2016

Blackfoot-Swan Landscape Restoration Project

LANDSCAPE ASSESSMENT FOR TERRESTRIAL FOREST ECOSYSTEMS

JONATHAN B. HAUFLER, CAROLYN A. MEHL, AND SCOTT YEATS

ECOSYSTEM MANAGEMENT RESEARCH INTITUTE | Seeley Lake, Montana | emri.org

This landscape assessment was completed through a Challenge Cost Share Agreement between the Ecosystem Management Research Institute and the USDA Forest Service Lolo, Flathead, and Helena National Forests (#14-CS-11011000-037)

Table of Contents

INTRODUCTION

Purpose

The Blackfoot-Swan Landscape Restoration Project (BSLRP) was initiated by Region 1 of the USDA Forest Service in 2014. The intial title for the project was the Restoration Initiative for the Blackfoot and Swan (RIBS) but was changed to its new title in 2015. The initiation letter identified the project as a restoration initiative with the primary objectives of reducing uncharacteristic wildfire risk and conserving terrestrial and aquatic biodiversity, while also considering future climate change. BSLRP is a NEPA process to develop and implement specific desired actions over a potential ten year timeframe. The original need for BSLRP was recognized as an output of the Southwest Crown of the Continent (SWCC) Collaborative Forest Landscape Restoration Program (CFLRP). The SWCC initiative includes a landscape of approximately 1.4 million acres and is addressing a number of restoration, wildfire, and watershed objectives.

Because of the breadth of BSLRP's stated purposes and the large scale of its application, it is important to consider the project in terms of its relationship to the larger landscape including its historical, current, and desired future conditions. An important component of which is to understand the terrestrial landscape including its ecosystem diversity, species considerations, and cumulative changes that have occurred from historical or reference conditions, in order to determine the best desired future conditions (Keane et al. 2009). This terrestrial landscape assessment was initiated to provide the framework and analysis for characterizing upland forest ecosystem diversity in the project area for both historical and current conditions and to evaluate species of concern in relation to changes in ecosystem diversity. It is the product of the Ecosystem Management Research Institute and does not necessarily represent U.S. Forest Service BSLRP or Forest Plan Revision Interdisciplinary Teams' analyses, review, or public input.

To effectively conserve biodiversity, a conservation strategy should be selected using two criteria; 1) evidence of a strong scientific foundation to conserve biodiversity over the long-term, and 2) the ability to conduct land management such that it is compatible with the strategy. Inconsistent, partially applied, or generalized approaches to implementing the strategy may compromise or even undermine the scientific foundation, thereby reducing the likelihood of achieving the long term objectives. The U.S. Forest Service has recently updated and described its primary conservation strategy for biodiversity. This strategy and its scientific foundation are presented in the ecological sustainability objectives of the 2012 USFS Forest Planning Rule and is further supported by the 2016 USFS Ecosystem Restoration Policy. Both of these directives focus on historical ecosystem diversity as the foundation for the conservation strategy and defining restoration, as well as the basis for identifying desired restoration conditions of forests and grasslands. Future management decisions that deviate from the desired restoration conditions should be identified based on other social or economic objectives, or because achieving or maintaining historical ecosystem diversity is deemed unfeasible or unsustainable. The Ecosystem Restoration Policy stated: "Ecological restoration focuses on reestablishing the composition, structure, pattern, and ecological processes necessary to facilitate terrestrial and aquatic ecosystem sustainability, resilience, and health under current and future conditions….The desired future condition of an ecosystem should be informed by an assessment of spatial and temporal variation in ecosystem characteristics under historical disturbance processes during a specified reference period." This landscape assessment is based on the conservation strategy for ecological sustainability as described in the Forest Planning Rule and the Ecological Restoration Policy of the Forest Service. The strategy requires a sufficiently rigorous classification and analysis of ecosystem diversity at appropriate scales to identify the range of native ecosystem conditions that function as the foundation for the conservation strategy and the scientific basis for future restoration goals. The terrestrial landscape assessment emphasized developing an appropriate classification of upland forest ecosystem diversity, quantifying and characterizing the ecosystem diversity for historical conditions, comparing these conditions to current conditions to identify cumulative changes, describing historical reference conditions to aid in determining desired future conditions, and assessing the implications of changes in habitat conditions for selected terrestrial wildlife species.

Project Area

The area analyzed for the terrestrial landscape assessment is the same area identified as the SWCC-CFLRP project area (Figure 1) and represents approximately 1.42 million acres. The BSLRP project made modifications to this boundary as identified in Figure 1, with a reduction in acres to approximately 1.28 million. While there is considerable overlap, slight differences will result when comparing acreage values between the two project areas. From an ecological perspective however, the results will be functionally the same.

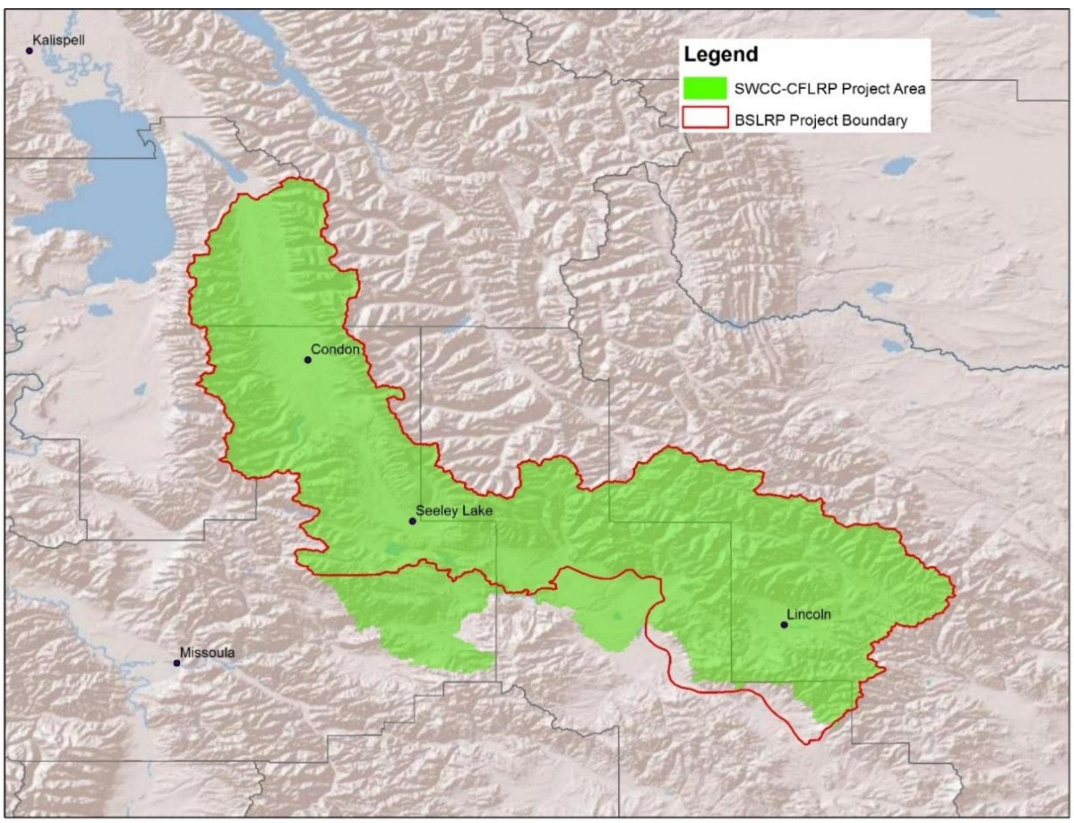


Figure 1. Map of the SWCC-CFLRP project area and the boundary of the BSLRP Area.

CONSERVATION STRATEGY

As discussed, the conservation strategy for biodiversity used in this landscape assessment parallels that described in the Forest Planning Rule and the Ecosystem Restoration Policy. The application of this conservation strategy to a large-scale NEPA project such as BSLRP is somewhat different from the forest planning process, however the scientific foundations and primary analyses and assumptions of the strategy are the same. The conservation strategy is based on applying a coarse-filter approach as the primary focus for ensuring ecological sustainability and biodiversity conservation. A coarse-filter approach, as defined and described in the Final Programmatic EIS for the 2012 Planning rule (USDA Forest Service 2012), maintains or restores sufficient amounts of all native ecosystems in a designated area. For this assessment, upland forest ecosystems were evaluated. A coarse-filter approach is based on the premise that historically occurring ecosystem diversity, when properly classified, characterized, and provided in sufficient amounts and patterns will provide the diverse habitat conditions needed to support all biodiversity (Haufler et al. 1999, 2002). This is accomplished with careful consideration to the scale and resolution of ecosystem classification used to quantify historical ecosystem diversity. It also requires a comparison of the amounts and conditions that occurred historically to the current ecosystem diversity present in the landscape to understand the cumulative changes to native ecosystem diversity. An ecosystem is defined (U.S. Forest Service Land Management Planning Handbook 2015) as “a spatially explicit, relatively homogeneous unit of the Earth that includes all interacting organisms and elements of the abiotic environment withinin its boundaries. An ecosystem is commonly described in terms of its:

1. Composition. The biological elements within the different levels of biological organization, from genes and species to communities and ecosystems.
2. 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.
3. Function. Ecological processes that sustain composition and structure, such as energy flow, nutrient cycling and retention, soil development and retention, predation and herbivory, and natural disturbances such as wind, fire, and floods.
4. Connectivity. Ecological conditions that exist at several spatial and temporal scales that provide landscape linkages that permit the exchange of flow, sediments, and nutrients; the daily and seasonal movement of animals within home ranges; the dispersal and genetic interchange between populations; and the long distance range shifts of species, such as in response to climate change.”

Secondarily to the coarse-filter, a fine-filter or species assessment is also delveloped to analyze how conditions for species of concern may have changed from historical conditions and evaluates the habitat amounts for these species in the project area today. This also identifies a process for evaluating how planned future management for ecosystem diversity will provide for species of concern in relation to their historical habitat conditions. The scientific foundation of this conservation strategy requires a consistent methodology to identify and characterize all of the historically occurring ecosystems within and across a landscape, as well as careful and consistent methods to ensure sufficient representation of historically occurring ecosystems if the strategy is to be effective in meeting the biodiversity objective. A more detailed description of this strategy and its scientific foundations and application to BSLRP is provided in Appendix A.

Assessment Process

The process used in this terrestrial landscape assessment included the following steps:

1. Classify and characterize native ecosystem diversity
   a. Identify and delineate appropriate-sized landscapes or ecoregions for analysis
   b. Identify and map ecological sites (i.e., delineate the abiotic environment)
   c. Develop state and transition models for each ecological site that provide the ability to characterize historical compositions, structures, patterns, and ecological processes (i.e., delineate disturbance states)
   d. Quantify historical amounts of each ecosystem (i.e., ecological site and disturbance state)
   e. Describe historical reference conditions for each ecosystem
   f. Quantify current amounts of each ecosystem
   g. Quantify cumulative changes to native ecosystem diversity
2. Select and model habitat conditions for species of concern
   a. Determine historical habitat for each species based on historical ecosystem diversity
   b. Determine current habitat conditions for each species based on current conditions
3. Identify primary stressers that have caused changes to ecosystem diversity
4. Discuss the implications of the landscape assessment on developing desired future conditions.

The general steps followed in this landscape assessment process are depicted in Figure 2.

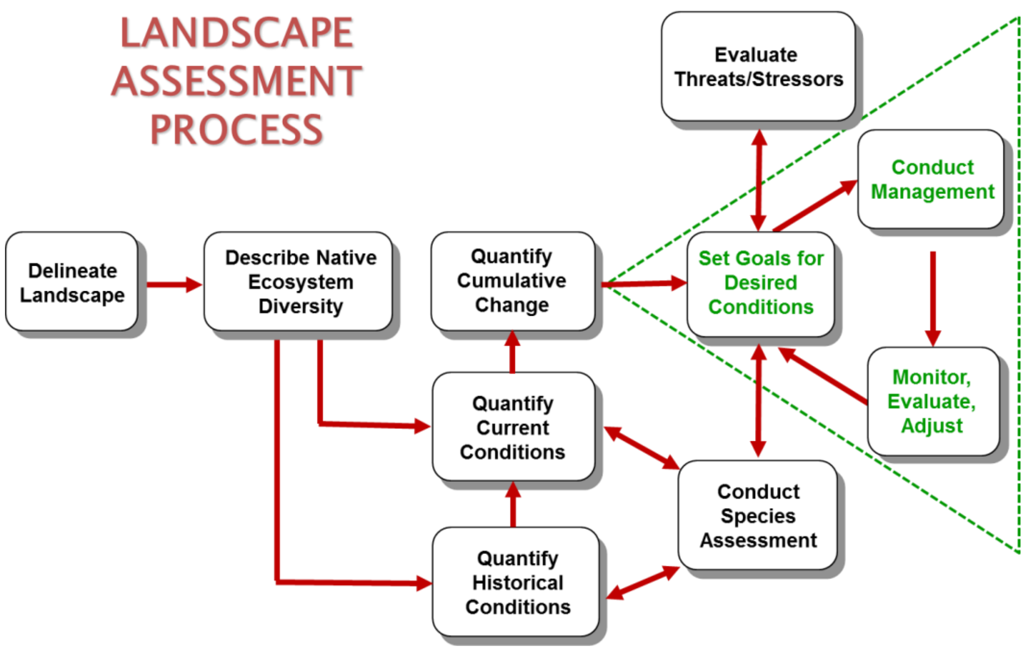


Figure 2. Steps in the landscape assessment process used in the terrestrial landscape assessment. The steps in green represent future considerations and actions and are not included in the landscape assessment. Results of the landscape assessment are used to inform the process for developing future desired conditions.

ECOSYSTEM DIVERSITY ASSESSMENT

The conservation strategy emphasizes characterizing the native (historical) ecosystem diversity of the project area as the foundation for assessing cumulative changes in a landscape. This includes classifying ecosystem diversity at a scale and resolution appropriate to function effectively for the identified conservation strategy. It also includes quantifying the historical and current amounts of each ecosystem to determine the changes that have occurred, and describing reference conditions for each ecosystem in terms of compositions, structures, and processes in order to clearly identify the conditions needed to meet the conservation strategy objectives. The goal of restoration is to return an ecosystem to a previous condition; usually one that represents an historical or resilient condition (USFS Ecosystem Restoration Policy, Clewell and Aronson 2013). For terrestrial ecosystems, this usually means characterizing, and where needed restoring the historical plant communities that occurred within a landscape. Maintaining or restoring native ecosystem diversity as well as determining the best locations for restoration requires an understanding of the interaction of the abiotic environment with historical disturbance processes (Nichols et al. 1998). More specifically, it means understanding the dynamic range of native ecosystems that occurred historically due to the interaction of site influences and disturbance processes. Characterizing and quantifying historical disturbance processes is therefore an important component of a terrestrial landscape assessment.

Historical Disturbance Processes

Fire was the primary disturbance agent in forest ecosystems of the northern Rockies, directly influencing plant species composition, structure, and patterns (Hutto et al. 2016, Marcoux et al. 2015, Larson and Churchill 2012, Heyerdahl et al. 2012, 2008, Long 2009, 2003, Hessburg and Agee 2003, Keane et al. 2002, Arno et al. 2000, Agee 1993, Romme and Despain 1989, Fischer and Bradley 1987, Wellner 1970). Insects and disease were also historical disturbance agents, however, their influence on forest ecosystems has increased considerably in the last century due to fire suppression activities and some logging practices, and more recently due to the effects of climate change, with corresponding changes to plant community compositions and structures (Mershel et al. 2014, Franklin et al. 2014, Larson et al. 2009, Fettig et al. 2007, Parker et al. 2006, Kipfmueller and Kupfer 2005, Keane et al. 2002, Hessburg et al. 2000, Anderson et al. 1987). Insects and disease also interact with fire and can precede and contribute to the occurrence and severity of fire (Parker et al. 2006, Bigler et al. 2005, Howe and Baker 2003, Turner et al. 1999), while fire in turn may cause increases in tree susceptibility to certain insects (Merschel et al. 2014, Fettig et al. 2007, Parker et al. 2006). In the Rocky Mountain region, snow avalanches are another disturbance event, albeit on a smaller, more localized scale, that can also contribute to ecosystem diversity (Bebi et al. 2009).

The historical role of fire in the northern Rockies is presented below along with a brief discussion of the role of insects and diseases. Recent trends in reported changes over the past 100 years to these disturbance processes are also presented.

Fire

Fire was a natural part of the Northern Rockies landscape for thousands of years and many species of plants and animals have become fire-adapted or even fire-dependent over time in response to this important disturbance event (Marcoux et al. 2015, Heyerdahl et al. 2012, 2008, Larson and Churchill 2012, Keane et al. 2002, Arno et al. 2000, 1985, Agee 1993, Arno 1980, Ayres 1900). Based on historical accounts (Arno 1980, Gruell 1982, Wellner 1970, Ayres 1900) and recent fire-scar or fire mapping studies (Marcoux et al. 2015, Heyerdahl et al. 2012, 2008, Agee 2004, 1998, 1993, Barrett 2002, Arno et al. 1997, Barrett et al. 1997, 1991, Fischer and Bradley 1987), fire in the northern Rockies was a relatively frequent disturbance event prior to Euro-American settlement. Many scientific reports as well as anecdotal reports have documented the widespread occurrence of fire throughout the region. The causes of these fires were both natural (i.e., lightning) and human-initiated (i.e., Native Americans) (Keane et al. 2006, Hessburg and Agee 2003, Barrett and Arno 1982, Barrett 1981). Native Americans interacted and influenced historical ecosystem diversity for thousands of years but typically their influence was an extension of naturally occurring disturbance processes that would benefit their subsistence strategies, such as using fire to create better wildlife habitat for hunted species, to improve berry producing shrubs, or to open travel corridors (Williams 2003, Arno et al. 1997, Barrett 1981), although Barrett et al. (2005) cautioned to not overstate the role of Native American ignited fires compared to lightning caused fires.

Insects and Disease

Insects and diseases are also important disturbance processes affecting forest ecosystems in the northern Rockies (Hagle et al. 2003). Brunelle et al. (2008) reported on evidence of bark beetles occurring in high elevation forests in the Holocene based on core sampling of lake sediments, confirming their long historical presence. Various insects and disease are native to the northern Rockies and contributed to disturbance processes. Hagle et al. (2003) provided a good description of the primary insects and diseases affecting conifers in the northern and central Rockies. These included Armillaria root disease, Douglas-fir beetle, Douglas-fir tussock moth, fir engraver, Indian paint fungus, mountain pine beetle, spruce beetle, western pine beetle and western spruce budworm. All of these were reported to experience population increases in response to high tree densities (Powell 1999). Parker et al. (2006) provide a good overview of the various interactions of insects, diseases, and fires occurring in the northern Rockies. Insects and diseases will be influenced by ecological site as well as past disturbance history at a specific location (Stine et al. 2014, Parker et al. 2006, McDonald et al. 2000).

Recent Trends

Over the last 30 or more years, forest ecologists have conducted field research to document the effects caused by changes to historical fire regimes in the northern Rockies. These documented changes include increased densities of trees in many locations where low to moderate severity fires occurred historically (Hessburg et al. 2016, Marcoux et al. 2015, Hanberry 2014, Abella et al, 2007, Carr 2007, Keane et al. 2002, Keane et al. 2006, Keeling et al. 2006, Hessburg et al. 2005, Fitzgerald 2004, Arno et al. 2000, Hartwell et al. 2000, Hessburg et al. 2000b, O'Laughlin 1998, Arno et al. 1995, Habeck 1990), increases in shade tolerant species and reductions in early seral species (Hessburg et al. 2016, Hanberry 2014, Abella et al, 2007, Keane et al. 2006, Keeling et al. 2006, Parker et al. 2006, Fitzgerald 2004, Hessburg et al. 2000b, Hartwell et al. 2000, Arno et al. 1995, Habeck 1994, 1990), reductions in numbers of large trees (Hessburg et al. 2016, Hagmann et al. 2013), increases in insect and diseases (Hessburg et al. 2016, Parker et al. 2006, Keane et al. 2002, Hessburg et al. 2000, Kolb et al. 1998, Hessburg et al, 1994, Veblen et al. 1994, Anderson et al. 1987), and increases in intensities of fires (Cansler and McKenzie 2014, Morgan et al. 1996).

Recent papers by Hutto et al. (2016), Baker (2015), Odion et al. (2014), Williams and Baker (2012b) and others report high severity fires occurred historically in dry and mixed conifer forests in the west, with Baker (2015) reporting current rates of high severity fires in dry forests of the west are within historical ranges of variability. The reported findings of these authors seem to differ from the results of many previous researchers. Yet when an ecosystem-level context is considered, some of these apparent differences can likely be explained by a number of factors including:

- Lack of distinction between fire regimes and fire severity,
- Generalized descriptions of fire regimes or severities based on coarse classifications of vegetation, such as “mixed conifer forests” rather than finer classifications as used in some previous research and in this assessment,
- Failure to recognize ecoregional differences in fire regimes for similar overstory dominant tree species, such as using ponderosa fire regime information from the southwestern U.S. and applying it to the northern Rockies,
- Failure to recognize local variations in fire regimes caused by ecological site such as topography, aspect, and other landscape influences, and
- Different methodologies used in assessing historical fire regimes and severities.

While it is clear high severity fires were an historical disturbance in the northern Rockies that includes the SWCC landscape assessment area, it is also clear there has been substantial changes in fire regimes from what occurred historically. When viewed across a variety of information sources including fire scar analyses, historical accounts, historical photographs, and historical forest surveys, there is little doubt that fire was a much more prevalent occurrence in historical landscapes than occurs today. In much of the northern Rockies, mixed severity fire regimes were a predominant influence (Marcoux et al. 2015, Hanberry 2014, Abella et al, 2007, Carr 2007, Arno et al. 2000, Hessburg et al. 2000, O’Laughlin 1998, Harvey 1998, 1994, Habeck 1994, 1990, Ayres 1900). Ayres (1900) conducted a reconnaissance survey in the project area in the late 1800’s that included photographs. His results presented substantial evidence of the role of historical fire in the landscape assessment area, including areas of high severity, moderate severity, and low severity fire. Higher elevation and steeper slopes were more often influenced by high severity fire that set the forest back successionally, while lower elevation and flatter terrain were shown to have been influenced by low to moderate severity fires, resulting in more open or mixed structured stands of relatively large and old western larch and ponderosa pine. Further, the very presence of these large trees and fire tolerant species is a clear indicator of non-lethal and mixed-severity fire regimes that produce the beneficial growing conditions that allow them to occur (Marcoux et al. 2015). These data are specific to the northern Rockies, but are also supported by studies in other northwestern ecoregions such as the findings of Hessburg et al. (2016) and Hagmann et al. (2014, 2013) for the eastern Cascades that present strong evidence for changes in forest conditions occurring over the past 100-150 years.

An additional result of changes to ecosystem compositions and structures due to fire exclusion and other anthropogenic influences has been increases in the frequency and amounts of insect and disease outbreaks (Hessburg et al. 2016). Fettig et al. (2007) provided strong evidence that as stand densities and basal areas increase, so will susceptibility to associated insect infestations, such as pine beetles, Douglas-fir beetles, and spruce beetles. Keane et al. (2002) reviewed literature on cascading effects of fire exclusions, including changes to insect and diseases in forest ecosystems, and noted many insect and diseases had heightened activity as a result of fire exclusion as reported by Veblen et al. (1994). Increased activity of dwarf mistletoe and mountain pine beetle was reported where greater densities of lodgepole and ponderosa pines occurred caused primarily by the absence of fire (Wilson and Tkacz 1996, Covington et al. 1994, Zimmerman and Laven 1984, Alexander and Hawksworth 1976). Similarly, fire exclusion was reported to increase the occurrence of spruce budworm epidemics (Hadley and Veblen 1993, Swetnam and Lynch 1993, Holland 1986, Carlson et al. 1983). Fire exclusion typically allows subalpine fir and Engelmann spruce to increase in whitebark pine ecosystems and thereby can increase the occurrence of root rot and other diseases (Alexander et al. 1990, Arno and Hoff 1989). Hessburg et al. (2016) also noted the increased incidence of insects and diseases.

Fettig et al. (2007), Garrison-Johnson et al. (2003) and Negron et al. (1999) reported higher incidence of Douglas-fir beetle in stands with greater densities of Douglas-fir. Changes in stand densities observed over the past 100 years would therefore be expected to have caused greater outbreaks of Douglas-fir beetles on susceptible sites. Garrison-Johnson (2003) noted these effects were also influenced by the specific type of site supporting a stand of Douglas-fir. Bassman et al. (2003) reported that widespread changes in forest compositions caused by past harvests and fire exclusions have increased insect and pathogen populations. Parker et al. (2006) noted: "Trees weakened by pathogens and/or insects may also suffer greater mortality during fire than healthy trees (Harrington and Hawksworth, 1990; Conklin and Armstrong, 2001)." Thus, fire exclusion has resulted in increased stand densities in many locations, and has caused shifts towards greater compositions of shade tolerant species, both of which often lead to greater occurrences of insects and diseases, particularly on drier sites. These outbreaks can further stress trees, making them more susceptible to fire-caused mortality when fire does return to a stand (Parker et al. 2006).

Classification

Ecological classification is the process of defining discreet categories of ecological conditions based on common attributes, using environmental and/or biological variables (Grossman et al. 1999). It is essential for describing and quantifying ecosystem diversity. Forest ecosystems, as defined and used in this assessment each have specific compositions, structures, and processes and as such can provide a detailed description of a variety of associated vegetation attributes. Ecosystem diversity is the variety and relative extent of these ecosystems, and an ecosystem is considered to have ecological integrity when its dominant ecological characteristics such as composition, structure and functions are within the range of conditions that occurred historically (FSH 1909.12- Land Management Planning Handbook 2015). Thus, each specific ecosystem can be characterized and described in terms of its composition, structure, disturbance processes, and connectivity. Ecosystem diversity should incorporate both biotic and abiotic components in classifications (Grossman et al. 1999).

Ecological classifications serve to allow mapping of different ecological units, but with recognition that classification is independent of mapping (Grossman et al. 1999). Ecological classification is a process of arranging units of quantitative information into groupings with common properties (Grossman et al. 1999), and is done at an appropriate resolution to address the objectives for use of the classification

(Haufler et al. 1999c). Mapping is the representation of classified units and is constrained by the spatial sources of information (Grossman et al. 1999).

Native ecosystem diversity results from the interaction of the biotic and abiotic variables creating the spatial heterogeneity of an area (Winter et al. 2011). Classification of ecosystem diversity, therefore, has the objective of identifying the combination of biotic and abiotic factors at appropriate resolutions to allow the determination of more discrete, homogeneous units that describe and quantify the full array of ecosystems in a planning landscape. Hierarchical-type classifications allow for the delineation of planning landscapes containing similar geology, climate, or other conditions (Grossman et al. 1999) and reduces the variability in the ecosystems requiring classification within each landscape.

The classification of ecosystems often begins with delineations of the abiotic environment to characterize the inherent diversity of the target landscape. Coarser classifications of the abiotic environment increase the variability in the types of conditions included in a specific class (Mershel et al. 2014, Abella and Denton 2009), reducing the ability to assign specific characteristics to that class, such as predominant fire regimes or other disturbance processes, or the specific biotic communities that can occur on each abiotic site. It is important to identify the appropriate resolution to ensure the classification is ultimately useful for its intended purpose. Similarly, classifying the specific biotic communities occurring within an abiotic site requires evaluation of the resolution. As Marcoux et al. (2013) reported, finer resolution classification of vegetation conditions and biophysical environments allowed for more site specific characterizations of fire regimes occurring on each site. However, too fine a resolution increases the complexity of describing and mapping ecosystems, and may become unwieldy for management purposes.

One way of testing whether ecosystems are classified with sufficient resolution is by evaluating how selected common plant species may be distributed within specific site classes. If a species only occurs over part of the range of conditions for a site class, such as a species occurring only in the eastern versus western parts of a delineated landscape, then the classification may be too coarse. For example, if a tree species, such as western larch, is a major species for a specific site class, but only occurs on the western half of a delineated landscape, then this would indicate that the classification may be too coarse. Two solutions would be to further separate or sub-delineate the assessment area to reduce the variability in species distributions, or to break that site class into additional classes to capture differences in species composition between different portions of the landscape. Deciding appropriate classification resolution should always be done in the context of the objectives of the overall conservation strategy.

Two types and scales of ecological classification are important to meeting the conservation strategy objectives for the BSLRP terrestrial assessment – the landscape level and ecosystem-level. The following sections describe each of these levels in more detail and with application to the objectives of the BSLRP project.

Landscape Level

Landscape-level classification systems identify the regional boundaries within which ecosystem diversity is then classified. Over the past several decades, these regional boundaries have become more frequently referred to as ecoregional boundaries (Grossman et al. 1999) when they are used in biodiversity and natural resource planning efforts. Each ecoregion boundary is typically delineated based on similarities in climate, physiography, hydrology, vegetation, and wildlife habitat potential. In addition, natural disturbances are often constrained by the underlying physical features of soils and topography captured in a landscape-level or ecoregional boundary classification system. Various classification systems have been developed for these purposes including Major Land Resource Areas (MLRAs) by the U.S. Natural Resources Conservation Service and the National Hierarchy of Ecological Units (Cleland et al. 1997), also known as ECOMAP, as developed and used by the U.S. Forest Service. We selected the section-level ecological unit of the National Hierarchy classification as the starting point for landscape–level classification to meet the objectives of this project for 2 primary reasons: 1) the section-level size is large enough to encompass the primary processes needed to maintain/restore historical ecosystem diversity but not so large to preclude classifying ecosystems with a sufficient level of detail; and 2) the section-level is delineated using geo-climatic, soil, and potential vegetation inputs that will help reduce the variability when classifying ecosystem diversity. The section-level ecological unit will represent the landscape-level classification for the purpose of this landscape assessment and will be referred to as ecoregion boundaries from this point forward. Some refinements were made to the existing ecoregion boundaries to further reduce the expected variability in ecosystem diversity and are described in the following section.

Methods

Since the initial development of ECOMAP in 1997, several versions of the section-level ecoregions have been mapped and made available in GIS format. Each version was evaluated relative its application to ecoregion delineation for the BSLRP landscape assessment. To that end, portions of the ecoregion map developed for the Interior Columbia River Basin Ecosystem Management Project (Hessburg et al. 2000a) and the 2007 ECOMAP version were selected to best meet the objectives for classifying ecosystem diversity. Figure 3 presents the overlay of the ICBEMP section-level boundaries (red lines) with the ECOMAP section-level boundaries (blue lines) in relation to the SWCC-CFLRP project area (green).

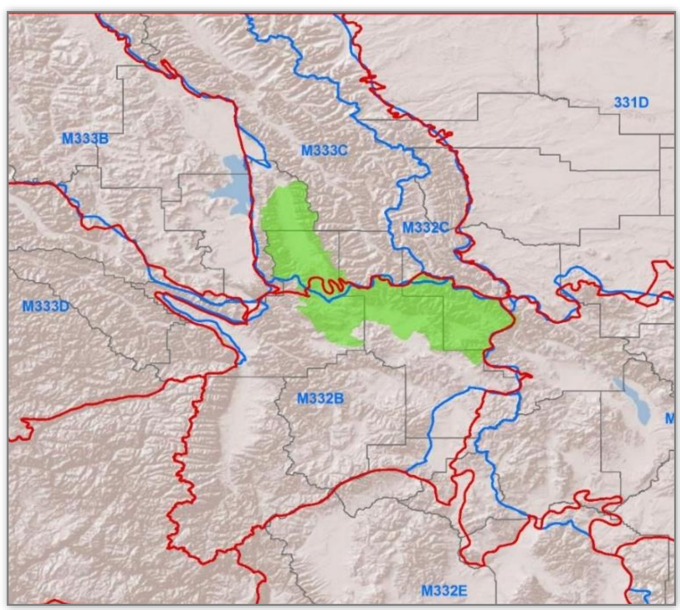


Figure 3. Section-level boundary comparison as developed for ICBEMP by Hessburg et al. 2000 - red lines - and ECOMAP (2007) - blue lines - in relation to the SWCC project area and the surrounding region.

After reviewing existing information on potential vegetation classifications, disturbance processes, and FIA plot data for tree species distribution, the boundaries were selected from each map that represented the best apparent accuracy for ecosystem diversity classification for this project. Further, an additional split was made in ecoregion M332B to address the much reduced occurrence of western larch and increased occurrence of limber pine and Rocky Mountain juniper in the eastern half of section M332B. As described previously, such changes in species distribution within the BSLRP project area affects the resolution of the classification of ecosystem diversity required to delineate specific plant communities. Keeping M332B as one ecoregion for ecosystem classification would have introduced more variability into the classification unless handled in one of two ways: 1) create two landscapes to reflect the change in species distributions, or 2) classify additional ecological sites or ecosystems within a single ecoregion/landscape to capture this change in species composition, structure, and response to disturbance. For the BSLRP project landscape classification, the decision was made to create two ecoregions/landscapes since this species change also corresponded to a jurisdictional change in national forests boundaries. An east and west split was created in M332B where the species change was most evident. A further level of boundary refinement was also accomplished by incorporating watershed boundaries (HUC 10-level) where they were reasonably close to the more generalized ecoregion boundaries. The assumption is this would further reduce the variability in the ecosystem diversity classification by removing slivers of ecosystems falling over into another watershed. An example of this would be the boundary between the east and west-side of the continental divide, where using the watershed boundary to refine the more generalized ecoregion boundary keeps the ecosystem diversity classification and mapping to only the targeted ecosystems occurring west of the divide. Landscape classification methods and data sources used to develop ecoregion boundaries are described in more detail in Appendix B.

