate variability into stand management. Recent efforts are making meaningful progress towards 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 (Larson and Churchill 2012, Churchill et al. 2013). This approach has been employed recently and demonstrates the innovation that is emerging in forest management to achieve these varied landscape objectives (Stine and Conway 2012, Churchill et al. 2013, Franklin et al. 2013). More extensive datasets are needed from additional physiographic regions and PVT settings to better understand the variability in within-patch clumpiness that can inform marking guidelines and silvicultural prescriptions throughout the mixed-conifer forest.

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 moist mixed-conifer 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.

 Off-site or non-local 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 tree hazard.

 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 outbreaks (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 (>~53 centimeters [21 inches] dbh) 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-sized 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, need to be retained whenever possible on eastside landscapes. However, strict and arbitrary limits that are not sensitive to site conditions, disturbance history, and topo-edaphic settings will 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.

**Section 6 SOCIAL AGREEMENT AND INSTITUTIONAL CAPACITY FOR RESTORING MOIST MIXED-CONIFER FORESTS**

 The social agreement and institutional capacity for restoring moist mixed-conifer 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 suggested ideas for how the Forest Service and its partners could approach restoration on national forests that would enhance its institutional capacity to implement the kinds of management strategies discussed in this synthesis document.

**6.a 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 attests to the need for forest restoration, as this science synthesis demonstrates. And research carried out nationwide finds that most people living in the wildland-urban interface perceive a high wildfire risk, and express a high level of public support for both thinning and prescribed-fire activities on public lands that exhibit high fire risk (McCaffrey 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. 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 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).

 Other studies of the social acceptability of forest restoration activities have found that support 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 (Winter et al. 2002, Brunson and Shindler 2004). 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 treatments and prescribed fire will be needed to restore the resilience of moist mixed-conifer forests. Research indicates that in general, there is social support for both mechanical and fire treatments, though prescribed fire has been found to be less socially acceptable than mechanical treatments 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 use because they result in “wasted” timber resources (Cacciapaglia et al. 2012). These landowners also saw fire as a bad 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). Thus activities that build trust between agencies and the public, and increase communication 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, decision making, 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) identifies 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 understandings of forest conditions and fire management practices can develop, and strong agency leadership exists in agency interactions with the public. It is important to remember, however, that social acceptability is dynamic by nature, and can change as people learn more or external variables shift (Shindler 2007). Thus, managers will need to pay continual attention to public values associated with forest restoration.

**6.b Collaboration**

 Decision-making 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. In Press); the best model will depend on local context and the nature of the issues. A number of community-based collaborative groups have formed around east-side national forests to address the ecological and economic issues associated with forestrestoration, 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.

Much research has been done to identify what is needed for successful community-based collaborative processes. McDermott et al. (2011) identify three sets of features. The first is external sources of support. These include the support and involvement of elected officials, agency leaders, and key decision-makers; enabling laws and policies; and community involvement. The second pertains to sufficient access to resources, such as funding, staffing, and information. The third regards the capacity to act, which depends on effective leadership, trust, and social capital. 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. Finding ways to successfully engage with, support, and help build the capacity of local community-based collaboratives east of the Cascades in Oregon and Washington is an important strategy for building social agreement around the management of moist mixed-conifer forests.

 Bartlett (2012) lays 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:

1. Include a broad range of participants;
2. Establish a common conceptual framework, purpose and need, and long-term desired condition;
3. Include scientific experts who serve as technical resources during meetings;
4. Move some intractable issues forward without complete consensus if necessary;
5. Include site visits to support decision-making and reach agreements; and
6. 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.

 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, sociology, and economics in the cadre of players who will help guide these national forest planning and management efforts.

**6.c 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 new 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 (or other public land managers) staff 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 of thousands of hectares (hundreds of thousands of acres).