Results

Figure 4 depicts the final ecoregion boundaries used as the landscape-level classification for the BSLRP landscape assessment. The SWCC-CFLRP project occurs in 3 ecoregions - the Northern Rockies Ecoregion (M333C) and the Northern Rockies and Bitterroot Valley West and East Ecoregions (M332B-West and M332B-East). Table 1 summarizes the total number of acres in each ecoregion and the number of acres and percent representing the SWCC project area.

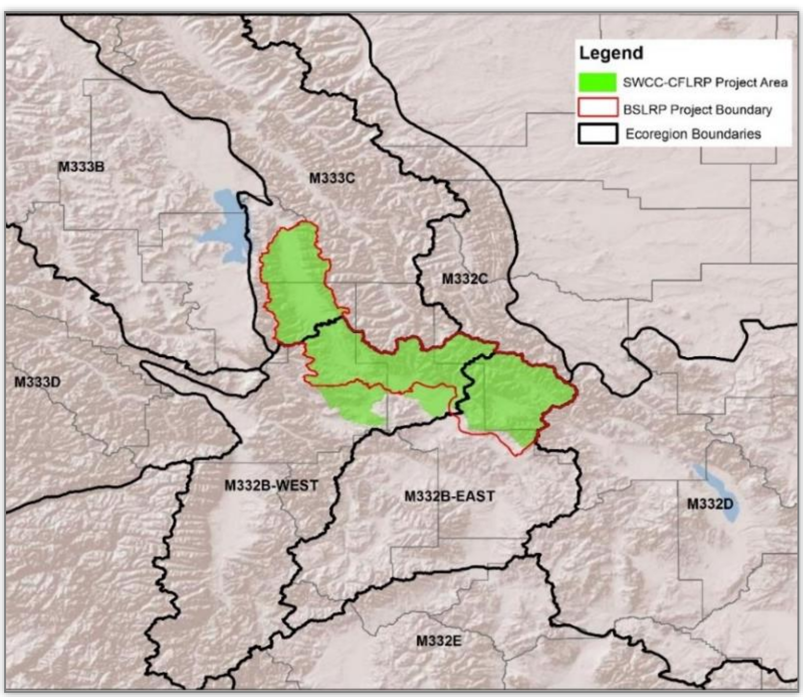


Figure 4. Map of final ecoregions (M333C, M332B-West, M332B-East) used in the landscape assessment.

Table 1. Total number of acres in each of the three ecoregions present in the project region as well as the number of acres in the SWCC landscape assessment area by ecoregion and % of the overall ecoregion represented by the landscape assessment area.

ECOREGION	TOTAL (ACRES)	SWCC Landscape Assessment Area	
		Acres	*% of Ecoregion*
Northern Rockies (M333C)	3,222,042	370,023	11.5
Northern Rockies and Bitterroot Valley –West (M332B-West)	3,050,021	654,910	21.5
Northern Rockies and Bitterroot Valley-East (M332B-East)	2,426,751	396,263	16.3
TOTALS	**8,698,814**	**1,421,196**	**16.3**

Ecosystem Level

The classification of native ecosystem diversity within a delineated ecoregion is dependent on describing and mapping, at an appropriate resolution, 1) the abiotic environment representing different ecological sites, and 2) the biotic associations representing the different disturbance states in response to both disturbance and successional processes. For the purposes of the BSLRP project landscape assessment, we refer to the combined classification of ecological sites and disturbance states as the ecosystem diversity framework. For upland forest ecosystems, ecological sites represent the variation in physical environment components and disturbance states represent the dynamic vegetation communities that can occur on each ecological site in response to natural disturbance regimes. The following sections discuss the importance of considering scale and data resolution when delineating the ecosystem diversity framework for ecological sites and disturbance states. In addition, the methods used to describe and map ecological sites and disturbance states are discussed.

Ecological Sites: The Abiotic Environment

Many physical factors interact to create environmental gradients important to the development of ecosystems. Examples of such physical factors include topography, underlying geology, soils, water, and climate (Hjort et al. 2015, Abella and Denton 2009). Climate determines the amount and timing of precipitation and temperatures. Climate is frequently described and characterized at a macro- or regional-level but there is also micro- or local-level climate that can influence a given site. For example, topography can affect the ability of air to carry moisture or can influence the flow of cool or warm air within a site and thereby influence the climate at a micro-site level. The characteristics of the underlying geology and soils influence the amounts and types of nutrients available to organisms and plants as well as the availability of water. All of these factors interact to influence the distribution of plants along these environmental gradients. Ecological sites help delineate these gradients. Where changes in soil, geomorphic setting, or moisture conditions are abrupt, plant community boundaries can be distinct. Where boundaries are more gradual, plant community change will be less distinct and occur along wider environmental gradients of soils and topography.

Habitat typing (Pfister et al. 1977, Daubenmire 1968) has been the most developed and accepted ecological site classification system for interpreting differences in forest ecosystems in the northern

Rockies. This site classification is based on late successional species conditions that would occur without any disturbances and they incorporate both overstory and understory species. It is not hierarchically arranged or delineated within specific landscapes, but does identify potential vegetation for sites at a fairly fine resolution. Individual habitat types are generally considered to be too fine a resolution for use in forest planning or management. Habitat types are typically aggregated into ecological site groupings such as potential vegetation groups (PVG), vegetation response groups (VRG), biophysical setting (BpS), fire response groups (FRG) or other such categorizations depending on the underlying objectives for use. Such groupings produce a classification at a coarser resolution that is compatible with current mapping and data management capabilities, and is also considered to be sufficient for coarse-filter application in an ecosystem diversity classification. However, this coarser grouping of ecological sites makes it all the more important to carefully delineate ecoregional boundaries at the landscape-level classification to minimize the potential biotic variability that can occur in the ecosystem-level classification if too coarse a resolution is selected.

Methods

The landscape assessment for BSLRP used “R1 Habitat Type Groupings” (upland forest systems only) as identified by Milburn et al. (2015), Region 1 (R1) of the U.S. Forest Service (Appendix C) as the ecological site component of the ecosystem diversity framework. These habitat type groupings were developed based on having “similar productivities, with similarities in historical disturbance regimes that have affected a similar range of tree composition, structural characteristics, and successional trends into mature forests” (Milburn et al. 2015). Region 1 developed a cross-walk between potential vegetation type (PVT) codes used in mapping (USDA Forest Service 2002 and 2004) and the R1 Habitat Type Groups and this crosswalk applied to the 2004 and 2002 (used where the 2004 data were not available) PVT GIS layers to develop a map of ecological sites for the project area. Further, these habitat type groupings were labeled using gradations of biophysical characteristics (i.e., warm-dry, cool-moist) common to the average temperature and precipitation conditions for that site. For the remainder of this document, these groupings will be referred to as upland forest ecological sites. A single upland grass and shrub category was mapped using NRCS Web Soil Survey mapping of grass and shrub ecological sites to delineate the boundary. A single riparian and wetland category was mapped using a combination of ecological sites for riparian types, USFWS National Wetlands Inventory, and PVT riparian and wetland (USDA Forest Service 2002 and 2004) forest and woodland information.

Results

Figure 5 presents a map of the resulting ecological sites for the project area including the 3 ecoregions delineated at the landscape level described previously. The 10 upland forest ecological sites are listed in Table 2 along with their acreages based on their mapped distributions within the 3 ecoregions, while Table 3 lists the amounts of additional coarse categories of land cover across the landscape assessment area. Table 4 lists the distribution of each ecological site by ownership category within each of the 3 ecoregions. Table 5 describes the general characteristics of each of the 10 upland forest ecological sites; note the 10

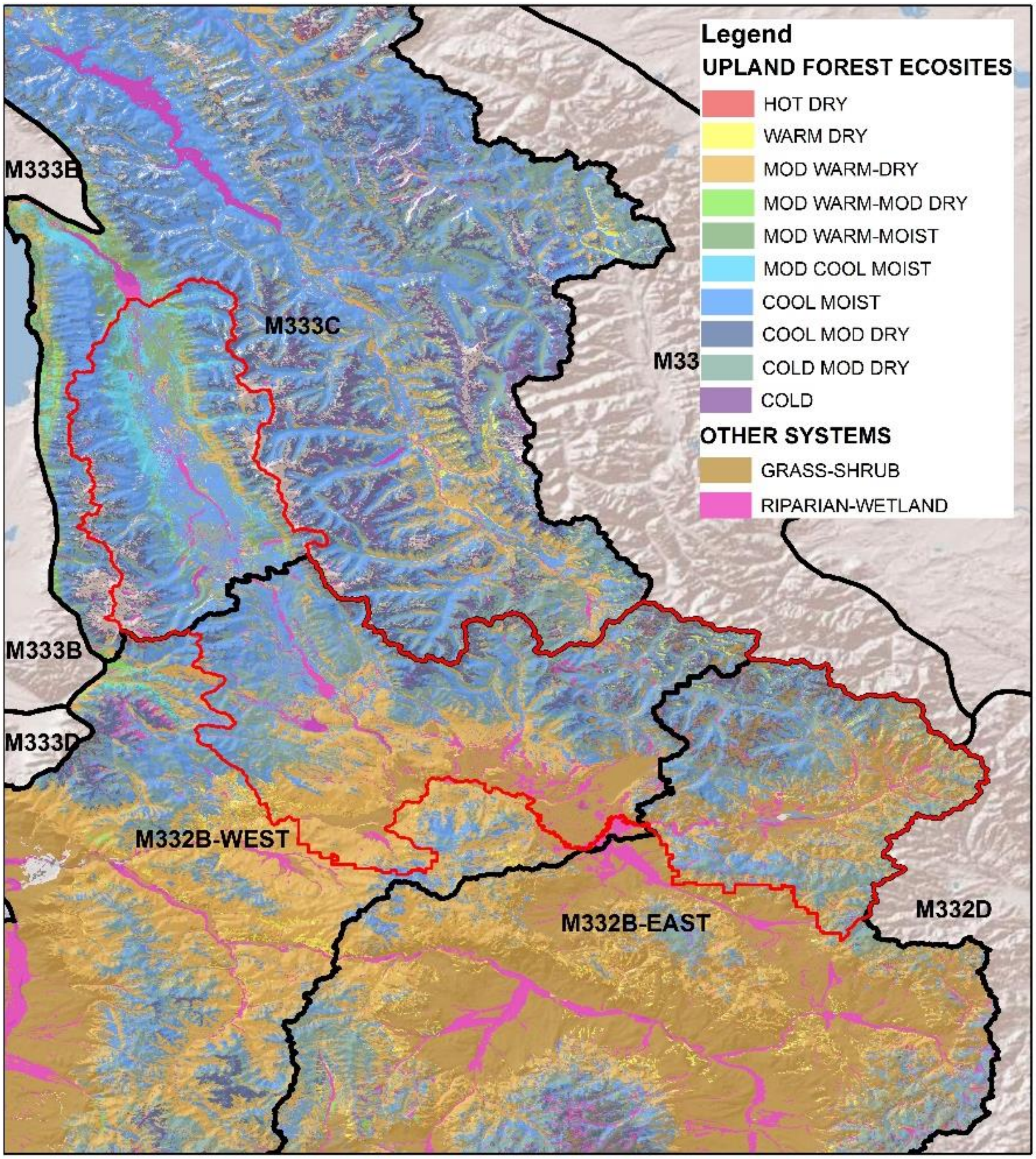


Figure 5. The distribution of 10 upland forest ecological sites as well as groupings of grass-shrub and riparian-wetland ecological sites within the project area and its delineated ecoregions.

ecological sites are not uniformly distributed across each of the 3 ecoregions nor within the landscape assessment area. Figure 6 displays the distribution of ecological sites across the different ecoregions in the project area. Ecological sites totaling less than 500 acres within an individual ecoregion and within the landscape assessment area were omitted from the analysis due to their minimal representation within the project area. These sites may be better represented within other areas of the larger region and should be addressed in other planning initiatives. Grass and shrub areas and riparian areas were each mapped as one category.

Table 2. Upland forested ecological sites identified for the landscape assessment area and their mapped acreages within the three delineated ecoregions including their percentages in the landscape assessment area compared to the overall ecoregions.

ECOLOGICAL SITES BY ECOREGION	ASSESSMENT AREA		ECOREGION	
	ACRES	% of SWCC	ACRES	SWCC %
HOT and DRY	**934**	**0.1%**	**5,949**	**15.7%**
Northern Rockies (M333C)	-	**0%**	1,269	**0.0%**
N. Rockies & Bitterroot Valley - West (M332B-W)	266	**0.02%**	1,019	**26.1%**
N. Rockies & Bitterroot Valley - East (M332B-E)	668	**0.1%**	3,661	**18.2%**
WARM and DRY	**11,055**	**1.0%**	**98,016**	**11.3%**
Northern Rockies (M333C)	89	**0.01%**	16,971	**0.5%**
N. Rockies & Bitterroot Valley - West (M332B-W)	6,047	**0.5%**	45,841	**13.2%**
N. Rockies & Bitterroot Valley - East (M332B-E)	4,919	**0.4%**	35,204	**14.0%**
MODERATELY WARM and DRY	**275,798**	**24.3%**	**1,739,282**	**15.9%**
Northern Rockies (M333C)	23,609	**2.1%**	222,005	**10.6%**
N. Rockies & Bitterroot Valley - West (M332B-W)	172,079	**15.17%**	1,023,076	**16.8%**
N. Rockies & Bitterroot Valley - East (M332B-E)	80,110	**7.1%**	494,201	**16.2%**
MODERATELY WARM and MODERATELY DRY	**3,417**	**0.3%**	**40,365**	**8.5%**
Northern Rockies (M333C)	3,404	**0.3%**	14,032	**24.3%**
N. Rockies & Bitterroot Valley - West (M332B-W)	13	**0.001%**	18,175	**0.1%**
N. Rockies & Bitterroot Valley - East (M332B-E)	-	**0%**	8,158	**0.0%**
MODERATELY WARM and MOIST	**37,284**	**3.3%**	**91,875**	**40.6%**
Northern Rockies (M333C)	37,270	**3.3%**	86,989	**42.8%**
N. Rockies & Bitterroot Valley - West (M332B-W)	14	**0.001%**	4,886	**0.3%**
N. Rockies & Bitterroot Valley - East (M332B-E)	na		na	
MODERATELY COOL and MOIST	**39,153**	**3.5%**	**121,167**	**32.3%**
Northern Rockies (M333C)	39,153	**3.5%**	116,290	**33.7%**
N. Rockies & Bitterroot Valley - West (M332B-W)	-	**0%**	4,820	**0.0%**
N. Rockies & Bitterroot Valley - East (M332B-E)	-	**0%**	57	**0.0%**
COOL and MOIST	**373,760**	**33.0%**	**2,059,488**	**18.1%**
Northern Rockies (M333C)	146,600	**12.9%**	1,345,642	**10.9%**
N. Rockies & Bitterroot Valley - West (M332B-W)	151,228	**13.33%**	464,421	**32.6%**
N. Rockies & Bitterroot Valley - East (M332B-E)	75,932	**6.7%**	249,425	**30.4%**
COOL and MODERATELY DRY	**288,778**	**25.5%**	**1,497,064**	**19.3%**
Northern Rockies (M333C)	45,242	**4.0%**	530,277	**8.5%**
N. Rockies & Bitterroot Valley - West (M332B-W)	144,974	**12.78%**	554,350	**26.2%**
N. Rockies & Bitterroot Valley - East (M332B-E)	98,562	**8.7%**	412,437	**23.9%**
COLD and MODERATELY DRY	**88,038**	**7.8%**	**558,477**	**15.8%**
Northern Rockies (M333C)	30,277	**2.7%**	351,728	**8.6%**
N. Rockies & Bitterroot Valley - West (M332B-W)	36,736	**3.24%**	128,249	**28.6%**
N. Rockies & Bitterroot Valley - East (M332B-E)	21,025	**1.9%**	78,500	**26.8%**
COLD-TIMBERLINE	**15,899**	**1.4%**	**182,360**	**8.7%**
Northern Rockies (M333C)	3,275	**0.3%**	126,878	**2.6%**
N. Rockies & Bitterroot Valley - West (M332B-W)	8,467	**0.75%**	44,340	**19.1%**
N. Rockies & Bitterroot Valley - East (M332B-E)	4,157	**0.4%**	11,142	**37.3%**
TOTAL ACRES	**1,134,116**		**6,394,043**	

Table 3. Total acres and percent of the upland forest, grass-shrub, riparian-wetland, and rock-barren ecological systems within the landscape assessment area.

ECOREGION	UPLAND FOREST		GRASS-SHRUB		RIPARIAN-WETLAND		ROCK-BARREN		TOTAL	
	ACRES	%	ACRES	%	ACRES	%	ACRES	%	ACRES	%
Northern Rockies Section (M333C)	328,936	23.1	5,378	0.4	19,495	1.4	16,214	1.1	370,023	26.0
N. Rockies & Bitterroot Valley Section - West (M332B-W)	519,825	36.6	91,652	6.4	38,650	2.7	4,783	0.3	654,910	46.1
N. Rockies & Bitterroot Valley Section - East (M332B-E)	285,373	20.1	88,456	6.2	16,952	1.2	5,482	0.4	396,263	27.9
TOTAL ACRES	**1,134,134**	**79.8**	**185,486**	**13.1**	**75,097**	**5.3**	**26,479**	**1.9**	**1,421,196**	**100.0**

Table 4. Distribution of ecological sites by land ownership across the three ecoregions in the SWCC landscape.

Ecoregion/Landowner	Ecological Site										TOTAL
	HD	WD	MWD	MWMD	MWM	MCM	CM	CMD	COLD	TIM	
M333C Ecoregion			**23,609**	**3,404**	**37,270**	**39,153**	**146,600**	**45,242**	**30,277**	**3,275**	**328,830**
US Forest Service	-	-	74.8	77.1	43.3	57.4	74.8	80.4	90.7	92.8	71.6
State of Montana	-	-	16.9	18.1	51.4	38.8	13.6	7.3	9.1	7.2	19.8
Private	-	-	8.3	4.5	5.1	3.2	11.3	12.0	-	-	8.2
Other Federal	-	-	-	0.3	0.2	0.6	0.2	0.4	0.3	-	0.4
M332B-WEST Ecoregion		**6,047**	**172,079**				**151,228**	**144,974**	**36,736**	**8,467**	**513,484**
US Forest Service	-	83.8	51.7	-	-	-	73.3	82.1	97.0	100.0	74.0
State of Montana	-	9.3	24.4	-	-	-	14.3	7.7	3.0	0.0	13.1
Clearwater-Blackfoot	-	0.4	10.1	-	-	-	6.7	8.7	0.1	0.0	7.4
Private	-	6.2	13.2	-	-	-	5.0	1.5	0.0	0.0	5.1
Other Federal	-	0.3	0.6	-	-	-	0.7	0.1	0.0	0.0	0.4
M332B-EAST Ecoregion	**668**	**4,919**	**80,110**				**75,932**	**98,562**	**21,025**	**4,157**	**285,373**
US Forest Service	98.2	77.3	80.1	-	-	-	90.2	96.1	99.9	100.0	89.9
Private	0.5	14.0	13.0	-	-	-	5.7	2.7	0.0	0.0	6.3
State of Montana	0.5	7.4	5.2	-	-	-	2.6	1.0	0.0	0.0	2.6
Other Federal	0.8	1.3	1.8	-	-	-	1.5	0.2	0.0	0.0	1.0

Table 5. Upland forested ecological sites within the landscape assessment area relative to their expected distributions, habitat types, and general amounts.

Ecological Site	Description
Hot &Dry (HD)	Some of the driest sites still capable of supporting trees. Frequently adjacent to or intermixed with grass/shrub ecological sites. Relatively rare in the project area occurring only in M332B-E&W. Limber pine habitat types are the most common for this ecological site in the project area and represent only 934 acres and less than 0.1% of the upland forest ecological sites. Trees occurring are often stunted and slow growing.
Warm & Dry (MWD)	The warm and dry extreme of forest environments. While they occur in all three ecoregions, they are more common in the southern half (M332B-E & W) of the project area. They represent 11,055 acres and 1% of the upland forested ecological sites. They occur primarily at low elevations and are often found intermixed where grass-shrub systems transition to forest systems. Ponderosa pine habitat types and dry Douglas-fir habitat types are typical.
Moderately Warm & Dry (MWD)	The moderately warm and dry forest environments. These sites occur most commonly at low to mid-elevations but may also be found at higher elevations on south or west-facing aspects. They may also occur transitional to the grass-shrub systems but require deeper and less droughty soils. They are characterized by more moist Douglas-fir habitat types. They represent 275,798 acres and 24.3% of the upland forested ecological sites
Moderately Warm & Moderately Dry (MWMD)	The moderately warm and moderately dry forest environments occur only in ecoregion M333C where the moderating effect of the Pacific-Maritime climate reaches its eastern and southern limit in the inland Northwest. These sites typically occur as a transition zone between the drier and moister sites, including characteristics of each, and are frequently found on the lower to mid-slope benches and well-drained slopes. They represent the moister Douglas-fir and the moderately moist grand fir habitat types. This is a minor ecological site at only 3,417 acres and 0.3% of the upland forested ecological sites.
Moderately Warm & Moist (MWM)	The moderately warm and moist forest environments. Occurs only in ecoregion M333C due again to the southern and eastern reach of the Pacific-Maritime climate. They are most common on the lower benches and valley bottoms on northerly aspects and consist principally of grand fir habitat types in the project area. This site represents 37,284 acres or 3.3% of the upland forested ecological sites.
Moderately Cool & Moist (MCM)	The moderately cool and moist forest environments. This site is also due to the southern and eastern reach of the Pacific Maritime climate and only occurs in ecoregion M333C. It is characterized by upland cedar and hemlock habitat types and often contains the greatest plant species diversity of ecological sites occurring in the project area. It represents 39,153 acres and 3.5% of the upland forested ecological sites.
Cool & Moist (CM)	The cool and moist forest environments. Most common to the mid-elevation zone and may occur at lower elevations where northwest and east-facing slopes and moist frost pockets are influenced by nightly cold-moist air flow patterns that may compensate for soil moisture. This ecological site is the most common and well distributed in the project area. It represent 373,760 acres and 33% of the upland forested ecological sites.
Cool & Moderately Dry to Moist (CMD)	The cooler and drier forest environments. More common to ecoregions M332B-E & W, but occurs in small amounts in M333C as well. Found primarily in mid-elevation zones but at its lower limit may occur on steep, north/east aspects but shift to south/west aspects at upper limits. At low elevations, often influenced by cold air drainage and inter-fingered with warmer sites in response to topography. It is represented by subalpine-fir habitat types and is the second most common at 288,778 ac and 25.5% of the upland forest ecological sites.
Cold (COLD)	The cold and dry forest environment at the upper elevation zone where it often transitions to Krummholz or alpine communities. The extreme climate contributes to a short growing season and early summer frosts. Soils are shallow with limited soil moisture. Habitat types include cold sub-alpine fir, mountain hemlock (M333C only), and persistent lodgepole pine. This site encompasses 88,038 acres and 7.8% of the upland forested ecological sites.
Timberline (TIM)	The high elevation cold, harsh sites at timberline. Habitat types are characterized by alpine larch and whitebark pine. This site represents 15,899 acres and 1.4% of the upland forest ecological sites in the project area.

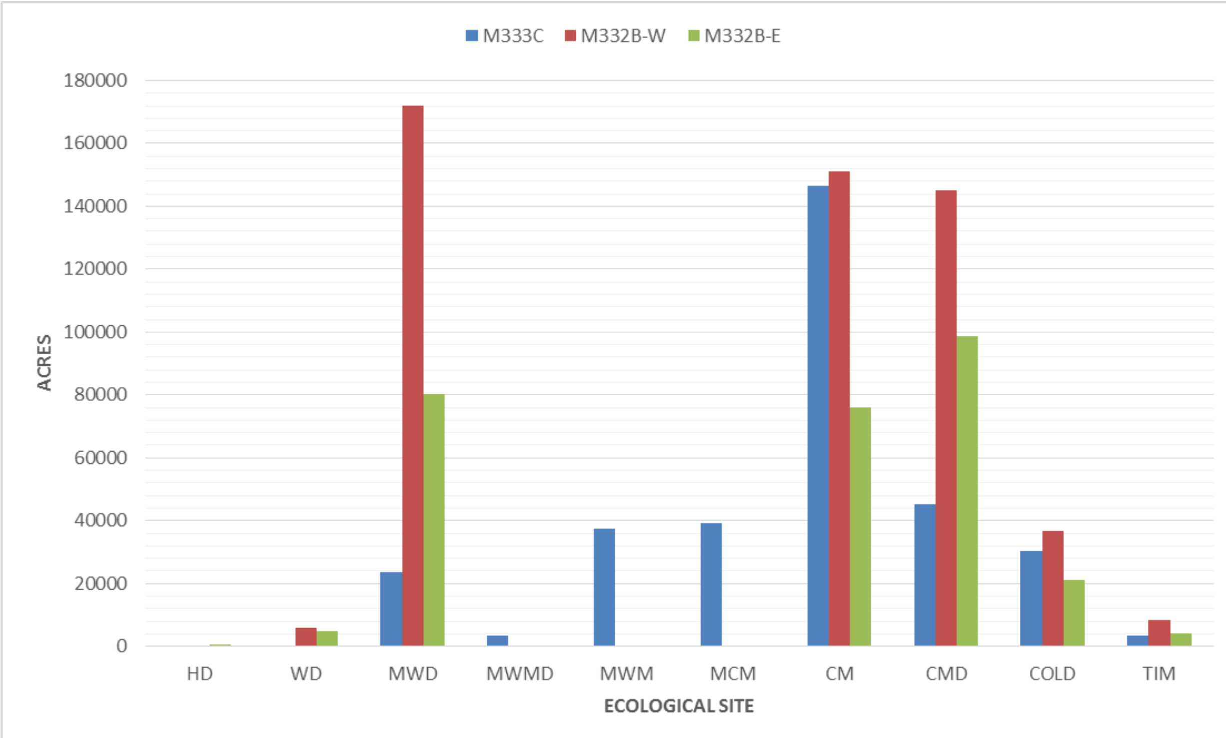


Figure 6. The distribution and number of acres of ecological sites occurring across each of the 3 ecoregions.

Disturbance States

Historical disturbance is responsible for the dynamic landscape and ecosystem-level processes that are important drivers of structure and vegetation patterns occurring across a wide range of spatial and temporal scales. Further, on any given ecological site, successional processes are also influencing the species composition and structure of the plant community over time. The associated animal community changes in response to changes in the plant community caused by succession. Disturbances serve to disrupt successional processes and influence the specific biotic community occurring on an ecological site at any specific time. The ability to identify and describe reference conditions for upland forest ecosystem diversity will be greatly informed by understanding the interaction and influence of successional and disturbance processes on ecosystem composition, structure, and function. In many cases, recognizable patterns are known or will emerge which allow us to describe and predict a plant community's response on a given ecological site to the frequency or intensity of a disturbance type.

Methods

For the purposes of ecosystem diversity, we use the term disturbance state to refer to an ecosystem (e.g., plant community and its associated animal community) that could occur on a specific ecological site in response to successional and disturbance processes. As discussed previously, ecological sites provide valuable information on the interaction of the physical environment with vegetation but they must be combined with a classification of disturbance states to identify the full range of conditions or ecosystem diversity possible, as influenced by historical disturbance processes.

For forest management planning, the number of disturbance states described for an ecological site requires balancing the potential combinations of species composition, structure, and functions that support the representation objectives of the coarse-filter conservation strategy with the logistical reality of acquiring and managing information and data. State and transition models (STM) are a common tool used to describe and illustrate the range of disturbance states that can occur on a particular ecological site and how an existing disturbance state can transition to another disturbance state in response to disturbance events, or lack thereof. STMs help to display patterns and mechanisms of vegetation response to identified successional or disturbance processes by identifying the triggers and drivers of transition among states (Bestelmeyer et al. 2009, Henderson 2008). They also document the current knowledge of disturbance states while allowing for future adjustment as new information becomes available. Transitions can occur rapidly such as in the event of a high severity fire event or more slowly such as through general successional progression. Some disturbance states can be relatively persistent over time even with frequent low severity disturbance such as low severity fire events.

The development of STM's should be based on the best information available on plant species and community response to disturbance, with recognition this information can sometimes be subjective and based on expert opinion. It may be impossible to definitively establish quantitative reference information on many historical states that simply do not exist today because of changes to historical disturbance processes or conditions. The goal still remains, however, to use the best information available and identify the known limitations of existing information. Defining historical reference conditions is an essential component of the coarse-filter conservation strategy for biodiversity. These limitations do not detract from their immediate usefulness in efforts to describe native ecosystem diversity with recognition that future research is needed to acquire additional data to support and strengthen their use.

Results

Twelve disturbance states defined by this project were used to reflect the range of upland forest conditions occurring on each of the 10 ecological sites to meet the objectives for describing ecosystem diversity (Table 6). Table 5 also identifies the classification criteria for differentiating the different states that were developed in consultation with silviculturists and wildlife ecologists representing the BSLRP team, Region 1, the Lolo, Helena and Flathead Forests, and the Lincoln, Seeley Lake and Swan Lake Ranger Districts. When combined with ecological sites, these disturbance states reflect the important differences in species composition, structure, and function in response to the complex interaction of successional progression over time and the severity of disturbance events. Specifically, structural diversity was defined using the combination of 5 size classes based on largest tree cohort and 3 canopy cover classes in response to succession and disturbance processes. Figure 7 depicts the different canopy classes for the very large tree size class for one example ecological site. The structural diversity classification developed to describe and quantify ecosystem diversity was consistent among the 10 ecological sites, although the proportions of the different states that occurred historically across the different ecological sites were expected to show differences.

Table 6. Structural characteristics used to define the successional progression, (e.g., GRASS-FORB-SHRUB-SEEDLING, SAPLING-SMALL TREE, etc.) and disturbance influences on canopy cover (e.g., OPEN, MODERATELY OPEN, CLOSED) and species composition (fire adapted vs. fire intolerant) used in the state and transition model.

SIZE CLASS (BASED ON LARGEST COHORT)			Canopy Cover Class (Tree Component Only)		
			OPEN (10-39%)	MODERATELY OPEN (40-59%)	CLOSED (≥60%)
GRASS-FORB-SHRUB-SEEDLING			DS 1		
	DBH Range	<1.0”			
	Avg. Age	<15 yrs.			
SAPLING-SMALL TREE			DS2		DS3
	DBH Range	1.0”– 4.9”			
	Avg. Age	40 to 60 yrs.			
MEDIUM TREE			DS4	DS5	DS6
	DBH Range	5.0” – 14.9”			
	Avg. Age	80 to 100 yrs.			
LARGE TREE			DS7	DS8	DS9
	DBH Range	15.0” – 19.9”			
	Avg. Age	120 to 180 yrs.			
VERY LARGE TREE			DS10	DS11	DS12
	DBH Range	≥20.0”			
	Avg. Age	>200 yrs.			

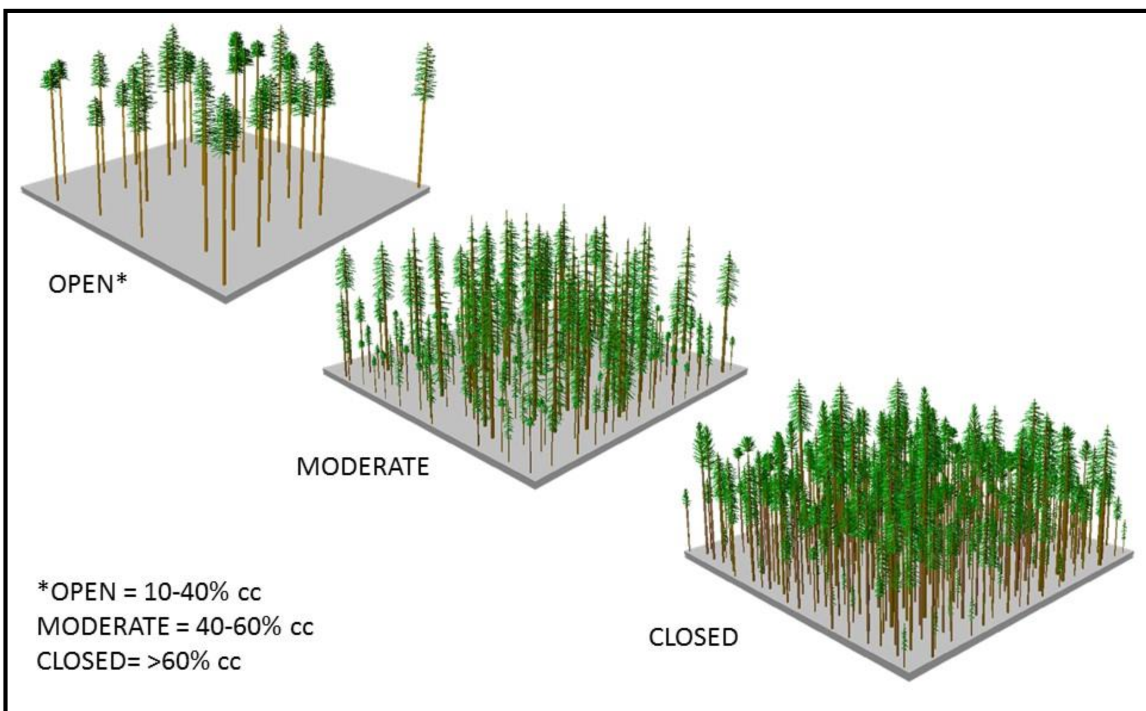


Figure 7. An example of FIA plot data (3 different plots) used to display the very large tree size-class in the open, moderate, and closed canopy cover classes for the moderately warm and dry ecological site. The FIA plot data are displayed using the Stand Visualization System (McGaughey 2004).

A state and transition model (STM) was developed to support the ecosystem diversity framework. It represents a relatively simple printed flowchart identifying the range of disturbance states that can occur on an ecological site and the disturbance processes influencing the transition from one state to another. Transitions can occur rapidly such as in the event of a fire or more slowly such as general successional progression. Sometimes multiple disturbance changes must occur simultaneously to trigger a transition to a different state.

A state and transition model (STM) was developed for upland forest systems using 12 disturbance states identified to represent and appropriate resolution to meet the objectives of the conservation strategy (Figure 8).

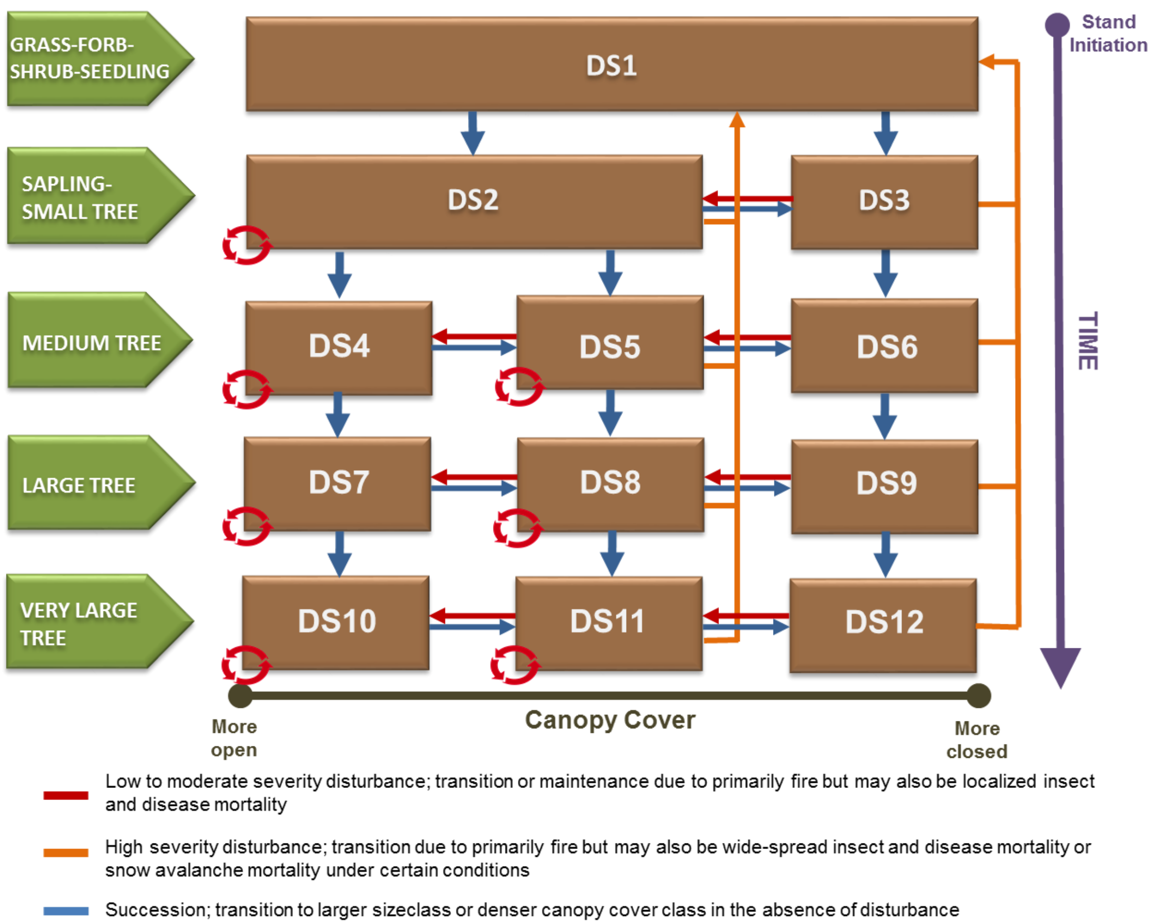


Figure 8. State and transition model for forest disturbance states used in the terrestrial landscape assessment. Twelve disturbance states are delineated based on the structural response to disturbance in terms of tree size and canopy cover.

The following sections provide a general description of each of the 12 disturbance states identified in the STM as influenced by the primary disturbance processes occurring in the project area.

Disturbance State 1 (DS1)

DS1 represents the grass-forb-shrub-seedling conditions resulting from a high severity disturbance event that has killed all or most of the existing tree component. Some forest ecologists refer to this post-disturbance period as the stand initiation stage (Franklin et al 2002) with the expectation that trees begin to re-establish on these sites in a relatively short-time frame (<15 years on average). The conditions immediately following a disturbance event are dependent on both the pre-disturbance conditions and the type and severity of the disturbance. The early post-disturbance live component of this state is represented by a combination of grass, forb, and shrub species as well as varying amounts of tree seedlings depending on the disturbance type, ecological site, and pre-disturbance conditions. DS1 is typically characterized by a low to moderate canopy cover that may infrequently transition to high canopy cover over time on some ecological sites.

Following a high severity fire event, fire adapted plant species will recolonize these sites to take advantage of any previously limited resources, such as light, moisture, and nutrients (Swanson et al. 2010). For a given ecological site, the recolonizing vegetation can take multiple successional pathways with different species compositions and structures. More disturbance adapted species may utilize evolutionary strategies such as suckering, long-lived seed banks, wind-blown seed distribution, higher growth rates, etc. to facilitate an advantage over other species post-disturbance. Many disturbance adapted species are also shade-intolerant species that will take advantage of the open conditions resulting from disturbance.

The trees and understory present at the time of a disturbance can be totally consumed, such as by extreme fire events, or the trees may be killed but remain standing, leaving behind important legacy structures such as standing dead trees, or later, down trees that then become coarse woody debris on the ground surface, and all may be used as habitat by various organisms and wildlife species (Swanson et al. 2010).

Following a high severity insect or disease event, all or most of the tree component is killed usually resulting in a complex structure of standing dead, dead-fall, and down dead trees. The understory vegetation remains unaffected by the actual disturbance event but will slowly shift composition in response to the changing opportunities for increased light, moisture, and nutrient availability. These sites are also frequently more susceptible to a moderate to high severity fire event due to the high fuel loading resulting from the dead tree structures. The complexity of these legacy structures is determined by the pre-disturbance condition and, depending on the tree species or decomposition rate for a particular ecological site, can persist for decades.

On some mid- to high-elevation ecological sites with vulnerable topography, snow avalanches can remove or severely damage the existing tree component on 10 to 100’s of acres. The frequency of snow avalanches can vary from multiple avalanches in a single season (Laternser and Schneebeli 2002) to centuries between avalanche events (Bebi et al. 2009). The variability in frequency leads to variability in post-avalanche conditions ranging from persistent shrub conditions on frequent avalanche sites to a late seral forest condition that is completely or mostly removed by an infrequent snow avalanche event (Bebi et al. 2009).

Disturbance State 2 (DS2)

DS2 represents the open to moderately-open sapling to small tree forest conditions resulting from successional progression from DS1. These open to moderately open conditions may be maintained or further influenced by relatively frequent low to moderate severity fire events that reduce the number of trees present and promote the survival of fire adapted tree species over fire intolerant species.

Disturbance State 3 (DS3)

DS3 represents the closed sapling to small tree forest conditions resulting from successional progression from DS1 or tree establishment in DS2 that leads to a more closed canopy. Species composition and structure are influenced by the absence of disturbance which typically benefits fire intolerant species and maintains a closed canopy condition. Disturbance events are rare in this state but when they do occur they will move the ecosystem to a new state such as DS1 in the case of a high severity event or DS2 in the case of a moderate severity event that thins the overstory.

Disturbance State 4 (DS4)

DS4 represents the open medium tree forest conditions resulting from successional progression from DS2. These open forest conditions may be maintained by frequent low to moderate severity fire events that may kill most of the fire intolerant trees present and thereby promote the survival of fire adapted tree species in both the overstory and understory. Insects and disease events are typically isolated to small clumps of trees where it does occur and further thin the canopy. A high severity event would be unlikely to occur in this disturbance state due to the low density of trees but under extreme weather conditions could move this state back to DS1.

Disturbance State 5 (DS5)

DS5 represents the moderately open medium tree forest conditions resulting from successional progression from DS2 or additional tree establishment in DS4 that moves the canopy to a moderately open condition. These forest conditions may be maintained as DS5, or may transition back to DS4, by recurring low to moderate severity disturbance events that kill some of the trees present. The surviving trees are characterized by primarily fire adapted or less shade tolerant species in the largest sizeclass and a mix of fire adapted and fire intolerant species in the smaller size classes. Insects and disease would be a moderate influence in this state especially where denser tree patches occur. A high severity fire event would be uncommon to this state but if it occurred, would transition the stand back to DS1.

Disturbance State 6 (DS6)

DS6 represents the closed medium tree forest conditions resulting from successional progression from DS3 or additional tree establishment in DS5 that leads to a closed canopy. Species composition and structure are maintained or influenced by the absence of disturbance which typically benefits fire intolerant species and maintains a closed canopy condition. Disturbance events are rare in this state but when they do occur they will move the ecosystem to a new state such as DS1 in the case of a high severity event or DS4 or 5 in the case of a moderate severity event.

Disturbance State 7 (DS7)

DS7 represents the open large tree forest conditions resulting from successional progression from DS4. These open forest conditions may be maintained by frequent low to moderate severity fire events that may kill most of the fire intolerant trees present and thereby promote the survival of fire adapted tree species in both the overstory and understory. Insects and disease events are typically isolated to small clumps of trees where it does occur. A widespread high severity fire event would be unlikely to occur in this disturbance state due to the low density of trees.

Disturbance State 8 (DS8)

DS8 represents the moderately open large tree forest conditions resulting from successional progression from DS5 or additional tree establishment in DS7 that pushes the canopy to a moderately open condition. These forest conditions may be maintained as DS8, or may transition back to DS7, by recurring low to moderate severity fire events that kill some of the trees present. The surviving trees are characterized by primarily fire adapted species in the largest size class and a mix of fire adapted and fire intolerant in the smaller size classes. Insects and disease would be a moderate influence in this state especially where denser tree patches occur. A widespread high severity fire event would be uncommon in this state but if it occurred, would transition the stand back to DS1.

Disturbance State 9 (DS9)

DS9 represents the closed large tree forest conditions resulting from successional progression from DS6 or additional tree establishment in DS8 that leads to a closed canopy. Species composition and structure are maintained or influenced by the absence of disturbance which typically benefits fire intolerant and shade tolerant species and maintains a closed canopy condition. Disturbance events are rare in this state but when they do occur they will move the ecosystem to a new state such as DS1 in the case of a widespread high severity event, or DS7 or 8 in the case of a moderate severity event.

Disturbance State 10 (DS10)

DS10 represents the open very large tree forest conditions resulting from successional progression from DS7. These open forest conditions may be maintained by frequent low to moderate severity fire events that may kill most of the fire intolerant trees present and thereby promote the survival of fire adapted tree species in both the overstory and understory. Insects and disease events are typically isolated to small clumps of trees where it does occur. A widespread high severity fire event that killed more than 75% of overstory trees would be unlikely to occur in this disturbance state due to the low density of trees but if it did occur, would transition the state back to DS1.

Disturbance State 11 (DS11)

DS11 represents the moderately open large tree forest conditions resulting from successional progression from DS8 or additional tree establishment in DS10 that pushes the canopy to a moderately open condition. These forest conditions may be maintained as DS11 or may transition back to DS10 by recurring low to moderate severity fire events that kill some of the trees present. The surviving trees are characterized by primarily fire adapted species in the largest size class and a mix of fire adapted and fire intolerant in the smaller size classes. Insects and disease would be a moderate influence in this state especially where denser tree patches occur. A widespread high severity fire event would be uncommon to in this state but would transition the stand back to DS1.

Disturbance State 12 (DS12)

DS12 represents the closed very large tree forest conditions resulting from successional progression from DS9 or additional tree establishment in DS11 that leads to a closed canopy. Species composition and structure are maintained or influenced by the absence of disturbance which typically benefits fire intolerant and shade tolerant species and maintains a closed canopy condition. Disturbance events are rare in this state but when they do occur they will move the ecosystem to a new state such as DS1 in the case of a widespread high severity event or DS10 or 11 in the case of a moderate severity event.

Ecosystem Diversity Framework

Ecosystem diversity is classified for the purposes of this landscape assessment using the combination of ecological sites and disturbance states, as described in previous sections, and is presented in a tool termed the ecosystem diversity framework. The ecosystem diversity framework also represents the coarse-filter for a targeted landscape/ecoregion and will be used throughout the remainder of this assessmemt to describe and summarize many of the results. Three separate ecosystem diversity frameworks are required to present the results of each of three ecoregions. As an example, Figure 9 represents the ecosystem diversity framework for ecoregion M332B-East. While there is overlap between the ecological site labels within each ecoregion, it is important to note sufficient environmental differences occur on these sites to produce changes in species assemblages and distribution among and between the ecoregions. Note also that each “cell” in the framework represents the combination of an ecological site and disturbance state and thereby represents an individual ecosystem. As such, each ecoregion is represented by a different number of ecosystems and overall ecosystem diversity with Ecoregion M333C having 96 ecosystems identified and M332B-West and M332B-East having 72 and 84 ecosystems identified, respectively.

TREE SIZECLASS	HOT-DRY			WARM-DRY			MOD WARM-DRY			COOL-MOIST			COOL-MOD DRY			COLD			TIMBERLINE		
	Canopy Cover[a]			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	1			1			1			1			1			1			1		
SAPLING - SMALL TREE	2		3	2		3	2		3	2		3	2		3	2		3	2		3
MEDIUM TREE	4	5	6	4	5	6	4	5	6	4	5	6	4	5	6	4	5	6	4	5	6
LARGE TREE	7	8	9	7	8	9	7	8	9	7	8	9	7	8	9	7	8	9	7	8	9
VERY LARGE TREE	10	11	12	10	11	12	10	11	12	10	11	12	10	11	12	10	11	12	10	11	12

Figure 9. An example of the ecosystem diversity framework developed for upland forest systems in ecoregion M332B-East. Columns represent the 7 ecological sites occurring in this ecoregion and the “cells” of the framework represent the 12 disturbance states for each ecological site as delineated using forest structural characteristics for size class and canopy cover. All 84 ecological site x disturbance state combinations represent upland forest ecosystem diversity occurring in ecoregion M332B-East with each cell considered an individual ecosystem.

Native Ecosystem Diversity – Reference Conditions

As discussed in the previous sections, forests in the northern Rockies have changed in many ways since Euro-American settlement. However, describing the regionally specific changes to native ecosystem diversity is a key component of a landscape assessment where the stated objective is to implement a conservation strategy for biodiversity and ecological sustainability. Historical references are used in this regard to help identify, describe and quantify the native ecosystem diversity that occurred in a given area (Egan and Howell 2001, McAlpine et al. 2016). Identifying clear restoration goals is essential for achieving and evaluating project outcomes (McAlpine et al. 2016). For the purpose of this assessment, an historical reference is defined as the native ecosystem diversity resulting from both historical disturbance (i.e., fire, grazing, etc.) and human-influenced disturbance (i.e., Native American). It is based on the assumption that native species, both plant and animal, evolved within a limited range of conditions resulting from these natural and human-influenced disturbance regimes and processes operating in that landscape (Holling 1973, Swanson et al. 1993, Landres et al. 1999). To evaluate changes to native ecosystem diversity and the resulting habitat conditions for species, historical references are usually confined to a period less than 1000 years prior to Euro-American settlement, as these reflect the habitat conditions most relevant to species present today (Haufler et al. 2002, Morgan et al. 1994).

For each ecoregion within the project area it is important to understand and quantify, to the extent possible, the range of variability of each historically occurring ecosystem. Native ecosystems were not static during any defined reference period. However, developing an understanding of the ecosystem diversity that occurred during an identified timeframe prior to Euro-American settlement provides critical reference information for defining the ecological integrity for an ecosystem and is critical information for developing and describing ecological restoration objectives for stand and landscape scales. Historical range of variability (used the same as natural range of variation) is an important concept because it emphasizes that many ecosystems varied in amounts, compositions, and structures due to the interaction of site characteristics, climate, and weather events (such as lightning and wind) that influenced historical disturbance effects (Graham et al. 2004, Aplet and Keeton 1999, Haufler et al. 1999).

To meet the objectives of the conservation strategy, reference conditions should be developed to provide a description of each native ecosystem identified in the ecosystem diversity framework using the best available information. For each ecosystem, the objective is to provide a description of its key characteristics. Key characteristics identified for upland forests of the landscape assessment area include:

- Dominant species composition in terms of the plant species comprising the overstory and understory vegetation,
- Vertical and horizontal vegetation structure descriptions and measures such as tree size classes, canopy cover, density, down woody debris, the number and sizes of snags, and
- Information on the disturbance regimes that produced these conditions.

For many of the ecosystems within the project area, information exists that can be used to inform the description of reference conditions. For example, forest ecosystems that had very long historical fire return intervals may still be within the range of conditions that occurred historically, and can serve as their own reference sites (Clewell and Aronson 2013). However, some ecosystems have had substantial change from historical conditions due to past anthropogenic activities such as logging, grazing by livestock, fire exclusion, or invasion by exotic species. These ecosystems may now be considered poor examples of historical conditions and not recommended as reference conditions. Instead, a reference model would need to be developed and approximated from the best available sources of information (Egan and Howell 2001). The following sections describe the methods used to develop reference conditions for historical disturbance regimes and native ecosystem diversity.

Disturbance Regimes and Severity

Historical disturbance regimes are the patterns of disturbance frequency and intensity that can be quantified using ecological evidence.

Fire

The term "fire regime" is often used to describe the different ways fire interacts with the land to influence the structure and species composition of vegetation, and characterizes the pattern of fire severity occurring across a specific area in the landscape (LANDFIRE, Kaufmann et al. 2007, Agee 1993). Fire regimes are useful for describing the various roles of fire in different ecosystems, and for describing changes in fire from historical conditions. The terms "fire severity" or "fire intensity" are used to refer to the degree of impact fire has on vegetation species composition and structure, and is frequently described using the degree of overstory tree mortality (http://www.firescience.gov/projects/09-2-01-9/supdocs/09-2-01-9_Chapter_3_Fire_Regimes.pdf). Keane et al. (2012) discussed how the term fire severity can have multiple interpretations, so its definition should be clearly identified when it is used in descriptions of fire regimes. Fire regimes generally incorporate the various levels of fire severity across similar sites and their effects on the dominant vegetation, although as noted by Keane et al. (2012), different interpretations of fire severity can result from comparisons of post-fire impacts on overstory or understory vegetation, soils, or other ecosystem components. In this assessment, we use the term fire severity to describe the extent of overstory tree mortality produced by a fire.

While the factors influencing fire severity are complex (Schoennagel et al. 2004), some factors can have more influence on fire severity than others including climate, weather events, biophysical settings, and vegetation conditions (Graham et al. 2004). Birch et al. (2015) evaluated environmental drivers of fire severity across recent fires in central Idaho and western Montana using multiple variables related to the burned sites. These variables included various vegetation conditions, topography (slope and aspect), fire danger indices, and daily weather and found the percent existing vegetation cover had the greatest influence on burn severity by a wide margin, followed by topography and biophysical setting. Weather variables (at 30-m resolution) like fuel moisture, relative humidity, and wind speed were also influential but considered somewhat less important than current conditions and site variables. They suggested that while climate and weather strongly influence fire extent, factors such as current vegetation, topography, and biophysical setting have the most influence on fire severity.

Fire frequency is dependent on many variables (Schoennagel et al. 2004), but in general, more frequent fires occur on warmer and drier sites and less frequent fires occur on cooler and moister sites (Merschel et al. 2014, Agee 1993). Similarly, larger burn patches are expected to occur under dry conditions and smaller burn patches occur under moist conditions (Stine et al. 2014). Fire ecologists frequently describe the effects of fire on forest ecosystems using three broad classes: non-lethal or low severity, mixed-severity, and lethal, stand-replacing or high severity (Hessburg et al. 2016, Heyerdahl et al. 2012, Amoroso et al. 2011, Kaufmann et al. 2007, Agee 2004, Hessburg and Agee 2003, Arno et al. 2000, Morgan et al. 1996, Barrett et al. 1991). Non-lethal fire regimes are more often associated with low to moderate elevation warmer and drier sites, mixed-severity fire regimes are more often associated with mid- to high elevation warmer and moister sites as well as cooler and drier sites, and lethal fire regimes are more often associated with mid-to high elevation cooler and moister sites (Marcoux et al. 2015, Stine et. al. 2014, Merschel 2014, Kaufmann et al. 2007). Sites that are influenced by the non-lethal and mixed-severity fire regimes are also frequently less steep than those sites influenced by the lethal fire regime (Lecina-Diaz et al. 2014), except where rock formations or patchy vegetation may actually slow the spread of fire and contribute to mixed-severity conditions such as at very high elevations. While these site characteristics are the more common drivers of fire regimes, additional site influences such as juxtaposition to adjacent fire regimes can create exceptions to these general rules.

The non-lethal fire regime is usually described as having relatively frequent, low to moderate severity fires that burn along the surface of the ground and remain within the forest understory, thereby being relatively non-lethal to the older trees in the overstory (http://www.firescience.gov/projects/09-2-01-9/supdocs/09-2-01-9_Chapter_3_Fire_Regimes.pdf, Kaufmann et al. 2007, Agee 2000). Mean fire return intervals for non-lethal fire regimes are usually less than 25 years for forests in the western United States (Kaufmann et al. 2007, Fischer and Bradley 1987, Kilgore 1981). The frequency of these fires influences both the species composition and vegetation structure within these forests. Fire-adapted tree species become dominant in the overstory. Under drought conditions, fires can occur over larger areas but still are unlikely to kill a high percentage of overstory trees. However, as reported by Parker et al. (2006), trees that are stressed have higher mortality rates post fire than those that aren't, and drought may also allow fires to damage roots (Kaufmann et al. 2007), which can further increase tree mortalities due to increased susceptibility to various diseases. However, the low density of trees maintained by frequent fires reduces the risk of severe insect and disease outbreaks (Parker et al. 2006, Keane et al. 2002, Hessburg et al. 2000, Kolb et al. 1998, Hessburg et al, 1994, Veblen et al. 1994, Anderson et al. 1987). Frequent fires help limit the occurrence of shade tolerant species such as Douglas-fir which are susceptible to root rot (Parker et al. 2006). The non-lethal fire regime contributes to the persistence of a multi-age stand, which in some cases may be composed of patches of even-aged groups. A wide range of age classes can occur, from saplings to old growth trees, but with relatively low numbers of trees per acre. However, when viewed at the stand level, forests influenced by a non-lethal regime typically have a clear presence of larger, older, fire-adapted trees in the overstory, even if their numbers are relatively low per acre (i.e., 8 to 30 tpa) (Kaufmann et al. 2007, Arno et al. 1997). For this reason, historical references to these forests often describe them as relatively "open and park-like". Stand history studies conducted within forests historically influenced by the non-lethal fire regime indicate they had relatively predictable species composition and structure (Smith and Fischer 1997) as this fire regime appears to act as an agent of ecosystem stability. The result is a fairly uniform forest pattern at both the landscape (i.e., 1000's of acres) and stand levels (i.e., 2-50 acres), though small inclusions of moderate or even high severity fire likely occurred (Kaufmann et al. 2007).

The lethal fire regime is characterized by infrequent, high-severity fire that consumes most of the forest understory and overstory as it moves across the landscape (http://www.firescience.gov/projects/09-2-01-9/supdocs/09-2-01-9_Chapter_3_Fire_Regimes.pdf). Lethal fire regimes result in a stand-replacing effect on forest conditions, in stark contrast to the persistent, yet less obvious effects of the non-lethal fire regime. The result is to set the forest back to an early seral stage and release fire-dependent species stimulated by severe fire events, such as lodgepole pine. Mean fire return intervals for the lethal fire regime are frequently described as greater than 100 years for forests in the western United States (Kaufmann et al. 2007, Agee 2004, Agee 1998). If undisturbed by new fires, the forest then proceeds along a normal successional trajectory for many years. Tree densities are typically high relative to the ecological site and early seral conditions are usually dominated by single age-classes. Tree species that are susceptible to fire can be a common component of the forest, particularly at late seral stages. Due to the higher densities of trees, the potential for insect and disease events is increased (Parker et al. 2006). The resulting forest patterns are large patches of variable age-classes and seral stages at the landscape level but relatively uniform age-classes and conditions at the stand level, though small inclusions of low to moderate severity fires likely occurred.

The mixed severity fire regime produces highly diverse forest conditions with elements of the non-lethal and lethal fire regimes occurring at a finer scale along with greater amounts of moderate severity fire (Hessburg et al. 2016). It is frequently described as having a complex mosaic of varying patch-sizes of low, moderate, and high severity fire effects. Some of these patches underburn as with a low severity fire and some have their overstory tree canopy mostly or completely killed, as with a high severity fire. Other areas may burn with moderate intensity fires that kill many trees, but maintain many of the more fire resistant trees (Kaufmann et al. 2007) such as western larch (Ayres 1900). Marcoux et al. (2015) examined mixed severity and high severity fire sites in southeastern British Columbia and found that western larch only occurred in mixed severity sites that had a history of low to moderate severity fire. At higher elevations where high severity fires occurred, sites were dominated by lodgepole pine with subalpine fir coming in after 250 years post-fire. They also reported that past harvest and fire suppression have homogenized forest structures in the area they studied. They reported the presence of western larch as well as "veteran" trees were indicators of a past mixed severity fire regime, but past harvests and fire suppression may make field identification of mixed severity locations challenging. Heyerdahl et al. (2012) characterized fire severities in an extensive system of plots placed in ponderosa pine/Douglas-fir forests in southeastern British Columbia. They found most of the plots exhibited a mixed severity fire regime dominated by low severity fires with fire scar intervals averaging 21 years, and with only 10% of plots exhibiting high severity fire over the past several centuries. Hessburg et al. (2016), Larson and Churchill (2012), and Kaufmann et al. (2007) discussed the spatial patterns produced by historical fire regimes and noted the variability in patterns from stands to larger areas.

Within areas influenced by the mixed-severity fire regime, the amount of low, moderate, or high severity burn is typically dependent on the specific location (Korb et al. 2013) as well as the weather conditions at the time of a fire. Warmer and drier sites exhibit a higher percentage of low severity fire conditions while cooler and moister sites would exhibit a higher percentage of high severity fire conditions (Agee 1993). Mean fire return intervals for mixed-severity fire regimes are frequently described as ranging from 25 to

100 years for forests of the western United States (Hessburg et al. 2016, Kaufmann et al. 2007, Arno et al. 2000, Agee 1998), but may extend to 200 years on some sites (http://www.firescience.gov/projects/09-2-01-9/supdocs/09-2-01-9_Chapter_3_Fire_Regimes.pdf). The potential for insect or disease events are variable depending on tree densities. The resulting forest patterns are variable at both the landscape and stand levels.

A fire regime classification system that is based on fire effects attempts to incorporate the physical attributes of the site and fire as well as the fire tolerance of the vegetation (Agee 1998). While recognizing that fire severities, and thereby fire regimes occur along an environmental gradient and may not be stable over space and time (Agee 1998), a classification system can help to communicate and quantify the potential influences of different fire regimes on a landscape.

Defining types of fire regimes and how they are being evaluated as well as the scale of the analysis area can influence the type of fire regime assigned to an area or ecological site, as described previously. How a landscape is classified will also have significant influence on how fire regimes are described or quantified (Merschel et al. 2014, Marcoux et al. 2013). Hermoso et al. (2012) reported on using vegetation classes to describe ecosystem diversity and Abella and Denton (2009) described finer scale classifications for characterizing differences in ponderosa pine stand conditions, the number (and type) of vegetation or biophysical classes used to describe fire regimes will influence the ability to identify finer delineations for specific locations. For example, some studies classified vegetation into general categories such as dry forests, mixed conifer forests, and moist forests (e.g., Baker 2015, Odion et al. 2014, Lydersen and North 2012, Williams and Baker 2012a, Hessburg et al. 2005, 2007), and reported considerable variation in fire regimes particularly in the mixed conifer and moist forest categories. Mershel et al. (2014) documented differences within what was classified as "mixed conifer" forests in eastern Oregon in how forests have responded to logging and fire suppression when compared to a classification that divide the mixed conifer classification into 4 different categories of mixed conifer forests. Similarly, classifying vegetation based solely on overstory species composition can increase the variability in associated fire regimes compared to finer classifications of landscape conditions that track abiotic environments (Williams and Baker 2012b, Abella and Denton 2009, Larson et al. 2009, Keeling et al. 2006, Meyn and Feller 2006, Sherriff and Veblen 2006, Howe and Baker 2003, Baker and Ehle 2001). Groupings of habitat types (Pfister et al. 1981) have been used effectively to describe similarities in forest dynamics and responses to disturbance processes (Milburn et al. 2015), and inclusion of classifications of potential vegetation have been recommended (Brown et al. 2004). Studies indicate finer scale classifications which include measures of abiotic environments or similar differences in ecological sites, can produce more specific descriptions of fire regimes for a particular region. Coarser scale classifications produce more generalized descriptions of fire regimes and thereby include more variation than may be useful for a specific landscape. The scale applied to historical fire regime classifications is an important factor to consider when reviewing and comparing the results of different studies, and this oversight often contributes to reported differences in findings.

Just as different classification systems result in different abilities to describe and quantify fire regimes for a specific location, regional classification differences also occur (Agee 2003). Failure to recognize regional differences when discussing fire regimes for plant communities containing similar species, can lead to over-generalizations and disagreements on the role of fire in stands dominated by the same or similar species. Rollins et al. (2002) found differences in fire regimes between wilderness areas in Arizona and Montana/Idaho were explained by regional differences in fuels and moisture status. Some fire regime studies applied to a specific vegetation type, such as ponderosa pine forests, discussed variation in types of fires without factoring in regional differences (e.g., Odion et al. 2014). For example, general descriptions of fire regimes in the western U.S. may discuss how ponderosa pine had stand replacing fire events that should be considered part of the normal fire regime for this vegetation type, and cite examples from different regions (Odion et al. 2014). Finer delineation of study areas clearly reveal ponderosa pine fire regimes were variable (Abella et al. 2007, Kaufmann et al. 2007) such as the Black Hills of South Dakota (Lentile et al. 2005, 2006) compared to Southwestern ponderosa pine (Covington et al. 2001, 1997, Mast et al. 1999, Fule et al. 1997), which also differed from higher elevation ponderosa pine fire regimes in the Front Range of Colorado (Williams and Baker 2012a, Kaufmann et al. 2007, Sherriff and Veblen 2006, Mast et al. 1998, Mast 1993). Murray et al. (1998) noted differences in fire regimes among different mountain ranges were attributable to differences in the surrounding landscapes that influenced fire sources. Korb et al. (2013) found differences in historical fire regimes among what they classified as 3 warm/dry mixed conifer stands occurring within a 50 km area, attributing the differences to variability in topography and other features. These studies illustrate the importance of describing fire regimes at the regional-level, while still recognizing differences can occur locally even within a region. When combined with consideration of the influences of different biophysical conditions and associated classification systems, additional emphasis for describing fire regimes at the region-level as well as across biophysical settings and vegetation types is warranted in order to properly characterize the function of fire as a disturbance process for ecosystem diversity.