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 staff, in collaboration with Research and Development 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 a significant learning opportunity. The U.S. 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 cross 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 moist mixed-conifer 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 prescribed 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 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 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 inches [~53 centimeters]) were intended to be an interim measure to screen out projects that were harvesting significant amounts of old forest and of remnant medium and large trees. There are a variety of reasons why managers were advised to avoid removal of trees above a certain diameter limit. 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 from forests also became significant beginning in the late ‘80s 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 used as a conservative 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 these 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 in the mid-1990s. However, other research on the ecology and history of pine and mixed-conifer forest (Hessburg et al. 1999b, 2005, Perry et al. 2004, Merschel 2012) indicates that many shade-tolerant trees over ~51 centimeters (20 inches) 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, recent study 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. Local variability coupled with an uncertain climate future requires an ability to adjust to site conditions and be nimble, as better 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. Such approaches would consider and remedy the key departures in forest structural and compositional patterns relative to their HRV and FRV, the latter 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 the warmer and drier conditions.

 As managers reevaluate their planning approaches using the concepts and findings contained in this document, they might also consider phasing out these interim directions. 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 in the stand or the landscape, or the potential of these areas to grow large trees. The limits on removing any tree larger than ~53 centimeters (21 inches) 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.

The concepts presented in this document address the underlying ecological objectives of the eastside screens. As these concepts are explored and implemented by managers, the ecological goals of 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 towards 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. With all its limitations, the existing approach may carry social resistance to change.

**Section 7 – CONCLUSIONS**

Active management of vegetation in MMC forest in eastern Oregon and Washington has declined over the last two decades because of uncertainties about management. During this time, wildfires and wildfire suppression have continued, insect outbreaks have contributed to fuels, and shade-tolerant trees have regenerated and been released. 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 paper offers scientific findings and management considerations to help managers take informed steps towards restoring resilience to MMC forests.

Dry, moist and wet mixed-conifer forests co-exist in landscape mosaics that have been altered by settlement and land use activities. On balance, the extant changes make these forests and landscapes subject to large fires, which express elevated mixed- and high-severity components, insect outbreaks, and drought-related tree mortality that negatively affect many ecological and social values. During the pre-settlement era, the dry *and* MMC forest landscapes were sculpted by low-, mixed-, and high-severity wildfires (see table 9) that maintained relatively open and understories where influenced by low-severity fires, patchy landscapes where dominated by mixed-severity fires, and simplified mosaics where influenced by high-severity fires. Landscape mosaics were dominated by a mix of species, sizes, and ages (see table 9). 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 vary 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 post-settlement activity on forests and implications to restoration are the same in MMC forests are related to 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 and 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 wetter mixed-conifer types where restoration and recreation of resilient vegetation is less needed or inappropriate given their disturbance regimes.

Given the stated restoration goals of the U.S. 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.

Stand (patch) and local landscape, as well as regional landscape-scale perspectives are needed to achieve restoration goals. Although we address all three scales in this document, 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 appendix A). 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. 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 document 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 community 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. 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 paper. 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 socio-ecological resilience, creating ecologically resilient landscapes cannot occur without support and consideration of the institutions and socio-economic components. Restoration and creation of resiliency 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 in building and currently widespread, will require a sustained, comprehensive, and collaborative approach to remedy.

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**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. While somewhat counterintuitive, 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 functions of an ecosystem through natural events such as fire, flood, wind, or earthquake; mortality caused by insect or disease outbreaks; 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. Frequency and intensity of these “natural” disturbances are the focus of interest.

**Ecophysiological** - Relating to the interrelationships between an organism's physical functioning and its 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; and

 4. Cultural services - such as educational, aesthetic, spiritual and cultural heritage values, as well as recreational experiences and tourism opportunities.

**ENSO** - El Nino Southern Oscillation; is 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 regime** - The characteristics of fire in a given ecosystem, such as the frequency, predictability, intensity, and seasonality of fire.

**Fire severity** - Fire severity denotes 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 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 and 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 listing, often appearing in the form of a curve or graph, of the frequency with which possible values of a variable have occurred.