In addition to classification and regional variations, methods used to describe fire regimes may contribute to different results. In particular, recent use of General Land Office (GLO) survey information as a tool for describing fire regimes has been used by a few authors (e.g., Baker 2015, Williams and Baker 2012b, 2011). However, many other fire researchers have questioned the validity of these methods (e.g., Fule et al. 2013) and there is an on-going debate on the accuracies of the findings using different methodologies. Hessburg et al. (2016) argued that while the GLO methodology can provide information useful to historical vegetation analyses, it is unsuitable for making spatially accurate assessments and determining fire regimes. Similarly, Stevens et al. (2016) questioned the use of FIA plot data for estimating fire regimes because the average stand age variable from these plots was not found to reflect occurrence of high severity fire regimes.

Methods

In order to describe reference conditions for disturbance states, the best available information on the historically occurring disturbance processes within a landscape should be developed (Keane et al. 2009, Egan and Howell 2001). Available literature on the primary disturbance processes occurring in the BSLRP project area was compiled. Information on historical fire regimes for the BSLRP project area was obtained from available published literature, technical reports, and study data. Where available, mean fire return intervals (MFRI) for specific ecological sites were summarized (Table 7) by researcher/author. Additionally, data on mean fire return intervals, expected fire regimes, and ecological site were reported in the project area by Barrett (2013, 2012), and Barrett and Jones (2001) (Table 8). A fire regime was assigned in this regard according to the predominant fire severity at a site. For example, NL indicates low severity fire was the predominate type of fire disturbance, MS indicates mixed severity fire predominated, and SR indicates high severity fire predominated.

The historical influences of insects and diseases were rated for each ecosystem in the landscape assessment area. Information on insect or disease occurrence or severity in different types of forest conditions was compiled from the literature and a rating of not occurring, low, moderate, or high risk assigned to each disturbance state for each ecological site. Sources for these ratings included Bassman et al. (2015), Randall et al. (2011), Parker et al. (2006), Bebi et al. (2003), Blocker et al. (2001), Byler and Hagle (2000), Hagle et al. (2000), Hessburg et al. (2000b), Olsen et al. (1996), Hessburg et al. (1994), and Lehmkuhl et al. (1994).

Reference conditions can be described using the above available historical information, existing stand information where applicable, and best scientific estimates where needed. The following sections compile and synthesize the information developed for ecological sites and disturbance states of the landscape assessment area relative to terrestrial forest ecosystem diversity. This information is organized and discussed by ecological site. For ecological sites where existing stand conditions can inform reference conditions, FIA plot data and other data were compiled and analyzed for key characteristics. For some ecosystems, part of the information contained in FIA data may be useful. For example, plots describing the high canopy cover-very large tree (DS12) disturbance state may in fact still largely represent the conditions that occurred historically. In addition, the very large tree component of these closed stands may also provide useful information on the composition of trees occurring in low or moderate canopy cover (DS10 or 11), where these stands may have established a dense understory of shade tolerant species due to anthropogenic changes in fire regimes or other factors over the past 100+ years. In such cases, the compositions, densities, or other characteristics of the overstory tree component can be used to help inform historical reference conditions. However, plot information on the densities of the understory trees may not be appropriate for establishing reference conditions if fire return intervals have been significantly altered. A full description of the methods used to develop the reference conditions is provided in Appendix E.

To develop reference models for structure, various data sources were evaluated for their application in developing reference conditions. The most consistently available plot data for developing descriptions is the FIA Program data resource. FIA plot data was used by first classifying each plot in terms of its ecological site, size class, and canopy cover using the following criteria:

1) Ecological site was assigned to a plot based on the R1 Habitat Type Groupings (Milburn et al. 2015; Appendix C) previously discussed under ecosystem-level classification,

Table 7. Mean fire return intervals (MFRI) by ecological site as reported by various authors in the project region. Number of plots used to determine MFRI are identified in parenthesis, where that information was available.

Source	Location	Ecological Site									
		HD	WD	MWD	MWMD	MWM	MCM	CM	CMD	COLD	TIM
Fisher and Bradley 1987	W. MT		5-25	5-50	15-50	50- >200	50- >200	>120	50-130	35- >300	35- >300
Fisher and Clayton 1983	E. MT	50-100									
Arno et al. 1995	M333C			LSF=20-30 & HSF=150-400+							
Arno et al. 1997	M333C			31 (1)							
Antos and Habeck 1981	M333C				100-200						
Barrett 2001, 2002	M333C			15-36 (15)		44-78 (5)		22-260 (35)	37-224 (42)		150 (1)
Freedman and Habeck 1985	M333C				30 (1)						
Davis 1980	M333C							21-175 (11)	47-175 (6)		
Heyerdahl et al. 2008	M333C				3-30						
Arno et al. 1997	M332B-W							24 (1)			
Barrett 2001	M332B-W		7-25 (10)	19-100 (13)					144 (1)		
Heyerdahl et al. 2008	M332B-W		2-30					9-42			
Grissino-Mayer et al. 2003	M332B-W		------ 2-14* ------							------ 50* ------	
Larson et al. 2009	M332B-W									19-54 mfri/100-350+ b/n high severity fires	
Barrett 2012, 2013	M332B-E			20-67 (16)					26-67 (15)		

* expected ecological site(s) based on covertype and site description

Table 8. Mean fire interval (MFI), minimum and maximum MFI, and number of plots summarized by fire regime, ecological site, and ecoregion. Data were summarized from fire history survey data (Barrett 2013, 2012, Barrett and Jones 2001).

	FIRE REGIME		Ecological Site								
			WD	MWD	MWMD	MWM	MCM	CM	CMD	COLD	TIM
ALL ECOREGIONS	NL	MFI	11	18	17			20	26		
		MIN-MAX	(9-17)	(6-30)	-			(17-24)	(23-29)		
		#PLOTS	6	51	1			4	2		
	MSA	MFI	30	32	34	44		28	44	32	
		MIN-MAX	(26-36)	(17-45)	(28-38)	(26-60)		(14-35)	(22-66)	(28-35)	
		#PLOTS	6	24	5	6		4	27	4	
	MSB	MFI		48	86	88	78	98	91	97	180
		MIN-MAX		(40-67)	(58-123)	(40-120)	(58-128)	(47-127)	(52-224)	(55-200)	(28-275)
		#PLOTS		16	6	4	5	23	26	7	7
	L	MFI		135	150	97	221	180	193	205	150
		MIN-MAX		(135-135)	-	-	(200-260)	(97-260)	(146-222)	(146-304)	(150-150)
		#PLOTS		2	1	1	6	22	12	15	2
M333C	NL	MFI		21				20			
		MIN-MAX		(15-27)				(17-24)			
		#PLOTS		12				4			
	MSA	MFI		31	34	44		28	48		
		MIN-MAX		(26-36)	(28-38)	(26-60)		(14-35)	(35-66)		
		#PLOTS		9	5	6		4	16		
	MSB	MFI			93	56	98	77	105	78	150
		MIN-MAX			(78-123)	(40-78)	(85-128)	(47-127)	(59-224)	-	(150-150)
		#PLOTS			3	3	4	21	12	1	2
	L	MFI			150	97	221	180	185	184	150
		MIN-MAX			-	-	(200-260)	(97-260)	(146-222)	(146-222)	(150-150)
		#PLOTS			1	1	6	22	11	9	2
M332B-WEST	NL	MFI	11	16					23		
		MIN-MAX	(9-17)	(6-30)					-		
		#PLOTS	5	21					1		
	MSA	MFI		28						32	
		MIN-MAX		(25-30)						(28-35)	
		#PLOTS		3						4	
	MSB	MFI		40				120	80	117	211
		MIN-MAX		(40-40)				(120-120)	(80-80)	(55-200)	(28-275)
		#PLOTS		2				2	5	6	5
	L	MFI		135					200	226	
		MIN-MAX		-					-	(200-304)	
		#PLOTS		1					1	6	
M332B-EAST	NL	MFI	11	15					29		
		MIN-MAX	-	(7-27)					-		
		#PLOTS	1	19					1		
	MSA	MFI		37					40		
		MIN-MAX		(17-45)					(22-45)		
		#PLOTS		12					11		
	MSB	MFI		57					89		
		MIN-MAX		(44-67)					(52-151)		
		#PLOTS		14					9		
	L	MFI		135							
		MIN-MAX		-							
		#PLOTS		1							

**See Appendix D (Table D-1) for a description of the fire regime as reported by Barrett and Jones (2001) and its assumed relationship to fire regimes described in this document.

2) Size class was assigned to a plot using sequential progression beginning with the largest tree cohort and progressing through the data until a size class was identified according to the criteria identified in Table 9. These criteria were developed by Region 1 under the direction of Eric Henderson and Chip Fisher for use in this sequential analysis. This method differs from many size class classifications in use today, as most are based on the dominant average conditions in a stand. The differences between these two methods and the implications for their use in describing ecosystem diversity are discussed below.
3) Percent canopy cover was available for less than half of the FIA plots in the landscape assessment area. As an alternative to canopy cover, stand density index (SDI) was calculated for each plot using methods developed by Woodall and Miles (2006). The SDI values used to identify low, moderate, or high canopy cover classes used in the structure classification are identified in Table 10. To determine the cut-off for SDI values for a canopy cover class, FIA canopy cover values where they occurred were plotted against the calculated SDI value. The linear regression was calculated ($r^{2=}$ 0.45). While showing a clear relationship between SDI and canopy cover, the lack of a better fit for the data included in the relationship means that many of the plots used for describing reference conditions may have been misclassified in terms of their canopy cover. Lack of specific canopy cover measurements in many FIA plots is thus a limitation on the ability of FIA plots to be accurately classified into an appropriate cover category.

Table 9. Sequential progression through plot data to determine the largest tree-size cohort present.

	STEP 1	STEP 2, if condition 1 is not met	STEP 3, if condition 2 is not met	STEP 4, if condition 3 is not met	STEP 5, if condition 4 is not met
Ecological Site	**TPA >=20in. DBH (VERY LARGE)**	**TPA >=15in. DBH (LARGE)**	**TPA >=5in. DBH (MEDIUM)**	**TPA >=0.1in. DBH (SMALL)**	**(GFSS)**
Hot-Dry	>=8	>=10	>=15	>=20	If not in another category
Warm-Dry	>=8	>=10	>=15	>=20	
Mod Warm-Dry	>=8	>=10	>=15	>=20	
Mod Warm-Mod Dry	>=8	>=10	>=15	>=20	
Mod Warm-Moist	>=10	>=10	>=15	>=20	
Mod Cool-Moist	>=10	>=10	>=15	>=20	
Cool-Moist	>=10	>=10	>=15	>=20	
Cool Mod-Dry	>=10	>=10	>=15	>=20	
Cold	>=10	>=10	>=15	>=20	
Timberline	>=10	>=10	>=15	>=20	

Table 10. Relationship of canopy cover classes and stand density index (SDI) for each size class as used to classify FIA plot data in the landscape assessment.

SIZECLASS	OPEN (<40% canopy Cover)	MODERATE (40-59% canopy cover)	CLOSED (>=60% canopy cover)
SMALL	----------------------- SDI <250	------------------------	SDI>=250
MEDIUM	SDI <150	SDI >=150 to 249	SDI >=250
LARGE	SDI <150	SDI >=150 to 249	SDI >=250
VERY LARGE	SDI <150	SDI >=150 to 249	SDI >=250

The sequential size class method used in this assessment differs from some classification systems that use the dominant size class for classifying which disturbance state a plot represented. The sequential method identifies if a minimum number of trees of a certain size occur, and if this minimum number is present, then that size class is assigned to the stand or the plot. This minimum number of trees is considered sufficient to classify the stand or plot as a forested condition. The trees present provide the indicated structural component, such as very large trees that might be preferred by various wildlife species. The sequential analysis will maintain that size class as long as the stand or plot supports that minimum number of trees in that size class. The stand may also have additional trees of smaller sizes occurring as well. These will influence whether a stand is considered an open overstory stand, moderate overstory stand, or closed overstory stand. However, using the sequential analysis, they will not cause the stand to be classified into another size class as long as enough of the larger sized trees remain

The alternative method of using the dominant sized trees to classify a stand very often masks the presence of larger trees that provide stand characteristics required by many wildlife species. Stevens et al. (2016) similarly reported on the limitations of using average stand age from FIA plots in interpretations of numbers of larger trees. For example, if a stand has 20 very large trees/acre but also has 200 medium sized trees/acre, it may be classified as a medium tree stand despite the fact that it has a significant presence of very large trees. If the medium trees are thinned, then the stand suddenly changes in classification to a very large tree stand, even though these trees were present all along. Another concern is the high variability in numbers of different size trees that get classified into smaller size classes when dominant tree sizes are used for classification. This causes two problems. First, it underestimates the amounts of large trees present (and the associated habitat provided by these large trees for species requiring these stand characteristics). In comparisons of classifications of FIA plots using dominant tree stand classification compared to the sequential method, the dominant tree classification reported only 38 plots as very large tree plots, while the sequential analysis classified 246 of the plots as meeting the minimum requirements for presence of very large trees (Table 11). The stands with very large trees present meet the defined criteria for this size class and provide these habitat components, but are not recognized as being present in the dominant tree classification. This would suggest a much smaller percentage of the landscape having very large trees present than actually occur. The second problem is the greatly increased variance in trees occurring in smaller size classes when using dominant tree size. Table 12 reveals when using dominant tree size for classifying FIA plots, some plots were reported as having 156 very large trees/acre but were still classified as a large tree stand because of the additional numbers of smaller sized trees in the plot. The sequential method assures that when a minimum number of larger size classes of trees are present, the stand will meet the criteria to be classified to that larger size class regardless of the presence or not or smaller sized trees.

Once a plot had been assigned an ecological site, size class, and canopy cover class per the methods described above, the following common mean values were calculated using established methods.

- Live trees per acre
- Dead trees per acre
- Basal area weighted mean diameter
- Coarse woody debris, tons per acre
- % Canopy cover of forbs, grass, and shrubs

Table 11. Comparison of FIA plot classifications based on a dominant size class system compared to a sequential size class system of classification. For example, the table shows the sequential size class system identified 246 plots that would meet the criteria of having enough trees to qualify as a very large tree stand, while the dominant size class system classified 38 stands as being very large tree size class.

		DOMINANT SIZE METHOD					**TOTAL PLOTS**
		GFSS	**Small**	**Medium**	**Large**	**Very Large**	
SEQUENTIAL SIZE METHOD	**GFSS**	32	5	1	1	2	**41**
	Small		28	2	2		**32**
	Medium		40	342	3		**385**
	Large		1	167	24	1	**193**
	Very Large			114	97	35	**246**
TOTAL PLOTS		**32**	**74**	**626**	**127**	**38**	**897**

Table 13 identifies the reference conditions calculated from FIA plot data that are likely to not be accurate for historical reference conditions. For these ecosystem characteristics, adjustments to the FIA values are needed. These adjustments should be made using available information about historical ecosystem conditions and best scientific interpretations.

Table 12. Example of tree size distributions for one ecological site comparing the dominant size class system with the sequential size class system of classification. This table shows that under the dominant size class system, a stand that was classified as a medium tree stand could have up to 87 very large trees/acre or 284 large trees/acre in the stand and still be classified as a medium tree stand.

		Dominant Size Method			Sequential Size Method		
		OPEN	MODERATE	CLOSED	OPEN	MODERATE	CLOSED
	DBH	AVG (MIN-MAX)	AVG (MIN-MAX)	AVG (MIN-MAX)	AVG (MIN-MAX)	AVG (MIN-MAX)	AVG (MIN-MAX)
GFS	<0.1	3688 (150-13794)			3688 (150-13794)		
	0.1-4.9	0			0		
	5.0-14.9	0			0		
	15.0-19.9	0			0		
	20.0+	0			0		
SMALL	<0.1	16312 (900-52200)	9699 (1200-18000)	25127 (240-75600)	5512 (75-19500)	16492 (16492-16492)	8099 (5997-10200)
	0.1-4.9	1850 (300-5100)	4462 (300-7200)	12408 (1140-28200)	469 (75-1200)	300 (300-300)	14399 (4198-24600)
	5.0-14.9	76 (0-318)	510 (0-985)	247 (0-545)	2 (0-6)	0	0
	15.0-19.9	0	0	0	0	0	0
	20.0+	0	0	0	0	0	0
MEDIUM	<0.1	3696 (0-15900)	10929 (0-104100)	9740 (0-73500)	5444 (0-52200)	11513 (0-104100)	12545 (75-75600)
	0.1-4.9	563 (0-3600)	1212 (0-6000)	1684 (0-11100)	703 (0-5100)	1672 (0-7200)	2589 (0-28200)
	5.0-14.9	333 (0-2070)	829 (48-4112)	608 (54-4179)	307 (18-2070)	935 (90-4112)	522 (112-3096)
	15.0-19.9	16 (0-165)	27 (0-284)	24 (0-243)	1 (0-6)	1 (0-6)	1 (0-6)
	20.0+	4 (0-26)	7 (0-87)	7 (0-82)	0 (0-6)	1 (0-8)	0 (0-6)
LARGE	<0.1	2775 (0-8400)	3229 (0-12600)	550 (300-1050)	2785 (0-9000)	8456 (0-64200)	8541 (0-73500)
	0.1-4.9	382 (0-1800)	700 (0-3300)	525 (150-825)	418 (0-1500)	954 (0-5400)	1053 (0-7500)
	5.0-14.9	335 (48-1114)	371 (48-1083)	223 (187-271)	136 (0-517)	520 (48-2878)	393 (54-1845)
	15.0-19.9	71 (6-215)	85 (6-496)	2 (0-6)	22 (12-32)	33 (12-118)	37 (12-166)
	20.0+	61 (4-146)	56 (0-156)	20 (12-24)	1 (0-6)	1 (0-6)	1 (0-6)
VERY LARGE	<0.1	1185 (0-4500)	5072 (150-17700)		3027 (0-15900)	7099 (0-83700)	8412 (0-34500)
	0.1-4.9	375 (0-2100)	1397 (0-6300)		570 (0-2700)	1218 (0-6300)	2306 (150-5400)
	5.0-14.9	158 (18-480)	200 (66-458)		330 (0-1114)	587 (48-2630)	1055 (187-4179)
	15.0-19.9	66 (12-140)	45 (0-222)		66 (0-215)	81 (0-496)	56 (0-243)
	20.0+	49 (12-99)	85 (18-258)		45 (10-146)	59 (10-258)	36 (10-82)

Table 13. Anthropogenic changes from activities such as fire exclusion, logging, or grazing that could cause differences in ecosystem characterisitics from what occurred historically to those measured in recent FIA plot sampling.

<table>
<tr><th>OPEN
(<40% Canopy Cover)</th><th>MODERATE
(40-60% Canopy Cover)</th><th>CLOSED
(>60% Canopy Cover)</th></tr>
<tr><td colspan="3">DS1</td></tr>
<tr><td colspan="3">Canopy cover of grasses vs forbs vs shrubs may have been altered by changes to fire regime or previous grazing</td></tr>
<tr><td colspan="3">Timber harvest or salvage logging may affect stand structures such as snags and remaining live trees</td></tr>
<tr><td colspan="2">DS2</td><td>DS3</td></tr>
<tr><td colspan="2">Canopy cover of grasses vs forbs vs shrubs may have been altered by changes to fire regime or previous grazing</td><td>Canopy cover of grasses vs forbs vs shrubs may have been altered by changes to fire regime or previous grazing</td></tr>
<tr><td colspan="2">Timber harvest may affect stand structures such as snags and remaining live trees</td><td>Timber harvest may affect stand structures such as snags and remaining live trees</td></tr>
<tr><td>DS4, DS7, DS10</td><td>DS5, DS8, DS11</td><td>DS6, DS9, DS12</td></tr>
<tr><td>More small trees in understory due to lack of low-moderate severity fire</td><td>More small trees in understory due to lack of low-moderate severity fire</td><td>Mortality/snags may have increased due to increased insects and disease</td></tr>
<tr><td>CWD more persistent due to lack of fire</td><td>CWD more persistent due to lack of fire</td><td rowspan="2">Loss of western white pine and whitebark pine structures on some ecosites</td></tr>
<tr><td>Loss of whitebark pine structures on some ecosites</td><td>Loss of western white pine and whitebark pine structures on some ecosites</td></tr>
<tr><td>Lower average tree diameters due to selective harvesting</td><td>Lower average tree diameters due to selective harvesting</td><td></td></tr>
<tr><td>Mortality/snags may have increased due to increased insects and disease</td><td>Mortality/snags may have increased due to increased insects and disease</td><td></td></tr>
<tr><td>Canopy cover of grasses-forbs-shrubs may have been altered by changes to fire regime or previous grazing</td><td>Canopy cover of grasses-forbs-shrubs may have been altered by changes to fire regime or previous grazing</td><td></td></tr>
</table>

Results

Insects and Disease

Figures 10-12 identify the expected historical risk of primary insect and diseases by ecosystem and is presented using the ecosystem diversity framework for each of the 3 ecoregions.

TREE SIZECLASS	MOD WARM-DRY Canopy Cover[a] Open	Moderate	Closed	MOD WARM-MOD DRY Canopy Cover Open	Moderate	Closed	MOD WARM-MOIST Canopy Cover Open	Moderate	Closed	MOD COOL-MOIST Canopy Cover Open	Moderate	Closed
GFS-SEEDLING	DS1: -			DS 1: -			DS 1: -			DS 1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -	DS2: -		DS3: -	DS2: -		DS3: -	DS2: -		DS3: -
MEDIUM TREE	DS4: PED, MPB	DS5: PED, MPB	DS6: PED, MPB, RD, DFTM	DS4: MPB	DS5: MPB, RD	DS6: MPB, RD	DS4:	DS5: MPB	DS6: MPB, RD	DS4:	DS5: MPB	DS6: MPB, RD, PED
LARGE TREE	DS7: PED, MPB	DS8: PED, MPB, RD	DS9: PED, MPB, RD, DFTM	DS7: MPB	DS8: MPB, RD	DS9: DFB, MPB, RD	DS7:	DS8: MPB	DS9: DFB, MPB, RD, SB/DFTB	DS7:	DS8: MPB	DS9: MPB, RD, PED
VERY LARGE TREE	DS10: PED, MPB	DS11: PED, MPB, RD	DS12: PED, MPB, RD, DFTM	DS10: MPB	DS11: MPB, RD	DS12: DFB, MPB, RD	DS10:	DS11: MPB	DS12: DFB, MPB, RD, SB/DFTM	DS10: DFB	DS11: DFB, RD	DS12: DFB, RD

TREE SIZECLASS	COOL-MOIST Canopy Cover Open	Moderate	Closed	COOL-MOD DRY Canopy Cover Open	Moderate	Closed	COLD Canopy Cover Open	Moderate	Closed	TIMBERLINE Canopy Cover Open	Moderate	Closed
GFS-SEEDLING	DS 1: -			DS 1: -			DS1: -			DS1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -	DS2: -		DS3: -	DS2: -		DS3: -	DS2: -		DS3: -
MEDIUM TREE	DS4:	DS5: WSBW	DS6: WSBW, MPB, RD, SD	DS4:	DS5: WSBW, MPB	DS6: WSBW, MPB, SB, RD	DS4: MPB	DS5: MPB	DS6: MPB	DS4: MPB	DS5: MPB	DS6: MPB
LARGE TREE	DS7: WSBW, DFB	DS8: WSBW, DFB, MPB, SB.DFTM	DS9: WSBW, DFB, RD, SB/DFTM	DS7:	DS8: DFB, MPB, RD	DS9: WSBW, RD/DFB, MPB, SB/DFTM	DS7: MPB	DS8: MPB, SB	DS9: MPB, SB	DS7: MPB	DS8: MPB, SB	DS9: MPB, SB
VERY LARGE TREE	DS10: WSBW	DS11: RD, DFB	DS12: WSBW, RD, DFB, SB/DFTM	DS10: DFB	DS11: DFB, WSBW, RD	DS12: DFB, WSBW, RD, SB/DFTM	DS10: MPB	DS11: MPB, SB	DS12: MPB, SB	DS10: MPB	DS11: MPB, SB	DS12: MPB, SB

DFB- Douglas-fir beetle (*Dendroctonus pseudotsugae*)
DFTM - Douglas-fir tussock moth (*Orgyia pseudotsugata*)
MPB - mountain pine beetle (*D. ponderosae*)
PED - pine engraver beetle (*Ips pini*)
RD - root disease including *Armillaria ostoya* e
SB - spruce beetle (*D. rufipennis*)
WSBW - western spruce budworm (Choristoneura occidentalis)

Figure 10. Estimated historical risk of primary insect and root disease influences for M333C as presented using the ecosystem diversity framework. Letters in red indicate a high risk of occurrence, orange letters indicate a moderate risk of occurrence, while green letters indicate a low risk of occurrence. Absence of an insect or root disease in an ecosystem indicates that it is very unlikely to occur in this ecosystem. See methods for references.

TREE SIZECLASS	WARM-DRY			MOD WARM-DRY			COOL-MOIST		
	Canopy Cover[a]			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
	DS1			DS1			DS 1		
GFS-SEEDLING	-			-			-		
	DS2		DS3	DS2		DS3	DS2		DS3
SAPLING - SMALL TREE	-		-	-		-	-		-
	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6
MEDIUM TREE	PED MPB	PED MPB	PED MPB	PED MPB	PED MPB	PED MPB RD DFTM		WSBW	WSBW MPB RD SD
	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9
LARGE TREE	PED MPB	PED MPB	PED MPB RD	PED MPB	PED MPB RD	PED MPB RD DFTM	WSBW DFB	WSBW DFB MPB SB.DFTM	WSBW DFB RD SB/DFTM
	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12
VERY LARGE TREE	PED MPB	PED MPB	PED MPB RD	PED MPB	PED MPB RD	PED MPB RD DFTM	WSBW	RD DFB	WSBW RD DFB SB/DFTM

TREE SIZECLASS	COOL-MOD DRY			COLD			TIMBERLINE		
	Canopy Cover			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
	DS 1			DS1			DS1		
GFS-SEEDLING	-			-			-		
	DS2		DS3	DS2		DS3	DS2		DS3
SAPLING - SMALL TREE	-		-	-		-	-		-
	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6
MEDIUM TREE		WSBW MPB	WSBW MPB SB RD	MPB	MPB	MPB	MPB	MPB	MPB
	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9
LARGE TREE		DFB MPB RD	WSBW RD/DFB MPB SB/DFTM	MPB	MPB SB	MPB SB	MPB	MPB SB	MPB SB
	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12
VERY LARGE TREE	DFB	DFB WSBW RD	DFB WSBW RD SB/DFTM	MPB	MPB SB	MPB SB	MPB	MPB SB	MPB SB

DFB- Douglas-fir beetle (*Dendroctonus pseudotsugae*)
DFTM - Douglas-fir tussock moth (*Orgyia pseudotsugata*)
MPB - mountain pine beetle (*D. ponderosae*)
PED - pine engraver beetle (*Ips pini*)
RD - root disease including *Armillaria ostoyae*
SB - spruce beetle (*D. rufipennis*)
WSBW - western spruce budworm (Choristoneura occidentalis)

Figure 11. Estimated historical risk of primary insect and root disease influences for M332B-West as presented using the ecosystem diversity framework. Letters in red indicate a high risk of occurrence, orange letters indicate a moderate risk of occurrence, while green letters indicate a low risk of occurrence. Absence of an insect or root disease in an ecosystem indicates that it is very unlikely to occur in this ecosystem. See methods for references.

HOT-DRY (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GFS-SEEDLING	DS1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -
MEDIUM TREE	DS4: PED MPB	DS5: PED MPB	DS6: PED MPB
LARGE TREE	DS7:	DS8: DFTM	DS9: DFTM RD
VERY LARGE TREE	DS10:	DS11: DFTM	DS12: DFTM RD

WARM-DRY (Canopy Cover[a])

TREE SIZECLASS	Open	Moderate	Closed
GFS-SEEDLING	DS1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -
MEDIUM TREE	DS4: PED MPB	DS5: PED MPB	DS6: PED MPB RD
LARGE TREE	DS7: PED MPB	DS8: PED MPB	DS9: PED MPB RD DFTM
VERY LARGE TREE	DS10: PED MPB	DS11: PED MPB	DS12: PED MPB RD DFTM

MOD WARM-DRY (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GFS-SEEDLING	DS1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -
MEDIUM TREE	DS4: PED MPB	DS5: PED MPB	DS6: PED MPB RD DFTM
LARGE TREE	DS7: PED MPB	DS8: PED MPB RD	DS9: PED MPB RD DFTM
VERY LARGE TREE	DS10: PED MPB	DS11: PED MPB RD	DS12: PED MPB RD DFTM

COOL-MOIST (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GFS-SEEDLING	DS 1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -
MEDIUM TREE	DS4:	DS5: WSBW	DS6: WSBW MPB RD SD
LARGE TREE	DS7: WSBW DFB	DS8: WSBW DFB MPB SB.DFTM	DS9: WSBW DFB RD SB/DFTM
VERY LARGE TREE	DS10: WSBW	DS11: RD DFB	DS12: WSBW RD DFB SB/DFTM

COOL-MOD DRY (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GFS-SEEDLING	DS 1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -
MEDIUM TREE	DS4:	DS5: WSBW MPB	DS6: WSBW MPB SB RD
LARGE TREE	DS7:	DS8: DFB MPB RD	DS9: WSBW RD/DFB MPB SB/DFTM
VERY LARGE TREE	DS10: DFB	DS11: DFB WSBW RD	DS12: DFB WSBW RD SB/DFTM

COLD (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GFS-SEEDLING	DS1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -
MEDIUM TREE	DS4: MPB	DS5: MPB	DS6: MPB
LARGE TREE	DS7: MPB	DS8: MPB SB	DS9: MPB SB
VERY LARGE TREE	DS10: MPB	DS11: MPB SB	DS12: MPB SB

TIMBERLINE (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GFS-SEEDLING	DS1: -		
SAPLING - SMALL TREE	DS2: -		DS3: -
MEDIUM TREE	DS4: MPB	DS5: MPB	DS6: MPB
LARGE TREE	DS7: MPB	DS8: MPB SB	DS9: MPB SB
VERY LARGE TREE	DS10: MPB	DS11: MPB SB	DS12: MPB SB

DFB- Douglas-fir beetle (*Dendroctonus pseudotsugae*)
DFTM - Douglas-fir tussock moth (*Orgyia pseudotsugata*)
MPB - mountain pine beetle (*D. ponderosae*)
PED - pine engraver beetle (*Ips pini*)
RD - root disease including *Armillaria ostoya* e
SB - spruce beetle (*D. rufipennis*)
WSBW - western spruce budworm (Choristoneura occidentalis)

Figure 12. Estimated historical risk of primary insect and root disease influences for M332B-East as presented using the ecosystem diversity framework. Letters in red indicate a high risk of occurrence, orange letters indicate a moderate risk of occurrence, while green letters indicate a low risk of occurrence. Absence of an insect or root disease in an ecosystem indicates that it is very unlikely to occur in this ecosystem. See methods for references.

Ecosystem Diversity and Fire

Point-in-time Historical Information

Losensky (1994) noted that timber harvest began in the project region with the establishment of mining camps in the 1860's and efforts to suppress natural processes such as fire also began around this time. Timber harvest increased dramatically after the establishment of the Great Northern Railway in 1892 but were mostly limited for many years to the zone adjacent to the railway.

Ayres (1900) provided an inventory of the forests in the Clearwater and Swan drainages. While some effects from logging and livestock grazing had already occurred in some portions of the project area by the date of his report, the descriptions of forest conditions still provide an excellent point-in-time reference. He estimated 240 sq. mi. out of the 728 sq. mi. he surveyed in the Clearwater and Swan Valleys showed evidence of fire in the past 40 years. In addition, he reported frequent less intense "light" fires occurred that did not have obvious impacts on the vegetation and weren't included in the 240 sq. mi. estimate. He also estimated that 600 sq. mi. of the 728 sq. mi. area had burned in the last 100 years, noting "Probably 90% of the valley has been burned over in the past 100 years, but evidences of such burns are so hidden by the forest left growing over them or grown since that it has been found impracticable to map all such areas." He found a lack of litter due to frequent fires and that underbrush was scant throughout much of the valley bottom or benches in the region. He provided numerous pictures of open stands of yellow (ponderosa) pine as well as pictures of stands of western larch that survived fires that thinned out lodgepole and red (Douglas) fir trees. He noted the composition of the forest was patchy, with various ages and species of trees. Ayres also reported that "The yellow pine lands, both about the headwaters of the Swan River and in the Clearwater drainage are, as usual, more free from young stock than the forests of other species, yet some of these tracts have a fair sprinkling of red fir, larch, and spruce coming in underneath the pine. As a rule these species do not reach tree size, being killed while small by repeated fires." Tables 14 and 15 present his estimates of timber in the Swan and Clearwater Valley.

Table 14. Merchantable timber estimates from Ayres (1900) for the Clearwater and Swan Valleys (8" tops) in million board feet (MBF) and relative amounts in percent.

Species	MBF (8" top)	% of Total
Larch (western larch)	1,050,000	56
Red fir (Douglas-fir)	500,000	27
Spruce (Engelmann spruce)	175,000	9
Yellow pine (ponderosa pine)	100,000	5
Lodgepole pine	30,000	2
White pine (western white pine)	10,000	0.5

Table 15. Timber estimates for trees that were considered by Ayres (1900) to be too small for merchantable timber, amounts presented in cords and relative amounts in percent.

Species	Cords	% of Total
Larch (western larch)	1,200,000	26
Lodgepole pine	1,000,000	21
Spruce (Engelmann spruce)	900,000	19
Balsam (subalpine fir)	500,000	11
Red fir (Douglas-fir)	400,000	9
Yellow pine (ponderosa pine)	100,000	2
Other	560,000	12

About 10 years prior to Ayres survey other surveyors working for the Public Land Survey of the U.S. General Land Office (GLO), were mapping the original grid of township and range lines and section boundaries across the West. These early surveys sometimes documented the condition of the land surface before considerable settlement had taken place. GLO field surveyor's notes have been used by many researchers to describe historic landscapes and vegetation conditions (i.e., Montana – Habeck 1994). While most of the BSLRP project area was not surveyed for vegetation during the late-1800 period, a few surveys did occur. Township 16N and Range 15W was surveyed around 1890. Ecological site mapping for this Township (e.g. Figure 5) depict 80% of the area as forested ecological sites with Moderately Warm and Dry representing 56% of the area. Cool and Moist and Cool and Moderately Dry ecological sites represented 17% and 6%, respectively. The other 20% represented smaller amounts of riparian and wetland sites including Seeley Lake and the Clearwater River.

The results of 1890 GLO survey were consistent with the notes and photographs documented a few years later by Ayres. Species encountered were primarily ponderosa pine, western larch, and Douglas-fir. Very few references to lodgepole pine, Engelmann spruce, and subalpine fir were recorded in the overall Township. Diameters recorded at section and corner stones, as well as along survey lines, were predominantly very large (diameters noted up to 67" for western larch) and large trees. General section descriptions noted primarily ponderosa pine, western larch, and Douglas-fir species of high quality, well-timbered stands. While the density of trees was not measured, the section descriptions and tree distances measured at cornerstones indicated relatively open to moderately open, yet well-timbered stands. These descriptions suggest the photographs taken by Ayres in this same Township (Placid Lake area - Ecoregion M332B-West) were typical forest condition for this township, not atypical.

Losensky (1994) reviewed timber surveys conducted in the early to mid-1930s for U.S. Forest Service lands in the region. The results were summarized by ecoregion and a process developed to use the 1930's information to make estimates of conditions around the late 1800's, by making adjustments for harvest. He noted over-mature conditions were likely underestimated due to lack of complete harvest information. For all of ecoregion M333C and the Clearwater watershed (mapped as part of M332B-West for this assessment) on its southern boundary, Losensky reported subalpine fir was the most common forest cover type (36.4%). The western larch-Douglas-fir cover-type (28.4%) was also common and occupied the valleys and lower to mid-slope positions. Lodgepole pine (27.1%) was a close third and was found on slopes above the western larch-Douglas-fir type. Engelmann spruce (6.4%) was found on moist benches, riparian areas, and high basins. White pine (0.5%) was located in protected areas and included an array of species such as western red cedar, grand fir, and western hemlock. Ponderosa pine (0.8%) was found mainly near the southern boundary of the ecoregion or on dry southwest slopes. The Douglas-fir cover-type (0.4%) was found on cool, dry sites generally above the limits of ponderosa pine.

Losensky reports age-class distribution for Ecoregion M333C as follows:

- Non-Stocked 15.8%
- Seedling-Sapling (<40 yrs.) 16.1%
- Poles (41-100 yrs.) 13.3%
- Mature (101-150 yrs.) 19.6%
- Over-Mature (>150 yrs.) 35.1%

Losensky also reported this ecoregion shows slightly less percentages than might be expected in the young age classes and greater percentages of over-mature stands in the western larch-Douglas-fir cover-type likely "due the extensive stands in the Valleys of the Swan and Clearwater drainages where they take on the appearance of open grown ponderosa pine stands with frequent underburns." Apparently, the 1930's timber surveys further support what appears to be an unusual pattern of older stands and more open, frequent fire-maintained conditions than would be expected in the rest of Ecoregion M333C.

For ecoregion M332B (M332B-West and M332B-East per this assessment) that does not include the Clearwater basin, Losensky reported lodgepole pine was the most common forest cover type (28%). Douglas-fir and ponderosa pine cover types followed at 16.3 and 15.8%, respectively. Ponderosa pine was primarily confined to lower slopes and valley bottoms with many of these stands being predominantly ponderosa pine and many in a savannah condition. The subalpine fir cover type was also relatively common (14.4%). Lodgepole pine was a major species in both the Douglas-fir and subalpine fir types. The western larch-Douglas-fir cover-type (4.0%) was less common than in M333C and the Clearwater Basin.

Losensky reported age-class distribution for Ecoregion M332B as follows:

- Non-Stocked 5.1 %
- Seedling-Sapling (<40 yrs.) 18.8%
- Poles (41-100 yrs.) 27.5%
- Mature (101-150 yrs.) 16.0%
- Over-Mature (>150 yrs.) 32.6%

He also noted over-mature stands were common in the ponderosa pine, western larch-Douglas-fir and spruce-fir cover types.

Reference Condition Data and Characteristics

Fire-adapted plant species such as ponderosa pine and western larch have developed physical adaptations such as thick bark to protect larger trees from low to moderate severity fires (Marcoux et al. 2015, Merschel et al. 2014, Fitzgerald 2004, Arno et al. 1995, Ayres 1900). Fire-dependent species have developed life cycle strategies to take advantage of fire events such as the serotinous cones of lodgepole pine or rapid growth rates in western white pine. Table 16 lists the dominant tree species of the northern Rockies and their general susceptibility to fire. These characteristics allow different species response to the different fire severities that occurred in the project area.

Stand reconstruction studies, fire scar studies, and historical observations provide empirical information on stand compositions and structural characteristics under historical disturbance processes. Fire was a major historical disturbance in the landscape assessment area. Restoration at the ecosystem level involves returning stands to compositions and structures produced under the historical disturbance processes. Thus, developing descriptions of reference conditions that identify and quantify the species composition and structural characteristics of each ecosystem in the ecosystem framework is a key component of a landscape assessment as it provides the foundation for setting stand level restoration objectives.

Table 16. Fire effects on primary tree species occurring in the BSLRP area. Sources of information include Marcoux et al. (2015), Merschel et al. (2014), Arno et al. 1997, 1995, Arno (1980), Flint (1925), and Ayres (1900).

Species	Degree of Fire Resistance	
	Medium-size or greater	*Seedlings/Saplings*
Western larch	Very High	Medium
Ponderosa pine	High	Medium
Douglas-fir	High	Low
Grand fir	Medium	Low
Lodgepole pine	Medium	Low
Western white pine	Medium	Low
Western redcedar	Medium	Low
Whitebark pine	Medium	Low
Alpine larch	Medium	Low
Limber pine	Medium	Low
Engelmann spruce	Low	Low
Mountain hemlock	Low	Low
Western hemlock	Low	Low
Rocky Mountain Juniper	Low	Low
Subalpine fir	Very low	Low

Reference conditions are presented using key characteristics of forest species compositions and structure. Tree species composition and distribution are presented for each of the 3 ecoregions using the ecosystem diversity framework (Figures 13-15). Species composition for understory grasses, forbs and shrubs are presented in Appendix E for each ecosystem in the BSLRP area. An example is provided here for the Moderately Warm and Dry ecological site (Table 17). Structural characteristics for live and dead trees, as well as general characteristics including basal area weighted diameter, coarse woody debris, and percent canopy cover for forbs, grasses, and shrubs are summarized by ecosystem for each of the 3 ecoregions. An example for all 3 ecoregions of the Moderately Warm and Dry ecological site (Tables 18-26) follows this paragraph. A database is available that can be queried to provide the specific reference conditions for use in setting key structural characteristics for ecosystem restoration purposes. Box 1 provides an example of a description of the historical reference conditions for the very large tree, open canopy ecosystem in the moderately warm dry ecological site for the M332B-West ecoregion. Similar descriptions could be developed for all ecosystems identified for the BSLRP project area.

Box 1. Example reference conditions for describing restoration objectives for the moderately warm and dry ecological site - ecoregion M332B-West.

This ecological site occurs most commonly at low to mid-elevation, particularly on dry southerly aspects where it is often transitional to grass-shrub ecoystems. Soils are of moderate depth and frequently droughty, with low to moderate productivity. It is primarily influenced by the non-lethal fire regime at less than 25 year fire return intervals but is also sometimes influenced by mixed-severity fire regimes on moister areas. Forest conditions resulting from this type of disturbance were characterized by an open overstory (<40% canopy cover) comprised primarily of ponderosa pine but with some Douglas-fir. Very large trees (>20” dbh) were the most common size class averaging 13 trees per acre within a range of 8 to 18, but may also have included an average of 11 large trees (15 to 20” dbh) within a range of 0 to 35. Trees were often scattered with some clumps. All age classes were present but smaller trees were low in numbers. Understory plants included antelope bitterbrush, common snowberry, snowbush ceonothus, bluebunch wheatgrass, rough fescue, arrowleaf balsamroot, common gallardia, fireweed, silky lupine, and woodland strawberry. Very large and large snags averaged 1 each per acre within a range of 0 to 7 and 0 to 18, respectively. Basal area weighted diameter averaged 19 within a range of 15 to 27.

MOD WARM-DRY (Canopy Cover[a])

TREE SIZECLASS	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS1: PIPO[b], LAOC, PICO		
SAPLING - SMALL TREE	DS2: PIPO, LAOC, PSME, PICO		DS3: PIPO, LAOC, PSME, PICO
MEDIUM TREE	DS4: PIPO, LAOC, PSME	DS5: PIPO, LAOC, PSME, PICO	DS6: LAOC, PIPO, PSME, PICO
LARGE TREE	DS7: PIPO, LAOC, PSME	DS8: LAOC, PIPO, PSME	DS9: LAOC, PSME, PIPO
VERY LARGE TREE	DS10: PIPO, PSME, LAOC	DS11: PIPO, LAOC, PSME	DS12: PIPO, PSME, LAOC

MOD WARM-MOD DRY (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS 1: LAOC, PICO, PIPO		
SAPLING - SMALL TREE	DS2: LAOC, PICO, PIPO, PSME		DS3: PICO, PSME, LAOC, ABGR
MEDIUM TREE	DS4: PIPO, PSME, LAOC	DS5: LAOC, PSME, PIPO, PICO	DS6: PICO, PSME, LAOC, ABGR
LARGE TREE	DS7: PIPO, LAOC, PSME	DS8: LAOC, PSME, PIPO, PICO	DS9: LAOC, ABGR, PSME, PICO
VERY LARGE TREE	DS10: PIPO, LAOC, PSME	DS11: LAOC, PSME, PIPO	DS12: ABGR, PSME, LAOC

MOD WARM-MOIST (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS 1: PICO, LAOC, PICO, PIMO		
SAPLING - SMALL TREE	DS2: PICO, LAOC, PSME, PIMO		DS3: PICO, PSME, LAOC, PIMO
MEDIUM TREE	DS4: LAOC, PIPO, PSME	DS5: LAOC, PSME, PIMO, PICO	DS6: PICO, PSME, LAOC, PIEN
LARGE TREE	DS7: LAOC, PIPO, PSME	DS8: LAOC, PSME, PIPO, PIMO, PICO	DS9: LAOC, PSME, PIEN, PIMO, PICO
VERY LARGE TREE	DS10: LAOC, PIPO, PSME	DS11: LAOC, PSME, PIPO, PIMO	DS12: ABGR, PSME, PIMO, PIEN

MOD COOL-MOIST (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS 1: PICO, LAOC, PIMO, BEPA, TSHE		
SAPLING - SMALL TREE	DS2: LAOC, PIMO, PSME, PICO		DS3: PICO, PSME, TSHE, BEPA
MEDIUM TREE	DS4: LAOC, PSME	DS5: PIMO, LAOC, PSME, PICO	DS6: PIMO, LAOC, PSME, PIEN
LARGE TREE	DS7: LAOC, PSME	DS8: PIMO, LAOC, PSME, PICO	DS9: PSME, LAOC, PIMO, TSHE, THPL
VERY LARGE TREE	DS10: LAOC, PSME	DS11: LAOC, PSME, PIMO	DS12: THPL, TSHE, PSME, ABGR, PIEN, ABLA

COOL-MOIST (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS 1: PICO, LAOC, PIMO, BEPA, PIPO		
SAPLING - SMALL TREE	DS2: LAOC, PIMO, PSME, PIPO		DS3: PICO, PSME, BEPA, PIMO
MEDIUM TREE	DS4: LAOC, PSME, PIPO	DS5: LAOC, PSME, PICO, PIMO	DS6: LAOC, PICO, PSME, PIMO
LARGE TREE	DS7: LAOC, PSME, PIPO	DS8: LAOC, PSME, PICO, PIMO	DS9: LAOC, PSME, PIMO, PIEN, PIEN
VERY LARGE TREE	DS10: LAOC, PSME, PIPO	DS11: LAOC, PSME, PIMO	DS12: ABLA, PSME, PIMO, ABGR, PIEN

COOL-MOD DRY (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS 1: PICO, LAOC, PIMO		
SAPLING - SMALL TREE	DS2: LAOC, PSME, PIMO, PICO		DS3: PICO, PIEN, PSME, PIMO
MEDIUM TREE	DS4: LAOC, PSME	DS5: PICO, PSME, LAOC, PIMO	DS6: PICO, PIEN, PSME, PIMO
LARGE TREE	DS7: LAOC, PSME	DS8: LAOC, PSME, PICO, PIMO	DS9: PSME, PIEN, PICO, PIMO, ABLA
VERY LARGE TREE	DS10: LAOC, PSME	DS11: LAOC, PSME, PIMO	DS12: ABLA, PIEN, PSME, PIMO

COLD (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS1: PICO, PIAL		
SAPLING - SMALL TREE	DS2: PIAL, PICO, PIEN		DS3: PICO, PIEN
MEDIUM TREE	DS4: PIAL, LALY	DS5: PIAL, LALY, PICO, PIEN	DS6: PICO, PIEN, ABLA, TSME
LARGE TREE	DS7: PIAL, LALY	DS8: PIAL, LALY, PICO, PIEN	DS9: PIEN, ABLA, TSME, PICO
VERY LARGE TREE	DS10: PIAL, LALY	DS11: PIAL, LALY, PIEN	DS12: ABLA, PIEN, TSME

TIMBERLINE (Canopy Cover)

TREE SIZECLASS	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS1: PIAL		
SAPLING - SMALL TREE	DS2: PIAL, LALY, PIEN		DS3: ABLA, PIEN
MEDIUM TREE	DS4: PIAL, LALY	DS5: PIAL, LALY, PIEN	DS6: ABLA, PIEN
LARGE TREE	DS7: PIAL, LALY	DS8: PIAL, LALY, PIEN	DS9: ABLA, PIEN
VERY LARGE TREE	DS10: PIAL, LALY	DS11: PIAL, LALY, PIEN	DS12: ABLA, PIEN

[a] OPEN = 10-39% canopy cover
MODERATE = ≥40-59% canopy
CLOSED = ≥60% canopy cover

[b] PIPO = *Pinus ponderosa*
PSME = *Psuedotsuga menziesii*
LAOC = *Larix occidentalis*
PICO = *Pinus contorta*
PIEN = *Picea engelmannii*
PIMO = *Pinus monticola*
ABGR = *Abies grandis*
ABLA = *Abies lasiocarpa*
TSHE = *Tsuga heterophylla*
LALY = *Larix lyallii*
PIAL = *Pinus albicaulis*
TSME= *Tsuga mertensia*
BEPA = *Betula papyrifera*

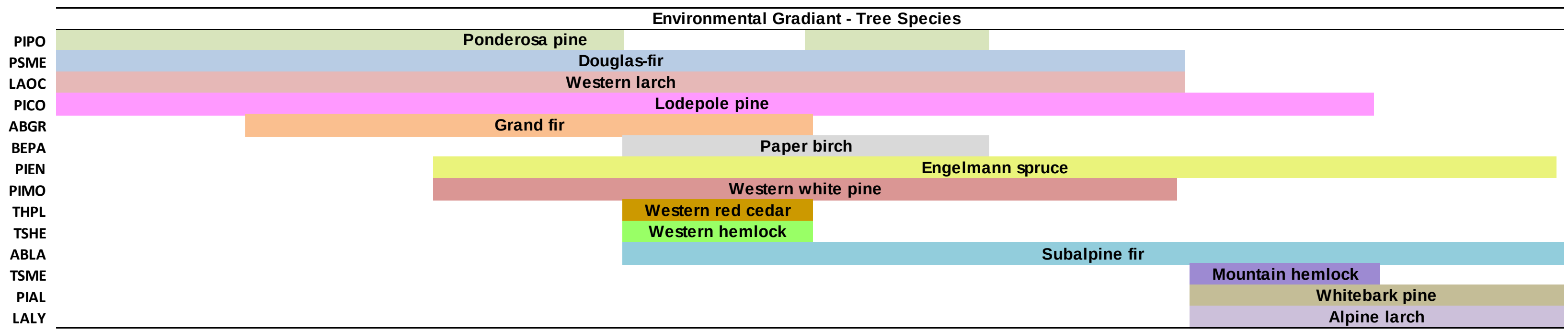


Figure 13. Historically occurring tree species distributions across the ecosystem diversity framework for ecoregion M333C based on the described methods.

TREE SIZECLASS	WARM-DRY Canopy Cover[a] Open / Moderate / Closed	MOD WARM-DRY Canopy Cover Open / Moderate / Closed	COOL-MOIST Canopy Cover Open / Moderate / Closed	COOL-MOD DRY Canopy Cover Open / Moderate / Closed	COLD Canopy Cover Open / Moderate / Closed	TIMBERLINE Canopy Cover Open / Moderate / Closed
GRASS-FORB-SHRUB-SEEDLING	DS1: PIPO	DS1: PIPO, LAOC, PICO	DS1: PICO, LAOC, PIPO	DS1: PICO, LAOC, PSME	DS1: PICO, PIAL	DS1: PIAL
SAPLING - SMALL TREE	DS2: PIPO, PSME, JUSC DS3: PIPO, PSME	DS2: PIPO, LAOC, PSME, PICO DS3: PIPO, LAOC, PSME, PICO	DS2: LAOC, PICO, PIPO, PSME DS3: PICO, PSME, PIEN, LAOC	DS2: LAOC, PSME, PICO DS3: PICO, PIEN, PSME	DS2: PIAL, PICO, PIEN DS3: PICO, PIEN	DS2: PIAL, LALY, PIEN DS3: ABLA, PIEN
MEDIUM TREE	DS4: PIPO, PSME, JUSC DS5: PIPO, PSME DS6: PIPO, PSME	DS4: PIPO, PSME, LAOC DS5: PIPO, LAOC, PSME, PICO DS6: PIPO, PSME, PICO, LAOC	DS4: LAOC, PSME, PIPO DS5: LAOC, PSME, PICO, PIPO DS6: PSME, PICO, LAOC, PIEN	DS4: LAOC, PSME DS5: PICO, LAOC, PSME DS6: PICO, PIEN, PSME, ABLA	DS4: PIAL, LALY DS5: PIAL, LALY, PICO, PIEN DS6: PICO, PIEN, ABLA	DS4: PIAL, LALY DS5: PIAL, LALY, PIEN DS6: ABLA, PIEN
LARGE TREE	DS7: PIPO, PSME, JUSC DS8: PIPO, PSME DS9: PIPO, PSME	DS7: PIPO, PSME, LAOC DS8: PIPO, LAOC, PSME, PICO DS9: PIPO, PSME, LAOC, PICO	DS7: LAOC, PSME, PIPO DS8: LAOC, PSME, PICO, PIPO DS9: LAOC, PSME, PIEN, ABLA	DS7: LAOC, PSME DS8: LAOC, PSME, PICO DS9: PSME, PIEN, PICO, ABLA	DS7: PIAL, LALY DS8: PIAL, LALY, PICO, PIEN DS9: PIEN, ABLA, PICO	DS7: PIAL, LALY DS8: PIAL, LALY, PIEN DS9: ABLA, PIEN, PIEN
VERY LARGE TREE	DS10: PIPO, PSME DS11: PIPO, PSME DS12: PIPO, PSME	DS10: PIPO, PSME, LAOC DS11: PIPO, LAOC, PSME DS12: PIPO, PSME, LAOC	DS10: LAOC, PSME, PIPO DS11: LAOC, PSME, PIPO DS12: ABLA, PSME, LAOC, PIEN	DS10: LAOC, PSME DS11: LAOC, PSME DS12: ABLA, PIEN, PSME	DS10: PIAL, LALY DS11: PIAL, LALY, PIEN DS12: ABAL, PIEN	DS10: PIAL, LALY DS11: PIAL, LALY, PIEN DS12: PIAL, ABLA, PIEN

[a] OPEN = 10-39% canopy cover
MODERATE = ≥40-59% canopy
CLOSED = ≥60% canopy cover

[b] PIPO = *Pinus ponderosa*
PSME = *Pseudotsuga menziesii*
LAOC = *Larix occidentalis*
PICO = *Pinus contorta*
PIEN = *Picea engelmannii*
ABLA = Abies lasiocarpa
LALY = *Larix lyallii*
PIAL = *Pinus albicaulis*
JUSC = *Juniperus scopulorum*

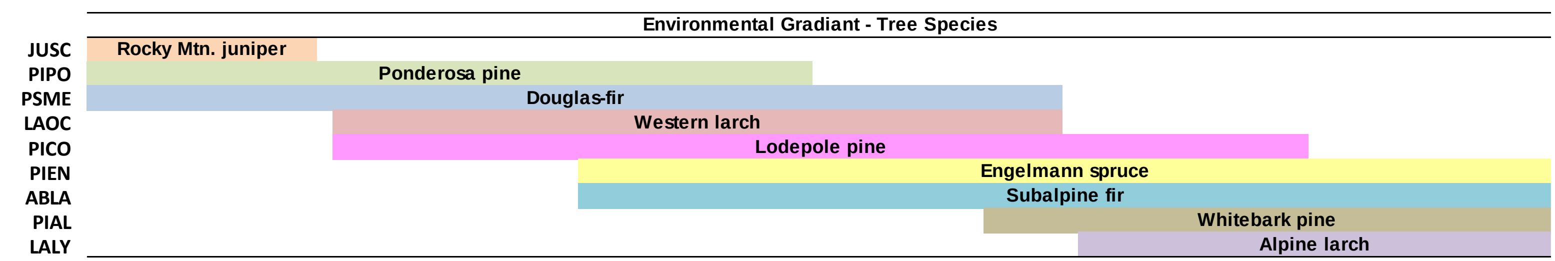


Figure 14. Historically occurring tree species distributions across the ecosystem diversity framework for ecoregion M332B-West.

TREE SIZECLASS	HOT-DRY	WARM-DRY	MOD WARM-DRY	COOL-MOIST	COOL-MOD DRY	COLD-MOD DRY TO MOIST	COLD-TIMBERLINE
Canopy Cover	Canopy Cover[a]: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed	Canopy Cover: Open / Moderate	Canopy Cover: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed
GRASS-FORB-SHRUB-SEEDLING	DS1: PIFL, PSME	DS1: PIPO	DS1: PIPO, PICO	DS1: PICO, PIPO, LAOC	DS1: PICO, PIPO	DS1: PICO, PIAL	DS1: PIAL
SAPLING - SMALL TREE	DS2: PIFL, PSME, JUSC DS3: PIFL, PSME	DS2: PIPO, PSME, JUSC DS3: PIPO, PSME	DS2: PIPO, PSME, PICO DS3: PIPO, PSME, PICO	DS2: PSME, PICO, PIPO, LAOC DS3: PICO, PSME, PIEN	DS2: PICO, PSME DS3: PICO, PIEN, PSME	DS2: PIAL, PICO, PIEN DS3: PIAL, PICO, PIEN	DS2: PIAL, PIEN DS3: PIAL, PIEN
MEDIUM TREE	DS4: PIFL, PSME, JUSC DS5: PIFL, PSME DS6: PIFL, PSME	DS4: PIPO, PSME, JUSC DS5: PIPO, PSME DS6: PIPO, PSME	DS4: PIPO, PSME DS5: PIPO, PSME, PICO DS6: PIPO, PSME, PICO	DS4: PSME, PIPO, LAOC DS5: PSME, PICO, PIPO, LAOC DS6: PSME, PICO, PIEN	DS4: PSME, PICO DS5: PICO, PSME, PIEN DS6: PICO, PIEN, PSME, ABLA	DS4: PIAL DS5: PIAL, PICO, PIEN DS6: PIAL, PICO, PIEN, ABLA	DS4 / DS5: PIAL, PIEN DS6: ABLA, PIEN
LARGE TREE	DS7: PIFL, PSME, JUSC DS8: PIFL, PSME DS9: PIFL, PSME	DS7: PIPO, PSME, JUSC DS8: PIPO, PSME DS9: PIPO, PSME	DS7: PIPO, PSME DS8: PIPO, PSME, PICO DS9: PIPO, PSME, PICO	DS7: PSME, PIPO, LAOC DS8: PSME, PICO, LAOC, PIPO DS9: PSME, PIEN, ABLA	DS7: PSME DS8: PSME, PICO, PIEN DS9: PSME, PIEN, PICO, ABLA	DS7: PIAL DS8: PIAL, PICO DS9: PIAL, PICO, PIEN, ABLA	DS4: PIAL DS5: PIAL, PIEN DS6: ABLA, PIEN
VERY LARGE TREE	DS10: PIFL, PSME DS11: PIFL, PSME DS12: PIFL, PSME	DS10: PIPO, PSME DS11: PIPO, PSME DS12: PIPO, PSME	DS10: PIPO, PSME DS11: PIPO, PSME DS12: PIPO, PSME	DS10: PSME, PIPO, LAOC DS11: PSME, LAOC, PIPO DS12: ABLA, PSME, PIEN	DS10: PSME, PIPO DS11: PSME, PIPO, PIEN DS12: ABLA, PIEN, PSME	DS10: PIAL DS11: PIAL, PICO DS12: PIAL, ABLA, PIEN	DS7: PIAL DS8: PIAL, PIEN DS9: ABLA, PIEN

[a] OPEN = 10-39% canopy cover
MODERATE = ≥40-59% canopy cover
CLOSED = ≥60% canopy cover

[b] PIPO = *Pinus ponderosa*
PSME = *Pseudotsuga menziesii*
LAOC = *Larix occidentalis*
PICO = *Pinus contorta*
PIEN = *Picea engelmannii*
ABLA = Abies lasiocarpa
PIAL = *Pinus albicaulis*
JUSC = *Juniperus scopulorum*
PIFL = *Pinus flexilus*

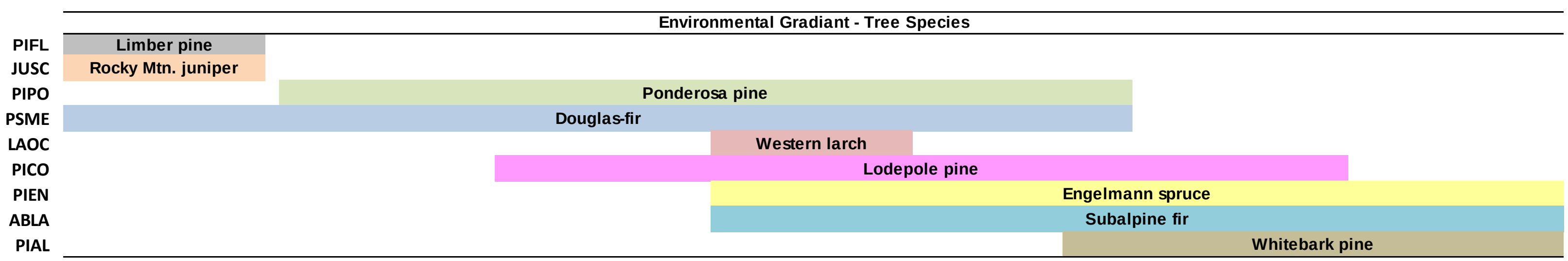


Figure 15. Historically occurring tree species distributions across the ecosystem diversity framework for ecoregion M332-East.

Table 17. Moderately Warm and Dry ecological site - Historically occurring herbaceous and shrub species distribution by disturbance state based on the described methods.

Common name	Scientific name	PLANTS Code[a]	Lifeform	DS1	DS2	DS3	DS4	DS5	DS6	DS7	DS8	DS9	DS10	DS11	DS12
Greene's mountain ash	*Sorbus scopulina*	SOSCS	Shrub/Tree	X	X	X		X	X		X	X		X	X
Antelope bitterbrush	*Purshia tridentata*	PUTR2	Shrub				X			X			X		
big sagebrush	*Artemisia tridentata*	ARTRV	Shrub				X			X			X		
chokecherry	*Prunus virginiana*	PRVI	Shrub	X	X			X			X			X	
common juniper	*Juniperus communis*	JUCO6	Shrub				X	X		X	X		X	X	
common snowberry	*Symphoricarpos albus*	SYAL	Shrub				X	X		X	X		X	X	
creeping juniper	*Juniperus horizontalis*	JUHO2	Shrub				X	X		X	X		X	X	
grouse whortleberry	*Vaccinium scoparium*	VASC	Shrub						X			X			X
kinnikinnick	*Arctostaphylos uva-ursi*	ARUV	Shrub					X	X		X	X		X	X
Lewis's mock orange	*Philadelphus lewisii*	PHLE4	Shrub	X	X		X			X			X		
mallow ninebark	*Physocarpus malviceus*	PHMA5	Shrub						X			X			X
oceanspray	*Holodiscus discolor*	HODI	Shrub	X	X		X	X		X	X		X	X	
Oregon boxleaf	*Pachistima myrsinites*	PAMY	Shrub						X			X			X
Oregon grape	*Berberis repens*	BERE	Shrub						X			X			X
prickly currant	*Ribes lacustre*	RILA	Shrub	X	X	X			X			X			X
prickly rose	*Rosa acicularis*	ROAC	Shrub						X			X			X
pygmy rose	*Rosa bridgesii*	ROBR3	Shrub												
redosier dogwood	*Cornus sericea*	COCA13	Shrub				X			X			X		
Rocky Mountain maple	*Acer glabrum*	ACGL	Shrub	X	X	X		X	X		X	X		X	X
russet buffaloberry	*Shepherdia canadensis*	SHCA	Shrub				X	X		X	X		X	X	
rusty menziesia	*Menziesia ferruginea*	MEFE	Shrub						X			X			X
Saskatoon serviceberry	*Amelanchier alnifolia*	AMAL2	Shrub	X	X	X		X			X			X	
Scouler's willow	*Salix scouleriana*	SASC	Shrub	X	X			X			X			X	
shrubby cinquefoil	*Dasiphora fruticosa*	DAFR6	Shrub												
Sitka alder	*Alnus viridis spp sinuata*	ALVIS	Shrub	X	X	X		X			X			X	
snowbrush ceanothus	*Ceanothus velutinus*	CEVE	Shrub	X	X		X			X			X		
sticky currant	*Ribes viscosissimum*	RIVI3	Shrub	X	X		X			X			X		

Table 17, continued. Moderately Warm and Dry ecological site - Historically occurring herbaceous and shrub species distribution by disturbance state.

Common name	Scientific name	PLANTS Code[a]	Lifeform	DS1	DS2	DS3	DS4	DS5	DS6	DS7	DS8	DS9	DS10	DS11	DS12
thimbleberry	*Rubus parviflorus*	RUPA	Shrub	X	X	X		X	X		X	X		X	X
thinleaf huckleberry	*Vaccinium membranaceum*	VAME	Shrub						X			X			X
twinflower	*Linnaea borealis*	LIBO3	Shrub					X	X		X	X		X	X
Utah honeysuckle	*Lonicera utahensis*	LOUT2	Shrub					X	X		X	X		X	X
white spiraea	*Spiraea betulifolia*	SPBE2	Shrub	X	X	X			X			X			X
Wood's rose	*Rosa woodsii*	ROWO	Shrub					X			X			X	
Geyer's sedge	*Carex geyeri*	CAGE2	Sedge	X	X	X		X	X		X	X		X	X
northwestern sedge	*Carex concinnoides*	CACO11	Sedge					X			X			X	
Ross's sedge	*Carex rossi*	CARO5	Sedge	X	X			X			X			X	
blue wildrye	*Elymus glaucus*	ELGL	Grass				X	X		X	X		X	X	
bluebunch wheatgrass	*Pseudoroegneria spicata*	PSSPS	Grass				X			X			X		
Idaho fescue	*Festuca idahoensis*	FEID	Grass					X	X		X	X		X	X
pinegrass	*Calamagrostis rubescens*	CARU	Grass						X			X			X
prairie Junegrass	*Koeleria macrantha*	KOMA	Grass	X	X	X	X	X		X	X		X	X	
rough fescue	*Festuca campestris*	FECA4	Grass				X	X		X	X		X	X	
western fescue	*Festuca occidentalis*	FEOC	Grass					X	X		X	X		X	X
Wheeler bluegrass	*Poa nervosa*	PONE2	Grass				X	X		X	X		X	X	
Alberta beardtongue	*Penstemon albertinus*	PEAL11	Forb					X			X			X	
arrowleaf balsamroot	*Balsamorhiza sagittata*	BASA3	Forb	X	X		X	X		X	X		X	X	
aspen fleabane	*Erigeron speciosus*	ERSP4	Forb												
ballhead sandwort	*Arenaria congesta*	ARCO5	Forb					X	X		X	X		X	X
Bonneville shootingstar	*Dodecatheon conjugens*	DOCO	Forb												
bride's bonnet	*Clintonia uniflora*	CLUN2	Forb					X	X		X	X		X	X
broadleaf arnica	*Arnica latifolia*	ARLA8	Forb												
common beargrass	*Xerophyllum tenax*	XETE	Forb					X	X		X	X		X	X
common gaillardia	*Gaillardia aristata*	GAAR	Forb	X	X		X			X			X		
common yarrow	*Achillea millefolium*	ACMI2	Forb	X	X	X		X			X			X	

Table 17, continued. Moderately Warm and Dry ecological site - Historically occurring herbaceous and shrub species distribution by disturbance state.