**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 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 of components of healthy ecosystems over time.

**Integrity** - An ecosystem has integrity if it retains its complexity, biotic and abiotic processes intact, 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 a mosaic of different vegetation or habitat conditions.

**Landscape** - A large land area composed of interacting ecosystems that are repeated due to 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 organized as follows:

* Region 1,000,000s hectares
* Landscapes 10,000s hectares
* Watersheds 100s to 1,000 hectares
* Stands (patches) 1-10s hectares
* Gaps 0.01 to .1 hectare

**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 include 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, the relative humidity increases to 100 percent and creates clouds and sometimes precipitation.

**Patch** - An area of homogeneous vegetation, in structure and composition.

**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 (for example the dry shrub PVG). They are often grouped by similar types of life forms.

**Potential vegetation yype (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)* type. Potential vegetation types are identified by indicator species of similar environmental conditions. For example, Douglas-fir indicates a cooler and moisterenvironment than ponderosa pine *(Pinus ponderosa).* Because of growth, mortality, anddisturbance 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 (National Forest System Land Management Planning, 36 CFR 219.19) restoration 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.

**Stand** - 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.

**Appendix A.** **List of practical considerations for landscape evaluation and restoration planning.**

 This list of management considerations is intended to illustrate 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 it relates 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 non-native plant and animal invasions, priority 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 socio-economic 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 socio-economic 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:
	1. What are the current patterns of physiognomic types, cover types, structural stages or successional classes, stand density and basal area classes?
	2. What is the density and distribution of large old, fire-tolerant/intolerant trees, density and basal area of shade-tolerant trees?
	3. What are current surface and canopy fuel conditions?
	4. What is the vulnerability of the current vegetation to major insect and pathogen disturbances?
	5. 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:
	1. What were the historical patterns of physiognomic types, cover types, structural stages or successional classes, stand density and basal area classes?
	2. What was the historical pattern and distribution of large old, fire-tolerant trees?
	3. 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?
	4. What is a set of guiding reference conditions (potential restoration and resilience targets) based on historical patterns of structure and composition?
	5. What is a desired set of class and landscape pattern metrics to characterize these reference conditions?
	6. 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:
	1. Where within the local landscapes have disturbance regimes and vegetation structure and composition have been significantly altered by Euro-American activities?
	2. How does this departure vary by topography, soils, 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:
	1. What do climate change models predict to be the most likely climate change scenarios for each ecoregion? Identify 1 or 2 most likely climate change scenarios for each ecoregion; use existing information from models or expert opinion.
	2. What are predicted future patterns of physiognomic types, cover types, structural stages or successional classes, stand density and basal area classes?
	3. What is a set of guiding reference conditions (potential restoration and resilience targets) based on future patterns of structure and composition?
	4. What is a desired set of class and landscape pattern metrics to characterize these reference conditions?
	5. 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.
	1. Tools such as [FlamMap](http://www.firemodels.org/index.php/national-systems/flammap), a fire behavior mapping and analysis program, are currently being used for this.
	2. Terrain routed wind flow projections can be obtained from [Wind Ninja](http://www.firelab.org/research-projects/physical-fire/145-windninja), [Wind Wizard](http://www.firelab.org/research-projects/physical-fire/127-wind-wizard) or similar simulation tools.
	3. To initialize wind speeds and directions, use meteorological data from nearby weather stations where available, or expert advice as needed.
	4. Determine the mapped distribution of burn probability, probable flame length, fireline intensity, rate of spread, crown fire ignition and spread potential.
	5. Note the hotspots on the landscape for high probability flame length and fireline intensity.
	6. 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:
	1. How have spatial patterns of surface and canopy fuels changed with respect to these references?
	2. How will changes translate to changes in expected fire behavior?
	3. What are the key changes by PVT setting?
	4. 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, beginning with listed and focal species.
	1. Map habitats of focal wildlife species at regional and local scales.
	2. Determine or estimate current population conditions for key species based on available information and habitat models.
	3. Conduct regional metapopulation and habitat analyses for large bodied animals with large home ranges.This would include but not be limited to ungulates (e.g., elk, moose, and caribou), carnivores (e.g., lynx, wolverine, grizzly bear, black bear, wolf, and bobcat) and predatory birds (e.g., owls, hawks, eagles).These analyses are useful for sustaining regional populations and detecting pinch points in space and time. Regional habitat analyses should also examine patterns and distributions of habitat conditions for their ability to allow unimpeded animal movement between suitable core habitats. With climatic warming, this regional landscape connectivity will likely decline for a number of species. See PNW GTR 485 (Wisdom et al. 2000) for more information on habitat analyses.
* 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.
	1. Identify all roads crossing streams, culverts, and other fish passage barriers.
	2. Identify road segments that confine stream channels, reduce access to off-channel features, and contribute the most sediment.
	3. Identify portions of the road network that most reduce subsurface water flow and accelerate runoff.
	4. Identify the portions of the road network that are most essential for access.
* Evaluation of the stream network of the current landscape is recommended.
	1. Identify and map existing anadromous and cold water fish habitats, especially for listed or focal species.
	2. Identify and map habitats with inherent potential for these same species.
	3. 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.
	1. Identify anadromous and cold water fish habitats that are prone to erosion, especially those of regionally listed or focal species.
	2. 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 socio-economic opportunity factors can be effective in spatially allocating treatment priorities and identifying proposed landscape treatment areas (PLTAs). Spatial decision support tools can help users jointly consider the spatial allocation of treatment priorities across a multitude of factors.
	1. Assess and map current land use and land designations and whatever constraints, challenges, or opportunities this suggests.
	2. Develop and identify PLTAs that restore key concerns identified in analysis.
	3. Among the PLTAs, vary the emphasis among the concerns to provide a wide variety of management alternatives to consider.
	4. Array and compare the social and economic benefits and costs of each alternative.
	5. 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?
	6. Develop a landscape diagnosis and prescription for the best subset of alternatives based on stakeholder inputs and the condition of the resources.
	7. Assess potential effects of proposed work on adjoining lands and possible effects of ongoing activities on adjacent lands.