Common name	Scientific name	PLANTS Code[a]	Lifeform	DS1	DS2	DS3	DS4	DS5	DS6	DS7	DS8	DS9	DS10	DS11	DS12
dwarf bilberry	*Vaccinium cespitosum*	VACE	Forb						X			X			X
elegant piperia	*Piperia elegans*	PIELE4	Forb												
feathery false lily of the valley	*Maianthemum racemosum*	MARAA	Forb												
fireweed	*Chamerion angustifolium*	CHANA2	Forb	X	X		X	X		X	X		X	X	
harebell	*Campanula rotundifolia*	CARO2	Forb					X			X			X	
heartleaf arnica	*Arnica cordifolia*	ARCO9	Forb					X	X		X	X		X	X
Holboell's rockcress	*Arabis holboellii*	ARHO2	Forb					X			X			X	
hookedspur violet	*Viola adunca*	VIAD	Forb												
Howell's pussytoes	*Antennaria howellii*	ANHOH	Forb					X			X			X	
maiden blue eyed Mary	*Collinsia parviflora*	COPA3	Forb	X	X	X			X			X			X
marsh valerian	*Valeriana dioica*	VADI	Forb												
Menzie's campion	*Silene menziesii*	SIME	Forb												
Missouri goldenrod	*Solidago missouriensis*	SOMI2	Forb	X	X		X	X		X	X		X	X	
Mountain deathcamas	*Zigadenus elegans*	ZIEL2	Forb					X	X		X	X		X	X
narrowleaf mountain trumpet	*Collomia linearis*	COLI2	Forb					X			X			X	
nineleaf biscuitroot	*Lomatium triternatum*	LOTR2	Forb					X			X			X	
nodding onion	*Allium cernuum*	ALCE2	Forb				X			X			X		
northern bedstraw	*Galium boreale*	GABO2	Forb					X			X			X	
pipsissewa	*Chimaphila umbellata*	CHUM	Forb						X			X			X
pointed tip mariposa lily	*Calochortus apiculatus*	CAAP	Forb					X			X			X	
prairie smoke	*Geum triflorum*	GETR	Forb						X			X			X
raceme pussytoes	*Antennaria racemosa*	ANRA	Forb						X			X			X
red baneberry	*Actaea rubra*	ACRU2	Forb						X			X			X
rock clematis	*Clematis columbiana*	CLPS2	Forb					X			X			X	
rosy pussytoes	*Antennaria rosea*	ANRO2	Forb				X			X			X		
roughfruit fairybells	*Prosartes trachycarpa*	PRTR4	Forb												
roundleaf alumroot	*Heuchera cylindrica*	HECY2	Forb					X			X			X	

Table 17, continued. Moderately Warm and Dry ecological site - Historically occurring herbaceous and shrub species distribution by disturbance state.

Common name	Scientific name	PLANTS Code[a]	Lifeform	DS1	DS2	DS3	DS4	DS5	DS6	DS7	DS8	DS9	DS10	DS11	DS12
Scouler's woollyweed	*Hieracium scouleri*	HISCA	Forb				X			X			X		
sidebells wintergreen	*Orthilia secunda*	ORSE	Forb												
silky lupine	*Lupinus sericeus*	LUSE4	Forb				X	X		X	X		X	X	
spreading dogbane	*Apocynum androsaemifolium*	APAN2	Forb	X	X	X			X			X			X
starry false lily of the valley	*Maianthemum stellatum*	MAST4	Forb												
sticky purple cinquefoil	*Potentilla glandulosa*	POGL9	Forb					X			X			X	
sticky purple geranium	*Geranium viscosissimum*	GEVI2	Forb	X	X	X			X			X			X
sweetcicely	*Osmorhiza berteroi*	OSBE	Forb												
timber milkvetch	*Astragalus miser*	ASMI9	Forb												
Virginia strawberry	*Fragaria virginiana*	FRVI	Forb				X	X		X	X		X	X	
western meadow-rue	*Thalictrum occidentale*	THOC	Forb						X			X			X
western rattlesnake plantain	*Goodyera oblongifolia*	GOOB2	Forb												
western showy aster	*Eurybia conspicua*	EUCO36	Forb												
western stoneseed	*Lithospermum ruderale*	LIRU4	Forb				X			X			X		
white hawkweed	*Hieracium albiflorum*	HIAL2	Forb												
white sweetvetch	*Hedysarum sulphurescens*	HESU	Forb	X	X			X			X			X	
wild sarsaparilla	*Aralia nudicualis*	ARNU2	Forb						X			X			X
woodland strawberry	*Fragaria vesca*	FRVE	Forb				X	X		X	X		X	X	
wormleaf stonecrop	*Sedum stenopetalum*	SEST2	Forb						X			X			X
yellow avalanche-lily	*Erythronium grandiflorum*	ERGR9	Forb												

Table 18. Moderately Warm and Dry ecological site in the M333C ecoregion listing the average, minimum, and maximum live trees per acre by DBH range within each disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA – data not available.

SIZE-CLASS	DBH RANGE	OPEN (<40% Canopy Cover) AVG (MIN-MAX)	OPEN #PLOTS	MODERATE (40-60% Canopy Cover) AVG (MIN-MAX)	MODERATE #PLOTS	CLOSED (>60% Canopy Cover) AVG (MIN-MAX)	CLOSED #PLOTS
GRASS-FORB-SHRUB-SEEDLING		DS1					
	<1.0"	9633 (0 - 19266)	2				
	1.0-4.9"	0 (0 - 0)					
	5.0-14.9"	6 (0 - 12)					
	15.0-19.9"	0 (0 - 0)					
	20.0+"	3 (0 - 6)					
SAPLING/ SMALL		DS2				DS3	
	<1.0"	1499 (1499 - 1499)	1			NA	0
	1.0-4.9"	225 (225 - 225)					
	5.0-14.9"	0 (0 - 0)					
	15.0-19.9"	0 (0 - 0)					
	20.0+"	0 (0 - 0)					
MEDIUM		DS4		DS5		DS6	
	<1.0"	931 (0 - 2657)	17	100 (0 - 375)	6	218 (0 - 900)	5
	1.0-4.9"	177 (0 - 686)		300 (0 - 600)		748 (75 - 1414)	
	5.0-14.9"	87 (19 - 224)		203 (138 - 265)		449 (200 - 683)	
	15.0-19.9"	2 (0 - 8)		1 (0 - 6)		1 (0 - 6)	
	20.0+"	1 (0 - 6)		0 (0 - 2)		1 (0 - 6)	
LARGE		DS7		DS8		DS9	
	<1.0"	557 (0 - 3523)	8	259 (0 - 1071)	10	257 (0 - 514)	2
	1.0-4.9"	25 (0 - 86)		115 (0 - 343)		21 (0 - 43)	
	5.0-14.9"	57 (30 - 165)		152 (90 - 235)		293 (231 - 355)	
	15.0-19.9"	16 (6 - 24)		21 (12 - 30)		13 (12 - 15)	
	20.0+"	3 (0 - 6)		2 (0 - 6)		0 (0 - 0)	
VERY LARGE		DS10		DS11		DS12	
	<1.0"	536 (300 - 771)	2	705 (0 - 2186)	6	75 (75 - 75)	1
	1.0-4.9"	193 (86 - 300)		171 (0 - 900)		0 (0 - 0)	
	5.0-14.9"	19 (15 - 24)		72 (12 - 139)		108 (108 - 108)	
	15.0-19.9"	13 (12 - 14)		15 (2 - 26)		42 (42 - 42)	
	20.0+"	11 (10 - 13)		19 (10 - 30)		24 (24 - 24)	

Table 19. Moderately Warm and Dry ecological site in the M333C ecoregion listing average, minimum, and maximum values for ecosystem structural characteristics by disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA = data not available, BA WTD DIA= Basal area weighted diameter and CWD = coarse woody debris.

SIZE-CLASS	ECOSYSTEM CHARACTERISTIC	OPEN (<40% Canopy Cover)		MODERATE (40-60% Canopy Cover)		CLOSED (>60% Canopy Cover)	
GRASS-FORB-SHRUB-SEEDLING		DS1					
		AVG (MIN-MAX)	#PLOTS				
	BA WTD DIA	21 (11 - 31)	2				
	CWD (TONS/AC)	NA	0				
	FORBS (CC%)	9 (7 - 11)	2				
	GRASS (CC%)	6 (2 - 11)	2				
	SHRUBS (CC%)	5 (3 - 7)	2				
SAPLING/ SMALL		DS2				DS3	
		AVG (MIN-MAX)	#PLOTS			AVG (MIN-MAX)	#PLOTS
	BA WTD DIA	1 (1 - 1)	1			NA	0
	CWD (TONS/AC)	NA	0			NA	0
	FORBS (CC%)	12 (12 - 12)	1			NA	0
	GRASS (CC%)	15 (15 - 15)	1			NA	0
	SHRUBS (CC%)	14 (14 - 14)	1			NA	0
MEDIUM		DS4		DS5		DS6	
		AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS
	BA WTD DIA	10 (5 - 25)	17	9 (6 - 10)	6	7 (6 - 9)	5
	CWD (TONS/AC)	5 (0 - 14)	3	NA	0	NA	0
	FORBS (CC%)	8 (1 - 19)	7	9 (2 - 23)	5	16 (9 - 23)	2
	GRASS (CC%)	8 (3 - 16)	7	6 (1 - 14)	5	1 (1 - 1)	2
	SHRUBS (CC%)	10 (2 - 17)	7	20 (10 - 52)	5	23 (15 - 32)	2
LARGE		DS7		DS8		DS9	
	BA WTD DIA	15 (10 - 17)	8	13 (10 - 15)	10	11 (11 - 12)	2
	CWD (TONS/AC)	24 (24 - 24)	1	0 (0 - 0)	1	10 (0 - 20)	2
	FORBS (CC%)	14 (5 - 33)	6	11 (2 - 21)	5	NA	0
	GRASS (CC%)	11 (2 - 21)	6	6 (1 - 15)	5	NA	0
	SHRUBS (CC%)	15 (4 - 29)	6	25 (7 - 69)	5	NA	0
VERY LARGE		DS10		DS11		DS12	
	BA WTD DIA	18 (18 - 19)	2	19 (15 - 25)	6	21 (21 - 21)	1
	CWD (TONS/AC)	0 (0 - 0)	1	3 (0 - 7)	2	NA	0
	FORBS (CC%)	NA	0	15 (12 - 17)	2	10 (10 - 10)	1
	GRASS (CC%)	NA	0	7 (6 - 9)	2	6 (6 - 6)	1
	SHRUBS (CC%)	NA	0	34 (20 - 47)	2	15 (15 - 15)	1

Table 20. Moderately Warm and Dry ecological site in the M333C ecoregion listing average, minimum, and maximum dead trees per acre by DBH range within each disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA – data not available.

SIZE-CLASS	DBH RANGE	OPEN (<40% Canopy Cover)		MODERATE (40-60% Canopy Cover)		CLOSED (>60% Canopy Cover)	
GRASS-FORB-SHRUB-SEEDLING		DS1					
		AVG (MIN-MAX)	#PLOTS				
	<1.0"	0 (0 - 0)					
	1.0-4.9"	0 (0 - 0)					
	5.0-14.9"	42 (24 - 60)	2				
	15.0-19.9"	12 (6 - 18)					
	20.0+"	9 (0 - 18)					
SAPLING/ SMALL		DS2				DS3	
		AVG (MIN-MAX)	#PLOTS			AVG (MIN-MAX)	#PLOTS
	<1.0"	0 (0 - 0)					
	1.0-4.9"	0 (0 - 0)					
	5.0-14.9"	48 (48 - 48)	1			NA	0
	15.0-19.9"	18 (18 - 18)					
	20.0+"	30 (30 - 30)					
MEDIUM		DS4		DS5		DS6	
		AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS
	<1.0"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	1.0-4.9"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	5.0-14.9"	16 (0 - 175)	17	5 (0 - 18)	6	12 (0 - 42)	5
	15.0-19.9"	2 (0 - 14)		0 (0 - 3)		2 (0 - 6)	
	20.0+"	1 (0 - 6)		0 (0 - 0)		2 (0 - 6)	
LARGE		DS7		DS8		DS9	
	<1.0"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	1.0-4.9"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	5.0-14.9"	48 (0 - 223)	8	29 (0 - 102)	10	110 (95 - 125)	2
	15.0-19.9"	5 (0 - 18)		3 (0 - 12)		0 (0 - 0)	
	20.0+"	1 (0 - 6)		0 (0 - 0)		0 (0 - 0)	
VERY LARGE		DS10		DS11		DS12	
	<1.0"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	1.0-4.9"	0 (0 - 0)		17 (0 - 100)		0 (0 - 0)	
	5.0-14.9"	19 (9 - 29)	2	14 (0 - 32)	6	18 (18 - 18)	1
	15.0-19.9"	1 (0 - 3)		1 (0 - 6)		0 (0 - 0)	
	20.0+"	2 (0 - 5)		2 (0 - 6)		0 (0 - 0)	

Table 21. Moderately Warm and Dry ecological site in the M332B-West ecoregion listing average, minimum, and maximum live trees per acre by DBH range within each disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA – data not available.

SIZE-CLASS	DBH RANGE	OPEN (<40% Canopy Cover) AVG (MIN-MAX)	#PLOTS	MODERATE (40-60% Canopy Cover) AVG (MIN-MAX)	#PLOTS	CLOSED (>60% Canopy Cover) AVG (MIN-MAX)	#PLOTS
GRASS-FORB-SHRUB-SEEDLING		DS1					
		AVG (MIN-MAX)	#PLOTS				
	<1.0"	229 (0 - 1199)	7				
	1.0-4.9"	0 (0 - 0)					
	5.0-14.9"	5 (0 - 12)					
	15.0-19.9"	1 (0 - 3)					
	20.0+"	1 (0 - 3)					
SAPLING/ SMALL		DS2				DS3	
		AVG (MIN-MAX)	#PLOTS			AVG (MIN-MAX)	#PLOTS
	<1.0"	915 (75 - 1860)	4			NA	0
	1.0-4.9"	247 (75 - 540)					
	5.0-14.9"	3 (0 - 6)					
	15.0-19.9"	1 (0 - 5)					
	20.0+"	2 (0 - 6)					
MEDIUM		DS4		DS5		DS6	
		AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS
	<1.0"	433 (0 - 5772)	89	680 (0 - 5847)	25	220 (0 - 1260)	15
	1.0-4.9"	121 (0 - 900)		306 (0 - 1140)		838 (180 - 4100)	
	5.0-14.9"	80 (12 - 252)		198 (102 - 371)		399 (238 - 638)	
	15.0-19.9"	2 (0 - 9)		3 (0 - 9)		2 (0 - 8)	
	20.0+"	0 (0 - 6)		1 (0 - 4)		0 (0 - 5)	
LARGE		DS7		DS8		DS9	
	<1.0"	659 (0 - 5940)	46	845 (0 - 9820)	30	733 (0 - 4920)	14
	1.0-4.9"	51 (0 - 500)		143 (0 - 750)		206 (0 - 600)	
	5.0-14.9"	52 (0 - 120)		146 (60 - 271)		270 (60 - 527)	
	15.0-19.9"	14 (4 - 24)		22 (6 - 38)		24 (10 - 78)	
	20.0+"	3 (0 - 7)		2 (0 - 7)		2 (0 - 7)	
VERY LARGE		DS10		DS11		DS12	
	<1.0"	236 (0 - 2100)	22	417 (0 - 3420)	20	660 (0 - 2100)	6
	1.0-4.9"	64 (0 - 420)		86 (0 - 540)		309 (0 - 840)	
	5.0-14.9"	33 (0 - 84)		73 (12 - 189)		115 (37 - 205)	
	15.0-19.9"	11 (0 - 35)		18 (0 - 43)		39 (14 - 62)	
	20.0+"	13 (8 - 18)		16 (9 - 25)		17 (9 - 28)	

Table 22. Moderately Warm and Dry ecological site in the M332B-West ecoregion listing the average, minimum, and maximum dead trees per acre by DBH range within each disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA – data not available.

SIZE-CLASS	DBH RANGE	OPEN (<40% Canopy Cover)		MODERATE (40-60% Canopy Cover)		CLOSED (>60% Canopy Cover)	
GRASS-FORB-SHRUB-SEEDLING		DS1					
		AVG (MIN-MAX)	#PLOTS				
	<1.0"	0 (0 - 0)	7				
	1.0-4.9"	0 (0 - 0)					
	5.0-14.9"	5 (0 - 30)					
	15.0-19.9"	1 (0 - 6)					
	20.0+"	2 (0 - 15)					
SAPLING/ SMALL		DS2				DS3	
		AVG (MIN-MAX)	#PLOTS			AVG (MIN-MAX)	#PLOTS
	<1.0"	0 (0 - 0)	4			NA	0
	1.0-4.9"	0 (0 - 0)					
	5.0-14.9"	8 (0 - 30)					
	15.0-19.9"	2 (0 - 6)					
	20.0+"	0 (0 - 0)					
MEDIUM		DS4		DS5		DS6	
		AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS
	<1.0"	0 (0 - 0)	89	0 (0 - 0)	25	0 (0 - 0)	15
	1.0-4.9"	0 (0 - 0)		32 (0 - 300)		33 (0 - 500)	
	5.0-14.9"	15 (0 - 199)		8 (0 - 64)		27 (0 - 107)	
	15.0-19.9"	1 (0 - 30)		1 (0 - 17)		0 (0 - 0)	
	20.0+"	1 (0 - 12)		0 (0 - 6)		0 (0 - 0)	
LARGE		DS7		DS8		DS9	
	<1.0"	0 (0 - 0)	46	0 (0 - 0)	30	0 (0 - 0)	14
	1.0-4.9"	2 (0 - 100)		7 (0 - 100)		43 (0 - 600)	
	5.0-14.9"	11 (0 - 132)		21 (0 - 95)		16 (0 - 54)	
	15.0-19.9"	2 (0 - 24)		1 (0 - 14)		1 (0 - 6)	
	20.0+"	1 (0 - 18)		0 (0 - 6)		0 (0 - 1)	
VERY LARGE		DS10		DS11		DS12	
	<1.0"	0 (0 - 0)	22	0 (0 - 0)	20	0 (0 - 0)	6
	1.0-4.9"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	5.0-14.9"	8 (0 - 42)		8 (0 - 42)		7 (0 - 29)	
	15.0-19.9"	1 (0 - 18)		1 (0 - 12)		4 (0 - 11)	
	20.0+"	1 (0 - 7)		1 (0 - 6)		2 (0 - 7)	

Table 23. Moderately Warm and Dry ecological site in the M332B-West ecoregion listing average, minimum, and maximum values for ecosystem structural characteristics by disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA = data not available, BA WTD DIA= Basal area weighted diameter and CWD = coarse woody debris.

SIZE-CLASS	ECOSYSTEM CHARACTERISTIC	OPEN (<40% Canopy Cover) AVG (MIN-MAX)	#PLOTS	MODERATE (40-60% Canopy Cover) AVG (MIN-MAX)	#PLOTS	CLOSED (>60% Canopy Cover) AVG (MIN-MAX)	#PLOTS
GRASS-FORB-SHRUB-SEEDLING		DS1					
	BA WTD DIA			18 (11 - 38)	7		
	CWD (TONS/AC)			NA	0		
	FORBS (CC%)			6 (3 - 12)	3		
	GRASS (CC%)			11 (1 - 19)	3		
	SHRUBS (CC%)			23 (2 - 63)	3		
SAPLING/ SMALL		DS2				DS3	
	BA WTD DIA	9 (1 - 22)	4			NA	0
	CWD (TONS/AC)	4 (4 - 4)	1			NA	0
	FORBS (CC%)	10 (3 - 16)	3			NA	0
	GRASS (CC%)	17 (3 - 32)	3			NA	0
	SHRUBS (CC%)	24 (4 - 35)	3			NA	0
MEDIUM		DS4		DS5		DS6	
	BA WTD DIA	10 (4 - 20)	89	9 (5 - 13)	25	7 (4 - 9)	15
	CWD (TONS/AC)	19 (19 - 19)	1	30 (30 - 30)	1	15 (15 - 15)	1
	FORBS (CC%)	7 (1 - 36)	54	6 (1 - 18)	6	7 (5 - 11)	5
	GRASS (CC%)	12 (1 - 36)	54	25 (1 - 72)	6	6 (1 - 17)	5
	SHRUBS (CC%)	14 (0 - 62)	54	14 (1 - 35)	6	18 (4 - 50)	5
LARGE		DS7		DS8		DS9	
	BA WTD DIA	15 (11 - 20)	46	13 (10 - 16)	30	12 (9 - 15)	14
	CWD (TONS/AC)	7 (0 - 37)	5	0 (0 - 0)	1	0 (0 - 0)	1
	FORBS (CC%)	7 (1 - 51)	23	6 (1 - 20)	12	5 (1 - 8)	4
	GRASS (CC%)	18 (1 - 45)	23	16 (1 - 33)	12	15 (1 - 28)	4
	SHRUBS (CC%)	15 (1 - 38)	23	16 (2 - 30)	12	17 (3 - 40)	4
VERY LARGE		DS10		DS11		DS12	
	BA WTD DIA	19 (15 - 27)	22	18 (13 - 32)	20	16 (12 - 19)	6
	CWD (TONS/AC)	0 (0 - 0)	2	NA	0	24 (24 - 24)	1
	FORBS (CC%)	8 (1 - 21)	12	5 (2 - 10)	8	9 (6 - 11)	2
	GRASS (CC%)	23 (4 - 80)	12	17 (7 - 26)	8	14 (6 - 22)	2
	SHRUBS (CC%)	12 (1 - 31)	12	20 (2 - 44)	8	7 (4 - 10)	2

Table 24. Moderately Warm and Dry ecological site in the M332B-East ecoregion listing the average, minimum, and maximum live trees per acre by DBH range within each disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA – data not available.

SIZE-CLASS	DBH RANGE	OPEN (<40% Canopy Cover) AVG (MIN-MAX)	#PLOTS	MODERATE (40-60% Canopy Cover) AVG (MIN-MAX)	#PLOTS	CLOSED (>60% Canopy Cover) AVG (MIN-MAX)	#PLOTS
GRASS-FORB-SHRUB-SEEDLING		DS1					
	<1.0"	928 (0 - 5397)	8				
	1.0-4.9"	0 (0 - 0)					
	5.0-14.9"	5 (0 - 12)					
	15.0-19.9"	0 (0 - 4)					
	20.0+"	0 (0 - 0)					
SAPLING/ SMALL		DS2				DS3	
	<1.0"	546 (0 - 1124)	7			NA	0
	1.0-4.9"	464 (100 - 1050)					
	5.0-14.9"	1 (0 - 6)					
	15.0-19.9"	0 (0 - 0)					
	20.0+"	0 (0 - 0)					
MEDIUM		DS4		DS5		DS6	
	<1.0"	782 (0 - 5248)	56	1312 (0 - 11700)	26	526 (0 - 3298)	28
	1.0-4.9"	107 (0 - 975)		186 (0 - 750)		720 (0 - 2999)	
	5.0-14.9"	83 (12 - 241)		199 (113 - 336)		366 (201 - 523)	
	15.0-19.9"	1 (0 - 6)		2 (0 - 6)		1 (0 - 8)	
	20.0+"	1 (0 - 6)		0 (0 - 6)		0 (0 - 6)	
LARGE		DS7		DS8		DS9	
	<1.0"	871 (0 - 2849)	16	745 (0 - 7721)	24	676 (0 - 5640)	25
	1.0-4.9"	76 (0 - 375)		130 (0 - 1124)		496 (0 - 2460)	
	5.0-14.9"	54 (0 - 96)		134 (48 - 216)		261 (144 - 445)	
	15.0-19.9"	21 (8 - 48)		22 (5 - 48)		23 (6 - 72)	
	20.0+"	1 (0 - 6)		3 (0 - 7)		2 (0 - 6)	
VERY LARGE		DS10		DS11		DS12	
	<1.0"	1065 (0 - 2324)	7	2404 (0 - 20091)	13	455 (0 - 4273)	12
	1.0-4.9"	86 (0 - 600)		105 (0 - 500)		198 (0 - 900)	
	5.0-14.9"	38 (0 - 66)		74 (24 - 144)		199 (72 - 313)	
	15.0-19.9"	17 (0 - 24)		19 (0 - 48)		24 (4 - 48)	
	20.0+"	16 (9 - 24)		15 (8 - 24)		18 (8 - 36)	

Table 25. Moderately Warm and Dry ecological site in the M332B-East ecoregion listing the average, minimum, and maximum dead trees per acre by DBH range within each disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA – data not available.

GRASS-FORB-SHRUB-SEEDLING — OPEN / MODERATE / CLOSED: DS1

SIZE-CLASS	DBH RANGE	DS1 AVG (MIN-MAX)	#PLOTS
GRASS-FORB-SHRUB-SEEDLING	<1.0"	0 (0 - 0)	
	1.0-4.9"	0 (0 - 0)	
	5.0-14.9"	102 (0 - 457)	8
	15.0-19.9"	13 (0 - 96)	
	20.0+"	0 (0 - 0)	

SAPLING/ SMALL — OPEN / MODERATE: DS2; CLOSED: DS3

SIZE-CLASS	DBH RANGE	DS2 AVG (MIN-MAX)	#PLOTS	DS3 AVG (MIN-MAX)	#PLOTS
SAPLING/ SMALL	<1.0"	0 (0 - 0)			
	1.0-4.9"	0 (0 - 0)			
	5.0-14.9"	7 (0 - 36)	7	NA	0
	15.0-19.9"	3 (0 - 18)			
	20.0+"	1 (0 - 6)			

SIZE-CLASS	DBH RANGE	OPEN (<40% Canopy Cover) DS4 AVG (MIN-MAX)	#PLOTS	MODERATE (40-60% Canopy Cover) DS5 AVG (MIN-MAX)	#PLOTS	CLOSED (>60% Canopy Cover) DS6 AVG (MIN-MAX)	#PLOTS
MEDIUM	<1.0"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	1.0-4.9"	5 (0 - 300)		15 (0 - 400)		43 (0 - 900)	
	5.0-14.9"	50 (0 - 481)	56	73 (0 - 457)	26	34 (0 - 144)	28
	15.0-19.9"	3 (0 - 48)		1 (0 - 24)		0 (0 - 2)	
	20.0+"	0 (0 - 2)		0 (0 - 2)		0 (0 - 0)	

SIZE-CLASS	DBH RANGE	DS7		DS8		DS9	
LARGE	<1.0"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	1.0-4.9"	0 (0 - 0)		0 (0 - 0)		8 (0 - 100)	
	5.0-14.9"	40 (0 - 337)	16	50 (0 - 313)	24	38 (0 - 168)	25
	15.0-19.9"	2 (0 - 24)		2 (0 - 12)		1 (0 - 18)	
	20.0+"	1 (0 - 12)		0 (0 - 1)		0 (0 - 2)	

SIZE-CLASS	DBH RANGE	DS10		DS11		DS12	
VERY LARGE	<1.0"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	1.0-4.9"	0 (0 - 0)		0 (0 - 0)		0 (0 - 0)	
	5.0-14.9"	10 (0 - 48)	7	26 (0 - 120)	13	17 (0 - 70)	12
	15.0-19.9"	0 (0 - 0)		0 (0 - 3)		3 (0 - 24)	
	20.0+"	1 (0 - 6)		1 (0 - 4)		1 (0 - 6)	

Table 26. Moderately Warm and Dry ecological site in the M332B-East ecoregion listing the average, minimum, and maximum values for ecosystem structural characteristics by disturbance state. #Plots represents the number of FIA plots used to summarize this information. NA = data not available, BA WTD DIA= Basal area weighted diameter and CWD = coarse woody debris.

SIZE-CLASS	ECOSYSTEM CHARACTERISTIC	OPEN (<40% Canopy Cover)		MODERATE (40-60% Canopy Cover)		CLOSED (>60% Canopy Cover)	
GRASS-FORB-SHRUB-SEEDLING				DS1			
				AVG (MIN-MAX)	#PLOTS		
	BA WTD DIA			11 (0 - 40)	8		
	CWD (TONS/AC)			13 (3 - 29)	3		
	FORBS (CC%)			7 (0 - 15)	5		
	GRASS (CC%)			21 (0 - 65)	5		
	SHRUBS (CC%)			21 (3 - 36)	5		
SAPLING/ SMALL		DS2				DS3	
		AVG (MIN-MAX)	#PLOTS			AVG (MIN-MAX)	#PLOTS
	BA WTD DIA	2 (1 - 4)	7			NA	0
	CWD (TONS/AC)	6 (6 - 6)	1			NA	0
	FORBS (CC%)	7 (2 - 11)	6			NA	0
	GRASS (CC%)	10 (5 - 17)	6			NA	0
	SHRUBS (CC%)	7 (2 - 18)	6			NA	0
MEDIUM		DS4		DS5		DS6	
		AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS	AVG (MIN-MAX)	#PLOTS
	BA WTD DIA	9 (0 - 17)	56	10 (6 - 14)	26	8 (4 - 13)	28
	CWD (TONS/AC)	16 (2 - 45)	14	24 (7 - 46)	9	26 (16 - 49)	5
	FORBS (CC%)	6 (0 - 29)	40	7 (1 - 29)	13	3 (1 - 10)	14
	GRASS (CC%)	15 (0 - 60)	40	12 (1 - 47)	13	9 (0 - 26)	14
	SHRUBS (CC%)	13 (0 - 72)	40	16 (0 - 53)	13	10 (1 - 29)	14
LARGE		DS7		DS8		DS9	
	BA WTD DIA	14 (10 - 18)	16	13 (9 - 17)	24	11 (7 - 15)	26
	CWD (TONS/AC)	15 (2 - 32)	6	21 (10 - 32)	5	19 (5 - 33)	6
	FORBS (CC%)	9 (0 - 30)	12	5 (1 - 9)	16	3 (0 - 15)	13
	GRASS (CC%)	14 (0 - 41)	12	16 (0 - 40)	16	8 (1 - 33)	13
	SHRUBS (CC%)	9 (1 - 62)	12	16 (1 - 65)	16	5 (1 - 18)	13
VERY LARGE		DS10		DS11		DS12	
	BA WTD DIA	17 (13 - 19)	7	17 (14 - 20)	13	15 (12 - 19)	11
	CWD (TONS/AC)	8 (3 - 18)	3	19 (4 - 38)	3	21 (21 - 21)	1
	FORBS (CC%)	5 (0 - 8)	4	10 (5 - 21)	3	3 (0 - 13)	6
	GRASS (CC%)	19 (1 - 35)	4	16 (0 - 40)	3	13 (1 - 28)	6
	SHRUBS (CC%)	9 (7 - 12)	4	6 (4 - 8)	3	4 (1 - 13)	6

Historical Range of Variability

A term often used in relation to reference conditions is the historical range of variability (HRV). Historical range of variability is an important concept because it emphasizes that many ecosystems varied in amounts, compositions, and structures due to variations in climate and weather events that influenced historical disturbance regimes (Aplet and Keeton 1999, Haufler et al. 1999). The term natural range of variability (NRV) is sometimes used synonymously with HRV but Egan and Howell (2001) caution against this term for restoration purposes as 1) it doesn't recognize Native American influences on ecosystems, and 2) the term "natural", by itself, is considered too ambiguous to be used to refer to historical reference conditions. The Forest Service uses the term NRV but stresses that this defines the range of conditions prior to Euro-American settlement and includes the influences of Native Americans on historical fire regimes (USFS Ecosystem Restoration Policy).

While native ecosystems were not static during any defined reference period, for ecosystem diversity assessment purposes relative to biodiversity objectives, describing and quantifying HRV is usually confined to a period less than 1000 years prior to Euro-American settlement, as these reflect the habitat conditions most relevant to the plant and animal species present today (Haufler et al. 2002, Swetnam et al. 1999, Morgan et al. 1994). Yet even within a 1000 year timeframe, plant species distributions were typically changing, disturbance regimes were changing, and animal species themselves were adjusting to these changes through behavioral and genetic adaptations.

Describing and quantifying HRV at the landscape level requires information on temporal changes in disturbance states in a spatially explicit format. Various sources of information such as notes from early explorers, fur trappers, and settler's accounts, historical photographs and paintings, natural resource expeditions, and pre-settlement land survey records have been used to describe the native vegetation of the United States before settlement impacts occurred (Egan and Howell 2001). However, this information typically only captures one point in time and is frequently non-spatial. More recently, fire scar analysis and tree mapping studies have been used to describe historical fire frequency as well as forest structures and species compositions. Fortunately, these studies are usually linked to an ecological site which then provides the mechanism to understand differences in disturbance ecology and its effect on different ecosystems within an ecoregion.

Methods

Developing HRV for a particular ecoregion requires a spatial quantification of historical conditions over a specified time-frame. One of the most common and effective methods used to accomplish this is computer simulation. Both non-spatial and spatial models have been developed for this purpose (Keane et al. 2004). While simulation models are recognized to have limitations, they can produce reasonable estimates of HRV (Keane et al. 2009) particularly when other sources of historical information such as historical observation and empirical studies are used to help evaluate and calibrate model results.

Historical range of variability was modeled for terrestrial ecosystems of the BSLRP project area using the spatially explicit landscape model SIMPPLLE (SIMulating Patterns and Processes at Landscape scales)(Chew et al. 2004). SIMPPLLE was used to simulate ecosystem dynamics as a result of primary historical disturbance events (e.g., fire, insects, and disease), climate, and spatially explicit landscape elements (e.g., ecological site, fire breaks, proximity to water, and elevation). SIMPPLLE uses process probabilities with stochastic components, and disturbance response parameters as specified by the user to assign disturbance patterns. The probability of a disturbance process originating or spreading from a specific unit on the landscape is determined not just by the ecosystem attributes, but also by what exists around it, what processes are occurring around it, and what processes have occurred in the past. Although SIMPPLLE has a variety of potential applications, it was specifically used in this project to derive the historical range of variability (HRV) in amounts for each terrestrial forest ecosystem relative to the ecosystem diversity framework used in this assessment.

The "Westside Region 1" module of SIMPPLLE developed by Region 1 of the U.S. Forest Service was used in this assessment for ecoregions M333C and M332B-West and the "Eastside Region 1" module was used for ecoregion M332B-East. System knowledge files are the primary input files and were developed for a specific region and to meet project objectives. This file contained the user developed "logic" that is based on the best available empirical and expert derived knowledge of drivers of ecological conditions.

SIMPPLLE was used to simulate the interaction of historical disturbance regimes and vegetation dynamics over a 1000 year period prior to Euro-American settlement of the BSLRP project area. Appendix F provides a more detailed description of the methods, assumptions, and input sources used in this model. Initial model outputs were compared to fire-scar studies conducted in or near the project area (Tables 7 and 8) to help calibrate the input information and verify that the model results were consistent with empirical information on mean fire return intervals and forest structure and species composition.

Results

Table 27 presents the SIMPLLE model results for mean fire return interval by ecological site and ecoregion. The results were further evaluated against non-spatial, point-in-time data such as historical photographs and early timber surveys, as discussed previously, to determine if these conditions are within the modeled range.

HRV was characterized using the average amount of acres, expressed as percent, that each terrestrial forest ecosystem occupied in three separate runs of SIMPPLLE covering 100 decade simulations (1000 years) for each of the 3 ecoregions in the BSLRP area. Specifically, the mean value was calculated by summing the number of acres representing each disturbance state in a single simulation (decadal output) and dividing by the total number of decades (100 in this case), and then further dividing by the total number of acres in that ecological site. A final mean value was calculated using the results of each of the 3 simulations. Similarly, a 95% confidence interval was calculated for each mean and then a final mean confidence interval was calculated by combining the results of the 3 simulations. Results of the SIMPPLLE model simulations of the upland forest ecosystems historical range of variability are summarized by each of the 3 ecoregions (Figure 16-18).

Table 27. The mean fire interval calculated from SIMPPLLE model results for light severity fire (LSF), moderate severity fire (MSF), and high severity fire (HSF) fire event by ecological site and ecoregion. The mean fire interval is also provided for all 3 fire severity types combined (ALL).

Ecoregion/ Fire Severity	HOT DRY	WARM DRY	MOD WARM DRY	MOD WARM MOD DRY	MOD WARM MOIST	MOD COOL MOIST	COOL MOIST	COOL MOD DRY	COLD	TIM
Ecoregion M333C										
LSF*	-	-	19.2	23.6	24.9	27.2	23.1	21.4	40.5	46.1
MSF	-	-	56.2	40.9	44.9	43.2	71.2	82.7	78.6	109.9
HSF	-	-	6692.4	2229.2	1914.6	2156.5	239.6	299.1	1209.6	1299.0
ALL	-	-	23.0	14.3	14.9	15.9	16.6	16.2	16.1	26.1
Ecoregion M332B-West										
LSF	-	22.4	18.3	-	-	-	22.5	23.2	29.3	40.5
MSF	-	75.9	65.0	-	-	-	65.0	70.0	74.0	110.0
HSF	-	30125.0	7888.0	-	-	-	266.0	228.0	820.0	694.0
ALL	-	17.3	14.2	-	-	-	15.8	16.2	20.5	28.4
Ecoregion M332B-East										
LSF	28.2	17.5	18.9	-	-	-	19.9	21.0	25.0	25.0
MSF	38.5	45.8	41.3	-	-	-	60.0	64.0	61.0	60.0
HSF	379000.0	249222.0	26491.0	-	-	-	341.0	276.0	605.0	628.0
ALL	16.3	12.7	13.0	-	-	-	14.3	15.0	17.1	17.0

* LSF-light severity fire, MSF-moderate severity fire, HSF-high severity fire

TREE SIZECLASS	MOD WARM-DRY			MOD WARM-MOD DRY			MOD WARM-MOIST			MOD COOL-MOIST			COOL-MOIST			COOL-MOD DRY			COLD			TIMBERLINE		
	Canopy Cover[a]			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING (<1.0" DBH)	DS1 0.3 (0-0.7)			DS 1 2.1 (1.1-3.1)			DS 1 2.7 (1.5-3.8)			DS 1 1.9 (1.1-2.7)			DS 1 16.7 (13.8-19.7)			DS 1 13.4 (11.1-15.7)			DS1 3.9 (3.3-4.5)			DS1 4.6 (3.8-5.5)		
SAPLING-SMALL (1.0"-4.9" DBH)	DS2 0.5 (0.1-0.9)		DS3 0.0 (0-0)	DS2 1.5 (0.6-2.3)		DS3 0.1 (0-0.1)	DS2 1.8 (0.8-2.7)		DS3 0.1 (0-0.3)	DS2 1.2 (0.6-19)		DS3 0.1 (0-0.3)	DS2 11.3 (8.9-13.8)		DS3 0.9 (0.4-14)	DS2 9.0 (7.1-10.9)		DS3 0.6 (0.3-1)	DS2 3.1 (2.5-3.7)		DS3 0.0 (0-0)	DS2 3.8 (2.9-4.8)		DS3 0.0 (0-0)
MEDIUM (5.0"-14.9" DBH)	DS4 0.5 (0.1-0.9)	DS5 0.1 (0-0.2)	DS6 0.0 (0-0)	DS4 0.9 (0.4-14)	DS5 0.9 (0.4-15)	DS6 0.3 (0.1-0.5)	DS4 1.0 (0.4-16)	DS5 1.2 (0.6-18)	DS6 0.5 (0.3-0.7)	DS4 0.7 (0.3-11)	DS5 0.8 (0.4-12)	DS6 0.4 (0.2-0.6)	DS4 5.4 (4.1-6.8)	DS5 6.6 (5.1-8.1)	DS6 3.5 (2.3-4.6)	DS4 4.6 (3.5-5.7)	DS5 5.1 (3.9-6.3)	DS6 2.6 (17-10)	DS4 3.8 (3.1-4.5)	DS5 2.4 (19-2.8)	DS6 0.5 (0.4-0.7)	DS4 5.2 (4.1-6.3)	DS5 2.7 (2-3.4)	DS6 0.3 (0.2-0.5)
LARGE (15.0"-19.9" DBH)	DS7 0.6 (0.2-1)	DS8 0.1 (0.1-0.2)	DS9 0.0 (0-0)	DS7 1.3 (0.5-2.1)	DS8 0.6 (0.4-0.9)	DS9 0.2 (0.1-0.3)	DS7 1.4 (0.6-2.2)	DS8 0.9 (0.6-13)	DS9 0.4 (0.2-0.6)	DS7 1.2 (0.6-17)	DS8 0.9 (0.7-12)	DS9 0.6 (0.4-0.8)	DS7 4.9 (3.3-6.6)	DS8 4.9 (4-5.9)	DS9 3.4 (2.3-4.5)	DS7 8.4 (6.8-10)	DS8 7.1 (6.2-8.1)	DS9 4.0 (3.1-5)	DS7 31.0 (27-35)	DS8 27.0 (25-30)	DS9 11.8 (9.5-14)	DS7 29.5 (26-33)	DS8 23.1 (21-25)	DS9 9.8 (8.2-11.5)
VERY LARGE (≥20.0" DBH)	DS10 70.4 (65-76)	DS11 24.7 (20-29)	DS12 2.6 (1-4)	DS10 38.5 (33-44)	DS11 33.5 (29-38)	DS12 20.1 (15-25)	DS10 35.0 (29-41)	DS11 32.4 (28-37)	DS12 22.6 (17-28)	DS10 32.1 (26-38)	DS11 33.9 (30-38)	DS12 26.2 (20-32)	DS10 14.9 (12-18)	DS11 14.3 (12-17)	DS12 13.1 (10-16)	DS10 22.2 (19-25)	DS11 15.0 (13-17)	DS12 8.0 (6-10)	DS10 10.0 (8-12)	DS11 4.1 (3-6)	DS12 2.4 (1-3)	DS10 11.7 (10-14)	DS11 5.1 (4-7)	DS12 4.0 (3-5)
ACRES	23,609			3,404			37,270			39,153			146,600			45,242			30,277			3,275		

[a] Open = 10-39%, Moderate = 40-60%, Closed = >60%

Figure 16. M333C Ecoregion - historical range of variability for ecosystem diversity. Numbers represent the mean percentage of each disturbance state by ecological site. Numbers in parenthesis represent the 95% confidence interval (alpha=0.05, n=100) around the mean.

TREE SIZECLASS	WARM-DRY			MOD WARM-DRY			COOL-MOIST			COOL-MOD DRY			COLD			TIMBERLINE		
	Canopy Cover[a]			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING (<1.0" DBH)	DS1			DS1			DS1			DS1			DS1			DS1		
	0.2 (0.1-0.3)			0.2 (0-0.3)			15.3 (12.7-17.9)			17.6 (14.8-20.3)			5.1 (4.3-5.9)			8.1 (6.8-9.4)		
SAPLING-SMALL (1.0"-4.9" DBH)	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3
	0.2 (0.1-0.3)		0.0 (0-0)	0.3 (0.1-0.4)		0.0 (0-0)	10.6 (8.5-12.8)		0.7 (0.3-1.1)	12.2 (9.9-14.5)		0.9 (0.4-1.3)	4.0 (3.2-4.8)		0.0 (0-0)	6.1 (4.9-7.3)		0.0 (0-0)
MEDIUM (5.0"-14.9" DBH)	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6
	0.3 (0.2-0.4)	0.1 (0.1-0.2)	0.0 (0-0)	0.3 (0.2-0.5)	0.1 (0.1-0.2)	0.0 (0-0)	5.5 (4.2-6.8)	6.1 (4.6-7.6)	2.5 (1.7-3.4)	6.0 (4.7-7.4)	6.8 (0-8.4)	3.3 (2.3-4.3)	5.0 (4.1-6)	3.1 (2.4-3.7)	0.7 (0.5-0.9)	6.5 (5.4-7.6)	5.1 (4.2-6)	1.7 (1.2-2.1)
LARGE (15.0"-19.9" DBH)	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9
	0.3 (0.2-0)	0.1 (0-0.1)	0.0 (0-0)	0.4 (0.2-0.6)	0.1 (0.1-0.2)	0.0 (0-0)	4.9 (3.3-6.4)	4.5 (3.7-5.3)	3.1 (2-4.2)	5.7 (4.2-7.2)	5.7 (4.8-6.6)	3.0 (2-3.9)	31.7 (28-36)	23.9 (21-26)	7.9 (6.2-9.6)	26.8 (23-30)	22.2 (20-24)	8.6 (7-10)
VERY LARGE (≥20.0" DBH)	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12
	77.0 (73-81)	20.3 (17-24)	1.5 (1-2)	76.5 (72-80)	19.6 (16-23)	2.4 (2-3)	18.3 (15-22)	15.8 (13-18)	12.6 (9-16)	13.9 (11-17)	12.7 (11-15)	12.3 (9-15)	12.1 (10-15)	4.9 (3-7)	1.6 (1-2)	9.2 (7-11)	4.0 (3-5)	1.7 (1-2)
ACRES	6,047			172,079			151,228			144,974			36,736			8,467		

[a] Open = 10-39%, Moderate = 40-60%, Closed = >60%

Figure 17. M332B-WEST Ecoregion - historical range of variability for upland forest ecosystem diversity. Numbers represent the mean percentage of each disturbance state by ecological site. Numbers in parenthesis represent the 95% confidence interval (alpha=0.05, n=100) around the mean.

TREE SIZECLASS	HOT-DRY			WARM-DRY			MOD WARM-DRY			COOL-MOIST			COOL-MOD DRY			COLD			TIMBERLINE		
	Canopy Cover[a]			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING (<1.0" DBH)	DS1			DS1			DS1			DS1			DS1			DS1			DS1		
	0.1 (0-0.2)			0.0 (0-0.1)			0.2 (0-0.3)			13.6 (10.7-16.5)			19.7 (16.4-23)			6.6 (3.4-5.8)			6.8 (5.4-8.1)		
SAPLING-SMALL (1.0"-4.9" DBH)	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3
	0.0 (0-0)		0.0 (0-0)	0.0 (0-0)		0.0 (0-0)	0.3 (0.1-0.4)		0.0 (0-0)	9.2 (6.8-11.7)		0.4 (0.1-0.6)	10.5 (7.9-13.2)		0.8 (0.3-14)	4.6 (3.4-5.8)		0.0 (0-0)	4.8 (3.6-6)		0.0 (0-0)
MEDIUM (5.0"-14.9" DBH)	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6
	0.0 (0-0)	0.0 (0-0)	0.0 (0-0)	0.0 (0.02-0.07)	0.0 (0-0.01)	0.0 (0-0)	0.4 (0.2-0.5)	0.1 (0-0.1)	0.0 (0-0)	5.9 (4.2-7.6)	4.0 (2.6-5.5)	1.9 (1.1-2.7)	6.1 (4.5-7.7)	4.9 (3.6-6.2)	2.8 (18-3.8)	5.9 (4.7-7)	3.3 (2.4-4.1)	0.7 (0.4-1)	6.1 (5-7.2)	3.3 (2.4-4.1)	0.6 (0.3-0.9)
LARGE (15.0"-19.9" DBH)	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9
	62.6 (56-70)	36.8 (30-44)	0.3 (0.2-0.5)	0.0 (0-0.1)	0.0 (0-0)	0.0 (0-0)	0.3 (0.1-0.5)	0.0 (0-0.1)	0.0 (0-0)	5.2 (3.5-7)	2.9 (2.1-3.6)	2.1 (1.1-3)	6.5 (4.7-8.3)	3.6 (2.7-4.5)	1.9 (1.1-2.6)	38.4 (33.9-42.9)	16.8 (13.9-19.7)	4.0 (2.7-5.3)	38.0 (34-42)	16.0 (13-19)	4.0 (2.9-5.1)
VERY LARGE (≥20.0" DBH)	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12
	0.0 (0-0)	0.0 (0-0)	0.1 (0-0)	91.8 (90-94)	7.8 (6-10)	0.3 (0.1-0.4)	83.4 (79-87)	13.7 (10-17)	1.7 (1-2)	30.7 (26-35)	14.7 (12-18)	9.3 (6-13)	20.8 (17-24)	12.6 (10-15)	9.8 (7-13)	13.9 (11-17)	4.3 (3-6)	1.6 (1-2)	15.1 (12-18)	4.1 (2-6)	1.3 (1-2)
ACRES	668			4,919			80,110			75,932			98,562			21,025			4,157		

[a] Open = 10-39%, Moderate = 40-60%, Closed = >60%

Figure 18. M332B-EAST Ecoregion - historical range of variability for ecosystem diversity. Numbers represent the mean percentage of each disturbance state by ecological site. Numbers in parenthesis represent the 95% confidence interval (alpha=0.05, n=100) around the mean.

Disturbance Severity and Regimes

Methods

Because much of the confusion and controversy concerning disturbance regimes, particularly fire regimes and fire severity, stem from differing use and application of terminology, fire severity and fire regime was defined for its use in this assessment. Low severity fire was defined as resulting in <25% overstory tree mortality, moderate severity fire as resulting in 25-75% overstory tree mortality, and high severity fire as resulting in >75% overstory tree mortality (Figure 19). The ability to map and characterize disturbance regimes is influenced by the types and scales of mapping and the disturbance pattern classification used. Andison (2012) demonstrated how different methods of mapping fire boundaries can influence interpretations of fire regimes. In this landscape assessment, relative to forest structural patterns, we differentiated 4 fire regimes according to the criteria shown in Figure 20 (an estimated cross-walk of these fire regimes and patterns with LANDFIRE fire regime groups, is provided in Appendix D, Table D-2). The non-lethal fire regime exhibited <10% high severity fire within an analysis window of defined size with >90% of the fire severity being low to moderate severity. The mixed-severity A disturbance regime exhibited 10% high severity fire but less than 50% high severity fire, with >50-90% of the fire severity being low to moderate. The mixed-severity B disturbance regime was characterized as having >50 but <90% high severity fire, with >10 but <50% low to moderate severity fire. The lethal disturbance regime was characterized as having >90% high severity fire within the analysis window.

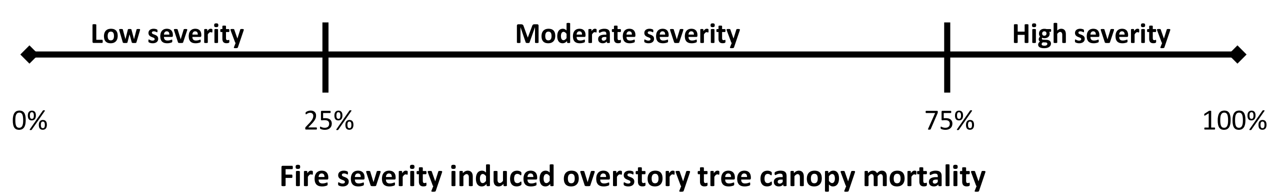


Figure 19. Fire severity classes identified for the landscape assessment based on the induced overstory tree canopy mortality.

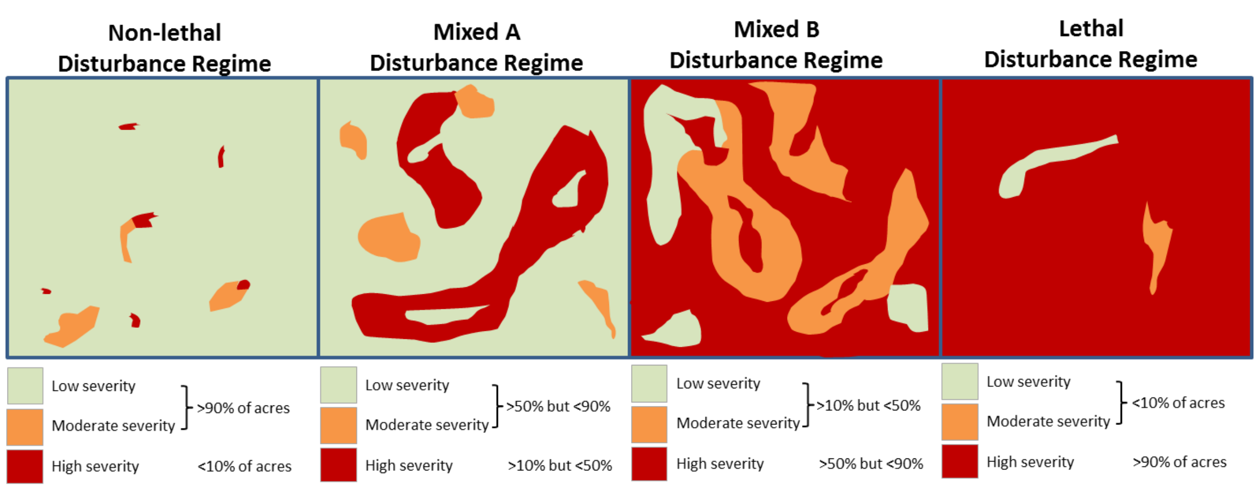


Figure 20. Example disturbance severity patterns that characterize the four disturbance fire regime classes of the landscape assessment area. These patterns are applied at the scale of roughly 90 acres for this assessment.

Failure to recognize disturbance regimes have a spatial scale associated with fire severity patterns has led to considerable confusion when attempting to compare the results of multiple studies. Since the scale of the depicted area will influence the interpretation of disturbance severity patterns, it is important to clearly identify the scale of application (Kaufmann et al. 2007). Most upland forests in the landscape assessment area could experience any combination of low, moderate, or high severity disturbance effects given different climate cycles, but describing and quantifying an historical disturbance regime requires a clear classification of disturbance patterns and a defined scale of application to effectively communicate and compare the results to other studies. For the purposes of this assessment, we have identified a scale of 90 acres for quantifying disturbance regime patterns using the classification in Figure 20. We conducted an evaluation of multiple scales (from 24 to 560 acres), as described in Appendix G, and concluded the 90 acre scale was the most effective for representing known disturbance regime patterns, using ecological sites within the assessment area.

Fire severities were evaluated based on the ecosystem diversity output of SIMPPLLE. SIMPPLLE modeled fire occurrences as well as other disturbances and successional processes for each decadal time step over the 1000 year simulation. The specific disturbance (such as fire severity) applied to a pixel was based on the climate conditions for that time step, a stochastic generation of fire start locations, and what was occurring in adjacent pixels. The climate assigned to each decade was based on the Pacific Decadal Oscillation climate data developed by Region 1 of the U.S. Forest Service (Eric Henderson, personal communication). Fires were mapped for each decade, with each 2.5 ha pixel in the modeled landscape experiencing a fire being assigned a low, moderate, or high severity status. Insect and disease disturbances were also assigned to pixels. These disturbances produced changes to the ecosystem assigned to a pixel based on the change/transition logic contained in the successional/disturbance pathways programmed for each ecosystem. The specific ecosystem assigned to a pixel could stay the same, move to a higher canopy cover or size class based on successional change, or move to another disturbance state in response to a disturbance, as shown in Figure 8. At the end of each time step, a new map of the ecosytems assigned to each pixel in the ecoregion was generated.

Fire regimes were evaluated in a geographic information system (GIS) using a 90 acre moving window and using the fire regime logic displayed in Figure 20. A description of these methods are provided in Appendix G. Fire regimes were quantified based on the ecosystem resulting from disturbance severities and averaged for 5 decadal points in time over the modeled 1000 years, rather than on the simulated fires. This was necessary due to the constraints of applying SIMPPLLE to decadal time steps, as discussed below, as fire patterns resulting from 10 year time steps would likely differ from patterns produced by running annual fire events. However, the resulting landscape ecosystem patterns should adequately reflect historical conditions, as these are based on achieving overall return rates of fire consistent with empirical and observational data of historical fire regimes.

SIMPPLLE is a powerful landscape dynamics model that can be used to help explain states and transitions among ecosystems in specific settings considering spatial arrangements, terrain, and other features within a landscape of interest. However, as with all models, a number of limitations exist which should be considered when interpreting or using SIMPPLLE's ecosystem diversity results.

SIMPPLLE forest simulations for this landscape assessment were based on 10 year (decadal) time steps. Thus, an analysis of 1000 years uses 100 steps, thereby simplifying the 1000 possible annual variations that would have occurred. The successional/disturbance pathways included in these analyses therefore attempt to capture the changes in forest compositions and structures that occur over 10 years, and assume a uniform condition for that 10 year block of time. These results are reasonable for more gradual processes such as succession, however, other disturbances such as insects and diseases, may take several years to develop and affect a specific stand. Disturbances such as fire, on the other hand, produce immediate effects. In attempting to identify a range of variation, results based on decadal analyses may be greater than estimates of variation produced from running annual increments. Thus, the ranges of variation produced from decadal outputs of SIMPPLLE must be viewed with some caution as they may represent a 10 factor greater variation than what might be produced using an annual analysis. For example, in determining fire amounts and types, the outputs reflect fire occurring over a 10 year timeframe. When interpreting these results, fire return can only be viewed as occurring once during this 10 year timeframe. This creates a minimum fire return interval of 10 years, which we know from fire history studies may be longer than occurred on some drier sties. The cumulative results for that decade must be displayed as the total amount of fire occurring during the 10 year time step, when in reality there would likely be varied fire patterns on an annual basis over that 10 year period. While a general description of the landscape dynamics can be developed using decadal analyses and the estimated amounts of disturbance states over time should be reflective of the overall effects of fire and other disturbances in the landscape, the potential effects of the decadal time step on the mapped sizes of individual fires should be considered and noted.

A factor related to the decadal limitations is how SIMPPLLE handles climate patterns. Climate effects in SIMPPLLE can be varied to simulate 3 combinations of decadal moisture and temperature patterns - normal and normal, wetter and cooler, or drier and warmer. The USFS Northern Region developed an historical climate data set based on estimated Pacific Decadal Oscillations (PDO's) derived from empirical data, which is the best scientific basis for this modeling. However, the combination of using decadal analyses and only identifying three climate classes, may limit the possible complexity of outcomes.

An additional limitation is the extent that stochasticity is applied to fire as a disturbance. SIMPPLLE incorporates stochastic properties into outputs relating to fire in several ways. The user-supplied number of fire starts are stochastically applied to the landscape. This allows for different runs to generate different results based on landscape characteristics that may determine how fire occurs in response to different fire starting locations. However, once fires start, their occurrence in each pixel is set by the designated climate pattern for that decade as either light severity, moderate severity, or high severity. An additional stochasticity is application of an "extreme wind event", which may cause a fire to change its designated intensity when this random event occurs during a model run. Otherwise, the type of fire is set according to the vegetation-response pathway designated for that particular ecological site, disturbance state, and assigned decadal climate patterns. In reality, fire severities are much more dependent on site conditions than currently allowed in the model. Haivng more flexibility to assign fire severity risk by ecological site would provide more opportunity for realistic stochasticity based on empirical data. While there is certainly considerable validity to the fire outputs from SIMPPLLE at the present, the model does reduce the potential variability that could occur in fire types at finer scales due to the primarily deterministic outputs produced for each decadal time period. This limitation is less of a factor in considering an overall landscape analysis over time, but it can have an influence on the spatial patterns produced in any given time step.

The scale of mapping is also a consideration in analyzing SIMPPLLE's outputs. The minimum scale used in this assessment was 2.5 ha (6 acre) pixels, which is a reasonable size for mapping forest stands at landscape scales. However, fire and other disturbances frequently operate at scales finer than 6 acres. For example, fire may burn small patches of vegetation at higher or lower intensities than 6 acre blocks, but SIMPPLLE is constrained to consider one type of fire severity for a 6 acre pixel. These potential differences should not cause significant differences in outputs when considered at landscape scales, however when looking at information such as average polygon sizes resulting from disturbance events, the results will be influenced by the minimum pixel size. This is especially important when evaluating and comparing spatial patterns generated by different mapping criteria.

Results

Examples of decadal output maps from SIMPPLLE displaying results across a range of climatic conditions are shown in Figures 21-23. Figure 24 provides a summary of the fire regime results of the SIMPPLLE model simulations for the BSLRP project area with forest ecosystem conditions characterized by the non-lethal, mixed-severity A, mixed-severity B, and lethal fire regimes.

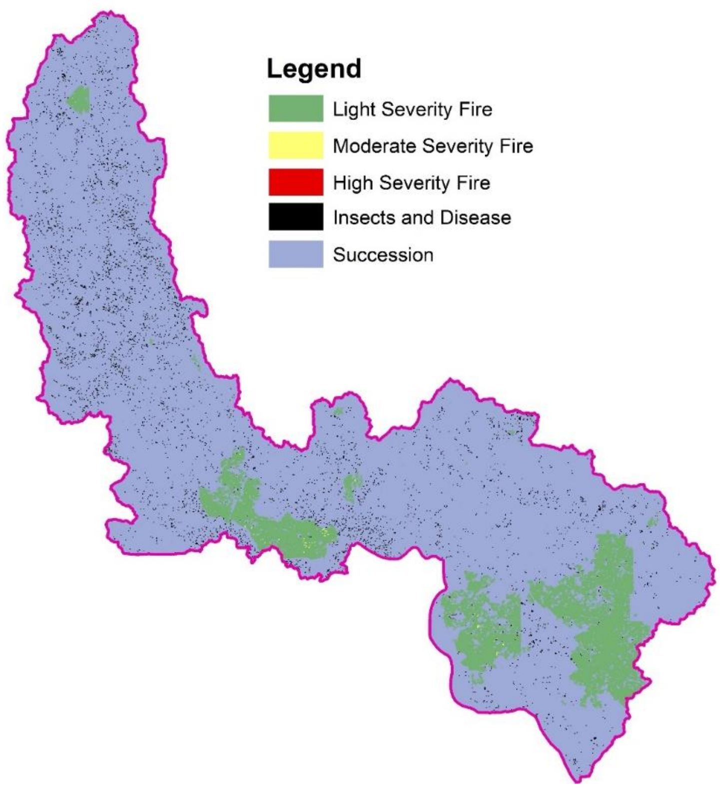


Figure 21. Disturbance processes mapped in SIMPPLLE for a decadal time step occurring during a cool and moist climatic period during the 1000 year simulation for the BSLRP project area.

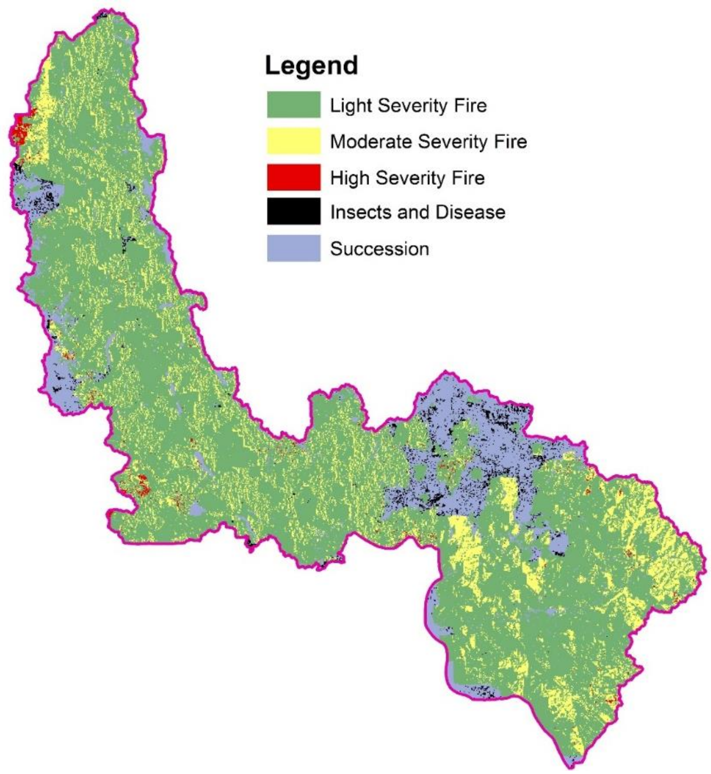


Figure 22. Disturbances processes mapped in SIMPPLLE for a decadal time step occurring during a normal or average climatic period during the 1000 year simulation for the BSLRP project area.