 

Figure 8. Schematic diagram of a suggested landscape evaluation workflow.

* 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 for the consideration of managers in conducting landscape evaluations to achieve sustainable land management objectives. They are not requirements for the use of the best available scientific information in project planning, but are useful to inform managers of the science relevant to a proposed action.

Appendix B Regional-Scale Fire Regimes Group Classification for Oregon and Washington (2002 Louisa Evers).

Most moist mixed-conifer fire regimes fall within types I and III a and b.

# 0-35 years, Low severity.

Typical climax plant communities include ponderosa pine, eastside/dry Douglas-fir, pine-oak woodlands, Jeffery 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).

# 0-35 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 25-35 years and occasionally up to 50 years, and mountain shrub communities (bitterbrush, snowberry, ninebark, ceanothus, Oregon chaparral, etc.) with typical return intervals of 10-25 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.

# 35-100+ 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 (10,000-100,000 acres [~5,000-50,000 hectares]) but subsequent mixed-severity fires are important for creating the 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.

## 50-100 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.

## 100-200 years, Mixed severity

Typical potential plant communities include western hemlock, Pacific silver fir, and whitebark pine at or below 45 degrees latitude and cool, mesic grand fir and Douglas-fir. Higher-severity fire tends to dominate in many events.

# 35-100+ 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 degrees latitude.

## 35-100+ years, High severity, Juxtaposed

Typified by what would normally be considered 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 5000 feet 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 degrees 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 100-150 years.

## 100-200 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 degrees 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 non-existent 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.

## 200-400 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.

1. The U.S. 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 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.” [↑](#footnote-ref-1)
2. 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. [↑](#footnote-ref-2)