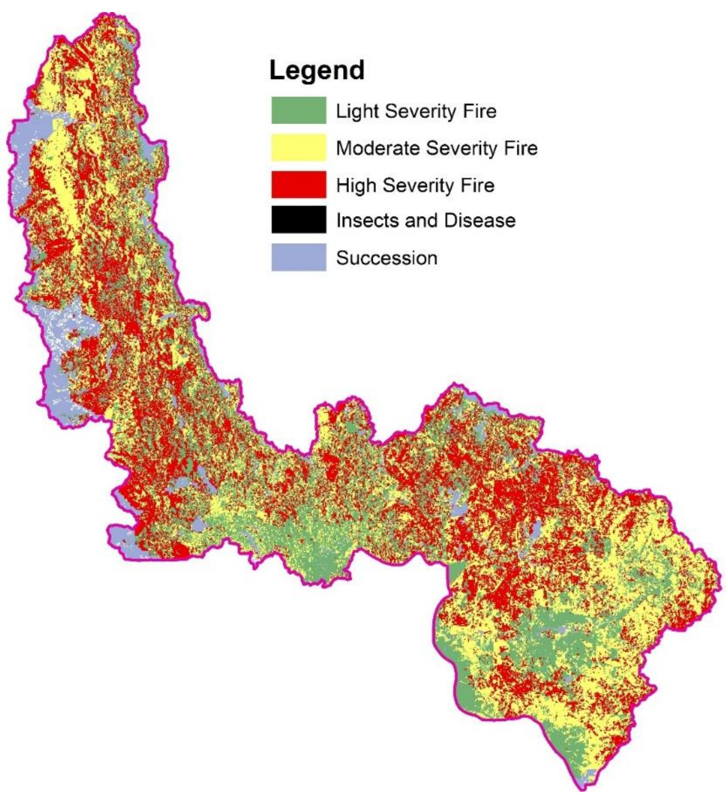


Figure 23. Disturbance processes mapped in SIMPPLLE for a decadal time step occurring during a warm and dry climatic period during the 1000 year simulation for the BSLRP project area.

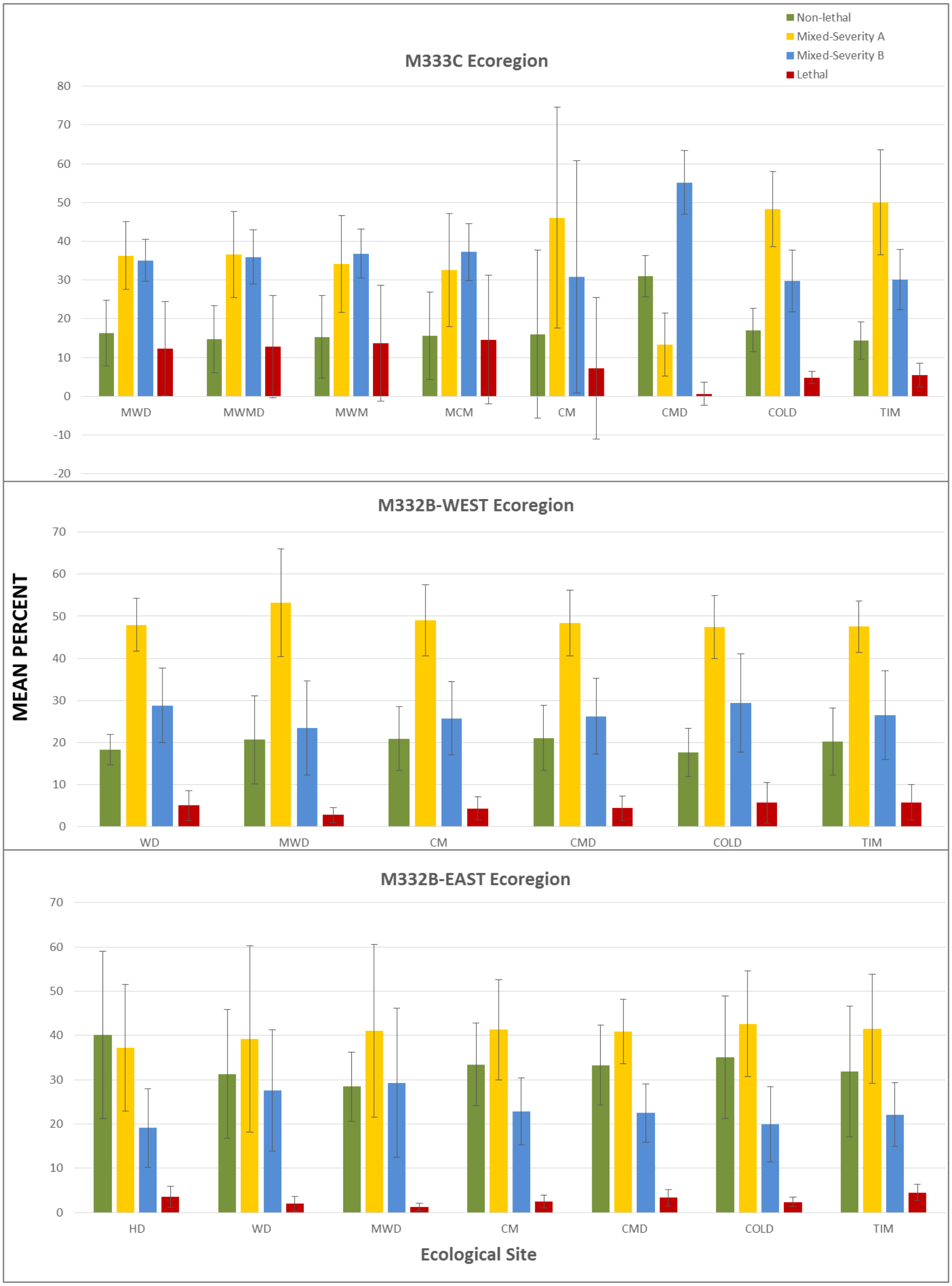


Figure 24. Mean percentage of fire regimes calculated using SIMPPLLE for each ecological site in each ecoregion of the BSLRP project area. Bars represent the 95% confidence interval around the mean value.

Historical Condition Discussion

The SIMPPLLE model outputs show a range of disturbances across decadal time steps over the 1000 years of analysis. These results show the variability and significant influence of fires when models are calibrated to be consistent with fire return intervals reported by empirical and observational information to have occurred historically in the project area. The SIMPPLLE outputs are highly sensitive to the linkage of fire regimes to the designated climate conditions assigned to each decadal time step. As figures 21-23 reveal, considerable variation was programmed into SIMPPLLE in response to these climate variations. During cool and wet periods fire was minimal (Figure 21), while during warm and dry periods (Figure 23) fire occurred in greater amounts and with greater severity.

Insects and disease had some, but relatively minimal influence, when compared to fire in the historical landscape. When they did occur, it was primarily during cool and moist decades, particularly when several cool and moist decades occurred back to back. During these times, small pockets of insects and diseases were noted. However, when more normal or warmer and drier decades occurred, fire again dominated the disturbance processes. With higher amounts of fire, forests were either pushed back successionally to younger age classes more resistant to insect and disease or were made more open by the effects of low and moderate severity fires. Open stand conditions have been documented to be more resistant to the effects of insects and disease, as discussed previously. Thus, while insects and disease did occur historically across the project area, the extent of their influence was much less when compared to the influence of fire as a disturbance process.

One interesting result was the minimal differences occurring in fire regimes across ecological sites within each of the 3 ecoregions (Figure 24). The lack of differences across ecological sites within an ecoregion was surprising. Past research and historical accounts have shown trends for more non-lethal and mixed severity A fire regimes in warmer and drier ecological sites with higher amounts of mixed severity B and lethal fire regimes in cooler and moister ecological sites. Our results noted some minor trends for this but they were less than expected based on empirical information. Several explanations are possible for this result. First, while SIMPPLLE was programmed to apply different fire severities under different climate conditions across ecological sites, the climate designated for a decadal time period had the greatest influence on amounts and types of fire, as noted above. With this largely deterministic effect of designated climate conditions for a time step, the effects of climate may have overridden the effects of differences in ecological sites. Additionally, with fire applied as a decadal process, fire sizes may have been substantially larger than would have occurred if SIMPPLLE were run in annual time steps. Annual time steps may have resulted in different fire size patterns that could have better shown differences in a patchy environment of ecological sites. Thus, the decadal time steps may have resulted in a more uniform distribution of fire regimes across the landscape then might have been produced with annual time steps.

Also of interest was the differences in fire regimes produced across the 3 ecoregions (Figure 24). While some differences were expected, they were believed to be secondary to differences occurring across ecological sites within an ecoregion. Instead, the results show a definite trend of more non-lethal and mixed severity A fire regimes in the M332B-East landscape compared to the M333C landscape, with the M332B-West displaying values in between the other two ecoregions. These results are attributed to the differences in the ecological sites that predominated in the different landscapes, with more drier sites occurring as one moved further east. Thus, while differences in fire regimes were not observed across ecological sites within an ecoregion, the observed differences in fire regimes across ecoregions would appear to be a response to the amounts and distributions of ecological sites, at ecoregional scales.

Fire size was not evaluated due to the decadal time constraints used in the Region 1 SIMPPLLE model, as discussed previously. We did examine the scale used in defining fire regimes. As previously dicussed, a fire regime is classified on the basis of the amount and pattern of different severities of fire occurring in a defined area. We were interested in using a scale that allowed for the determination of differences in fire regimes caused by differences in the abiotic environmental, which in this assessment were classified by different ecological sites. Larger sized areas are likely to include a mix of ecological sites, although these are likely to cluster with elevational differences. We examined scale of analysis from 24 acres up to 90 acres (Appendix G) and found no major differences across this range of analysis sizes. Therefore, we used a 90 acre scale to characterize fire regimes in this assessment. We also evaluated a 560 acre scale. The comparison of this scale is included in Appendix G. It revealed a decrease in differences in fire regimes at this scale across ecological sites, likely due to the inclusion of a greater mix of terrain, ecological sites, and other features resulting in a more uniform classification of fire regimes. Because we were interested in identifying differences in fire regimes, we used the 90 acre scale to allow for the greater variation in fire regimes while covering as large an area to evaluate pattern of fires without losing this variation.

Today's Ecosystem Diversity

Native ecosystems and habitats of the BSLRP area have and continue to be directly and indirectly altered by human actions. Although Native Americans interacted and influenced this landscape for thousands of years, those influences are incorporated in the historical reference. It is the extent of human influence over the last 150 years that is of primary interest when considering the cumulative changes to native ecosystem diversity and biodiversity of the area. More specifically, two primary types of ecosystem conversion or alteration have occurred within the BSLRP area and have contributed to the cumulative changes to native ecosystem diversity observed in the landscape today. These are: 1) the direct conversion of native ecosystems to some other land type or use (Alig 2007), and 2) the indirect alteration of native ecosystems through the suppression of historical disturbance processes or alteration of species compositions, structures, or functions resulting from human activities and spread of non-native species. In the project area, the primary causes of direct conversion of native terrestrial ecosystems included agriculture, roads, residential and urban development (including gravel pits, golf courses, airports, etc.), and rural farm development (i.e., residences/out-building sites/high density animal holding sites). The primary causes of indirect alteration of ecosystems include timber harvests, fire suppression, altered grazing regimes, as well as accidental or intentional introduction of non-native species.

Developing an understanding of the upland forest ecosystem conditions present in the BSLRP area today is an important step toward identifying and quantifying cumulative changes to native ecosystem diversity. Comparing the results of the historical analyses to current conditions allows for these comparisons to be made.

Methods

Current ecosystem diversity was quantified using the map of ecological sites and a map of current tree size classes based on the ecosystem diversity framework, as provided by the BSLRP team. Current stand conditions were developed from the USFS Region 1 VMAP classification and included some adjustments using FIA plot data to better identify large and very large tree conditions (methods described in Appendix H). VMAP is known to be less accurate in mapping tree size classes than it is for other vegetation attributes such as canopy cover. While the BSLRP team tried to adjust for known deficiencies in identifying large and very large tree conditions, the current stand conditions layer was still thought to underestimate the presence larger-sized trees based on anecdotal observations. For this reason, the current ecosystem conditions developed using the VMAP classification were compared to FIA plot data occurring in each of the 3 ecoregions and classified to the ecosystem diversity framework used in the assessment. FIA plot data allowed for a determination of the distribution of tree sizes across each ecoregion. However, there are insufficient numbers of FIA plots to fully populate the ecosystem diversity framework for each ecoregion, especially M333C and M332B-West which contain only 32 and 39 FIA plots for forest ecological sites, respectively. M332B-East has had an “intensified” FIA sampling, resulting in 192 plots for this ecoregion. While the FIA data are limited, they do provide some indication of the distribution of tree sizes for the ecological sites in each ecoregion, and can be used as a coarse check on the VMAP estimates of size classes of trees.

We classified the existing ecosystem diversity in the same manner as we did the historical landscapes, identifying the amounts of each disturbance state occurring within each ecological site. We assessed current fire regimes in the same manner as we did historical fire regimes by applying the fire severity expected to occur on each pixel under a normal climate condition. We then analyzed the resulting fire regime at the 90 acre scale based on the fire severity that was assigned to each pixel within each 90 acre area. Thus, for fire regimes, we did not model actual fires across the landscape, but the potential for fire severity based on the historical and current stand conditions evaluated in the same way.

Changes to the landscape were analyzed in two ways. First, the amounts of today’s ecosystem diversity were compared to those produced by the SIMPPLLE historical modelling. This allowed changes in amounts of different disturbance states to be identified. Second, the amount of direct conversion of forest ecosystems was evaluated. Evidence of converted conditions (i.e., non-ecosystem conditions) were identified from VMAP satellite imagery data for urban and developed categories. A road GIS layer was buffered to represent an 8 meter width and to estimate the expected surface impact. The Montana state cadastral GIS layer was used to identify properties with existing residences and a 4 acre circle centered on the lot was used to estimate the ecological loss from conversion and disturbance. The converted conditions layer was combined with the ecological site GIS layer to quantify today’s converted acres.

Results

Table 28 lists the direct conversions occurring within the project area. Figures 25-27 display the current ecosystem diversity for each of the 3 ecoregions based on VMAP interpretations of disturbance states occurring across underlying ecological sites. Figures 28-30 present the ecosystem diversity of the current landscapes as estimated from FIA plot analyses, where available.

Table 28. The amount (acres) of an ecological site directly converted to non-ecological uses by conversion type and percent overall for each of the 3 ecoregions.

Ecoregion/ Conversion Type	Ecological Site										TOTAL
	HD	WD	MWD	MWMD	MWM	MCM	CM	CMD	COLD	TIM	
ECOREGION M333C											
Town/Residential Development	NA[a]	NA	121	12	100	83	1408	0	413	0	2136
Roads	NA	NA	128	19	311	281	774	8	155	0	1675
Total Acres			**249**	**31**	**410**	**363**	**2182**	**8**	**568**	**0**	**3811**
% Conversion of Ecological Site			**1.05**	**0.91**	**1.10**	**0.93**	**1.49**	**0.02**	**1.87**	**0**	**1.2**
ECOREGION M332B-WEST											
Town/Residential Development	NA	11	1716	NA	NA	NA	1358	0	187	0	3271
Roads	NA	7	939	NA	NA	NA	963	19	630	5	2563
Cropland/Non-native Pasture	NA	2	113	NA	NA	NA	177	0	39	0	332
Rural Farm Development	NA	-	135	NA	NA	NA	143	0	44	0	322
Total Acres		**19**	**2903**				**2641**	**19**	**900**	**5**	**6487**
% Conversion of Ecological Site		**0.31**	**1.69**				**1.75**	**0.01**	**2.45**	**0.06**	**1.25**
ECOREGION M332B-EAST											
Town/Residential Development	0	31	289	NA	NA	NA	336	52	0	0	707
Roads	0	26	161	NA	NA	NA	128	44	1	0	360
Cropland/Non-native Pasture	0	16	106	NA	NA	NA	101	12	0	0	235
Rural Farm Development	0	6	88	NA	NA	NA	92	40	0	0	227
Total Acres	**0**	**79**	**644**				**656**	**149**	**1**	**0**	**1529**
% Conversion of Ecological Site		**1.60**	**0.80**				**0.86**	**0.15**	**0.01**	**0**	**0.54**

[a] NA – not applicable; ecological site does not occur in this ecoregion

TREE SIZE CLASS	Structure	MOD WARM-DRY	MOD WARM-MOD DRY	MOD WARM-MOIST	MOD COOL-MOIST	COOL-MOIST	COOL-MOD DRY	COLD	TIMBERLINE
	Canopy Cover (Open, Moderate, Closed)	Canopy Cover	Canopy Cover	Canopy Cover	Canopy Cover	Canopy Cover	Canopy Cover	Canopy Cover	Canopy Cover
GRASS-FORB-SHRUB-SEEDLING	DS1	13.2	3.7	5.7	5.5	9.4	20.7	13.1	27.9
SAPLING-SMALL TREE	DS2	17.8	21.0	9.9	10.9	13.3	18.9	18.2	34.0
SAPLING-SMALL TREE	DS3	0.6	0.2	2.2	6.1	1.4	0.4	0.1	0.0
MEDIUM TREE	DS4	13.7	6.3	6.7	4.0	7.9	9.3	21.6	18.2
MEDIUM TREE	DS5	11.7	13.3	8.1	5.8	8.6	7.0	13.4	8.5
MEDIUM TREE	DS6	18.4	33.5	23.0	28.1	19.4	15.3	10.9	1.4
LARGE TREE	DS7	6.5	3.1	3.4	1.6	4.2	4.7	4.1	3.6
LARGE TREE	DS8	10.9	8.8	14.3	8.7	11.5	9.8	14.6	3.2
LARGE TREE	DS9	3.9	8.6	21.0	24.7	7.5	5.4	4.0	3.2
VERY LARGE TREE	DS10	1.7	0.0	1.8	0.9	2.9	2.8	0.0	0.0
VERY LARGE TREE	DS11	1.4	1.0	3.1	2.0	8.8	4.2	0.0	0.0
VERY LARGE TREE	DS12	0.4	0.5	0.8	1.6	4.9	1.5	0.0	0.0
ACRES		23,609	3,404	37,270	39,153	146,600	45,242	30,277	3,275

Figure 25. Current ecosystem diversity based on VMAP classification, as provided by the BSLRP Team, for ecoregion M333C.

TREE SIZE CLASS	WARM-DRY			MOD WARM-DRY			COOL-MOIST			COOL-MOD DRY			COLD			TIMBERLINE		
	Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	DS1			DS1			DS1			DS1			DS1			DS1		
	30.1			18.2			16.9			23.9			16.4			19.1		
SAPLING-SMALL TREE	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3	DS2		DS3
	21.3		8.1	4.9		1.6	10.7		3.7	13.0		3.5	12.1		2.4	16.7		1.9
MEDIUM TREE	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6	DS4	DS5	DS6
	6.7	7.8	3.3	7.9	10.3	10.7	3.3	7.5	18.8	3.6	6.4	15.8	11.7	12.5	13.9	25.8	12.8	5.8
LARGE TREE	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9	DS7	DS8	DS9
	3.3	6.2	1.7	1.2	5.3	13.6	3.4	6.7	13.3	3.7	5.4	10.7	5.5	13.6	11.9	5.0	9.3	3.4
VERY LARGE TREE	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12	DS10	DS11	DS12
	3.4	6.0	2.1	5.5	15.2	5.7	2.6	7.2	5.9	2.9	6.3	4.8	0.0	0.0	0.0	0.0	0.0	0.0
ACRES		6,047			172,079			151,228			144,974			36,736			8,467	

Figure 26. Current ecosystem diversity based on VMAP classification, as provided by the BSLRP Team, for ecoregion M332B-West.

TREE SIZE CLASS	Canopy Cover[a]		HOT-DRY	WARM-DRY	MOD WARM-DRY	COOL-MOIST	COOL-MOD DRY	COLD	TIMBERLINE
GRASS-FORB-SHRUB-SEEDLING	Open Moderate Closed	DS1	28.9	1.1	4.8	0.0	7.8	0.0	0.0
SAPLING-SMALL TREE	Open Moderate	DS2	9.8	3.8	14.4	17.8	26.3	9.2	15.7
	Closed	DS3	0.4	0.3	1.1	5.2	5.8	3.4	3.5
MEDIUM TREE	Open	DS4	9.5	18.0	4.6	5.4	4.7	16.6	10.0
	Moderate	DS5	7.8	13.5	25.2	28.1	19.5	23.0	34.6
	Closed	DS6	3.3	6.4	18.6	31.1	27.2	46.1	32.5
LARGE TREE	Open	DS7	12.2	15.8	6.1	1.3	1.3	0.3	0.2
	Moderate	DS8	10.2	18.8	12.9	5.1	3.1	0.3	1.6
	Closed	DS9	0.3	2.0	1.9	2.0	1.3	1.2	1.4
VERY LARGE TREE	Open	DS10	10.8	13.3	6.2	0.6	0.9	0.0	0.2
	Moderate	DS11	6.3	6.9	4.1	2.7	1.6	0.0	0.2
	Closed	DS12	0.3	0.2	0.1	0.9	0.3	0.0	0.0
ACRES			668	4,919	80,110	75,932	98,562	21,025	4,157

Figure 27. Current ecosystem diversity based on VMAP classification, as provided by the BSLRP Team, for ecoregion M332B-East.

TREE SIZE CLASS		MOD WARM-DRY Canopy Cover[a] Open Moderate Closed	MOD WARM-MOD DRY Canopy Cover Open Moderate Closed	MOD WARM-MOIST Canopy Cover Open Moderate Closed	MOD COOL-MOIST Canopy Cover Open Moderate Closed	COOL-MOIST Canopy Cover Open Moderate Closed	COOL-MOD DRY Canopy Cover Open Moderate Closed	COLD Canopy Cover Open Moderate Closed	TIMBERLINE Canopy Cover Open Moderate Closed
GRASS-FORB-SHRUB-SEEDLING	DS1	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	-
SAPLING-SMALL TREE	DS2	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	9.1 (1)	25.0 (1)	-
	DS3	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	-
MEDIUM TREE	DS4	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	7.7 (1)	9.1 (1)	50.0 (2)	-
	DS5	100.0 (1)	100.0 (1)	0.0 (0)	0.0 (0)	23.1 (3)	18.2 (2)	0.0 (0)	-
	DS6	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	9.1 (1)	0.0 (0)	-
LARGE TREE	DS7	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	7.7 (1)	0.0 (0)	25.0 (1)	-
	DS8	0.0 (0)	0.0 (0)	100.0 (1)	0.0 (0)	7.7 (1)	9.1 (1)	0.0 (0)	-
	DS9	0.0 (0)	0.0 (0)	0.0 (0)	100.0 (1)	0.0 (0)	9.1 (1)	0.0 (0)	-
VERY LARGE TREE	DS10	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	30.8 (4)	9.1 (1)	0.0 (0)	-
	DS11	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	23.1 (3)	27.3 (3)	0.0 (0)	-
	DS12	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	-
ACRES		23,609	3,404	37,270	39,153	146,600	45,242	30,277	3,275

Figure 28. Current ecosystem diversity based on FIA plot data, where available in the region, classified using the sequential size method for ecoregion M333C.

TREE SIZE CLASS		WARM-DRY	MOD WARM-DRY	COOL-MOIST	COOL-MOD DRY	COLD	TIMBERLINE
	Canopy Cover	Canopy Cover[a]: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed	Canopy Cover: Open / Moderate / Closed
GRASS-FORB-SHRUB-SEEDLING	DS1	-	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	-
SAPLING-SMALL TREE	DS2	-	0.0 (0)	0.0 (0)	13.3 (2)	0.0 (0)	-
SAPLING-SMALL TREE	DS3	-	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)	-
MEDIUM TREE	DS4	-	33.3 (2)	26.7 (4)	6.7 (1)	37.5 (3)	-
MEDIUM TREE	DS5	-	0.0 (0)	0.0 (0)	33.3 (0)	0.0 (0)	-
MEDIUM TREE	DS6	-	0.0 (0)	0.0 (0)	13.3 (2)	0.0 (0)	-
LARGE TREE	DS7	-	0.0 (0)	0.0 (0)	13.3 (2)	0.0 (0)	-
LARGE TREE	DS8	-	16.7 (1)	20.0 (3)	0.0 (0)	37.5 (3)	-
LARGE TREE	DS9	-	0.0 (0)	13.3 (2)	6.7 (1)	25.0 (2)	-
VERY LARGE TREE	DS10	-	16.7 (1)	13.3 (2)	0.0 (0)	0.0 (0)	-
VERY LARGE TREE	DS11	-	33.3 (2)	26.7 (4)	6.7 (1)	0.0 (0)	-
VERY LARGE TREE	DS12	-	0.0 (0)	0.0 (0)	6.7 (1)	0.0 (0)	-
ACRES		6,047	172,079	151,228	144,974	36,736	8,467

Figure 29. Current ecosystem diversity based on FIA plot data, where available in the region, classified using the sequential size method for ecoregion M332B-West.

TREE SIZE CLASS		HOT-DRY	WARM-DRY	MOD WARM-DRY	COOL-MOIST	COOL-MOD DRY	COLD	TIMBERLINE
		Canopy Cover[a] Open / Moderate / Closed	Canopy Cover Open / Moderate / Closed	Canopy Cover Open / Moderate / Closed	Canopy Cover Open / Moderate / Closed	Canopy Cover Open / Moderate / Closed	Canopy Cover Open / Moderate / Closed	Canopy Cover Open / Moderate / Closed
GRASS-FORB-SHRUB-SEEDLING	DS1	0.0 (0)	0.0 (0)	3.9 (2)	9.1 (4)	16.7 (13)	25.0 (2)	0.0 (0)
SAPLING-SMALL TREE	DS2	0.0 (0)	0.0 (0)	3.9 (2)	4.5 (2)	3.8 (3)	0.0 (0)	33.3 (1)
	DS3	0.0 (0)	0.0 (0)	0.0 (0)	2.3 (1)	5.1 (4)	0.0 (0)	0.0 (0)
MEDIUM TREE	DS4	50.0 (1)	33.3 (2)	19.6 (10)	20.5 (9)	29.5 (23)	37.5 (3)	0.0 (0)
	DS5	0.0 (0)	33.3 (2)	19.6 (10)	18.2 (8)	15.4 (12)	25.0 (3)	33.3 (1)
	DS6	50.0 (1)	16.7 (1)	5.9 (3)	20.5 (9)	15.4 (12)	0.0 (0)	33.3 (1)
LARGE TREE	DS7	0.0 (0)	0.0 (0)	9.8 (5)	0.0 (0)	2.6 (2)	12.5 (1)	0.0 (0)
	DS8	0.0 (0)	0.0 (0)	5.9 (3)	6.8 (2)	2.6 (2)	0.0 (0)	0.0 (0)
	DS9	0.0 (0)	0.0 (0)	11.8 (6)	13.6 (6)	3.8 (3)	0.0 (0)	0.0 (0)
VERY LARGE TREE	DS10	0.0 (0)	0.0	7.8 (4)	0.0 (0)	0.0 (0)	0.0 (0)	0.0 (0)
	DS11	0.0 (0)	16.7 (1)	7.8 (4)	2.3 (1)	5.1 (4)	0.0 (0)	0.0 (0)
	DS12	0.0 (0)	0.0 (0)	3.9 (2)	2.3 (1)	0.0 (0)	0.0 (0)	0.0 (0)
ACRES		668	4,919	80,110	75,932	98,562	21,025	4,157

Figure 30. Current ecosystem diversity based on FIA plot data, where available in the region, classified using the sequential size method for ecoregion M332B-East.

Cumulative Changes in Ecosystem Diversity

Direct Conversion of Native Ecosystems

Direct land conversion estimated for upland forested ecological sites in the BSLRP area was low at 11,827 acres or 1% of the total upland forested acres. While the existing human footprint is not insignificant, direct conversions represent a small component of the overall project area.

Indirect Conversion of Native Ecosystems

The ability to quantify the number of acres present today that represent historically occurring forest ecosystem conditions in terms of key characteristics such as species composition, structure, and historical disturbance processes are a primary concern when using a coarse-filter strategy for biodiversity conservation. While most lands within the BSLRP area still provide forest conditions dominated by native plant species and continue to be used by a diversity of wildlife, in many cases native ecosystems are present in different amounts than occurred historically. Figures 31-33 display the specific ecosystems differing significantly under today's conditions when compared to the modeled historical ecosystems. Additionally, figure 34 depicts changes in the amount of each of the twelve disturbance states across all upland forested ecological sites within each ecoregion. Figure 35 displays changes in canopy cover classes across all disturbance states and ecological sites within each ecoregion. Figure 36 displays changes in tree size classes across all disturbance states and ecological sites within each ecoregion.

Fire regimes were also compared between modeled historical landscapes and today's conditions. Figure 37 displays the overall fire regime changes occurring within each ecoregion. Figures 38-40 display the range in variability for modeled historical fire regimes as compared to today's fire regimes across ecological sites and for each of the 3 ecoregions.

TREE SIZE CLASS	MOD WARM-DRY			MOD WARM-MOD DRY			MOD WARM MOIST			MOD COOL-MOIST		
	Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	12.8 >1000			1.6 176			3.0 212			3.6 290		
SAPLING - SMALL TREE	17.0 >1000		0.6	18.7 893		-0.02 91	7.0 344		1.9 734	8.9 549		5.8 >1000
MEDIUM TREE	13.4 >1000	11.7 >1000	18.4 >1000	5.8 >1000	12.9 >1000	33.3 >1000	6.1 >1000	7.6 >1000	22.6 >1000	3.6 >1000	5.5 >1000	27.8 >1000
LARGE TREE	5.9 >1000	10.8 >1000	3.8 >1000	1.8 238	8.2 >1000	8.4 >1000	2.0 249	13.4 >1000	20.6 >1000	0.5 139	7.8 954	24.1 >1000
VERY LARGE TREE	-68.7 3	-23.3 6	-2.3 14	-38.5 0	-32.5 3	-19.6 2	-33.2 5	-29.4 10	-21.8 4	-31.2 3	-32.0 6	-24.6 6
ACRES	23609			3404			37270			39153		

TREE SIZE CLASS	COOL-MOIST			COOL-MOD DRY			COLD			TIMBERLINE		
	Canopy Cover			Canopy Cover			Canopy Cover			Canopy Cover		
	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed	Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	-7.3 56			7.3 154			9.2 336			24.2 621		
SAPLING - SMALL TREE	-4.4 75		-0.8 65	4.9 135		-1.3 24	11.6 279		-0.1 44	26.8 425		-0.1 0
MEDIUM TREE	5.3 311	5.5 278	17.3 910	7.0 418	4.5 287	13.7 987	19.9 >1000	12.4 >1000	10.6 >1000	16.5 804	7.7 771	1.2 698
LARGE TREE	-0.7 85	6.5 232	4.1 222	-3.7 56	2.7 138	1.3 133	-26.9 13	-12.4 54	-7.7 34	-25.8 13	-19.8 14	-9.8 0
VERY LARGE TREE	-11.9 20	-5.4 62	-8.2 37	-19.3 13	-10.7 28	-6.5 19	-10.0 0	-4.1 0	-2.4 0	-11.7 0	-5.1 0	-4.0 0
ACRES	146600			45242			30277			3275		

within historical range of variability
less than historical range of variabiliity
more than historical range of variability

Figure 31. The cumulative change in upland forest ecosystems for the M333C ecoregion. Historical range of variability represents the 95% confidence interval for historical amounts of each ecosystem so current conditions less or more than the historical range of variability fall outside of 95% of historical values. Numbers in the second row of each cell represent the current representation of the mean historical amount of each ecosystem as a percentage of historical amounts. Amounts greater than 100 indicate an increased amount while numbers form 0-99 represent the remaining percentage representation of that ecosystem.

TREE SIZE CLASS	WARM-DRY Canopy Cover Open	Moderate	Closed	MOD WARM-DRY Canopy Cover Open	Moderate	Closed	COOL-MOIST Canopy Cover Open	Moderate	Closed	COOL-MOD DRY Canopy Cover Open	Moderate	Closed	COLD Canopy Cover Open	Moderate	Closed	TIMBERLINE Canopy Cover Open	Moderate	Closed
GRASS-FORB-SHRUB-SEEDLING	29.9 >1000			18.0 >1000			1.6 111			6.3 136			11.3 321			11.1 237		
SAPLING - SMALL TREE	20.9 >1000		8.1 >1000	4.4 >1000		1.6 >1000	-6.1 64		2.1 227	-6.0 68		1.3 157	3.6 143		2.1 884	4.1 132		1.2 278
MEDIUM TREE	6.5 >1000	7.8 >1000	3.3 >1000	7.7 >1000	10.2 >1000	10.7 >1000	0.7 129	4.5 256	17.3 >1000	0.9 132	3.1 196	13.9 806	9.4 516	11.2 924	13.5 >1000	22.9 886	10.7 598	4.8 601
LARGE TREE	3.0 >1000	6.1 >1000	1.6 >1000	0.8 297	5.2 >1000	13.6 >1000	-1.4 71	2.1 147	10.2 428	-1.9 66	-0.3 95	7.7 358	-26.2 17	-10.3 57	4.0 151	-21.8 19	-12.9 42	-5.2 39
VERY LARGE TREE	-73.6 4	-14.2 30	0.5 135	-71.0 7	-4.5 77	3.3 239	-15.7 14	-8.6 46	-6.7 47	-11.0 21	-6.4 50	-7.5 39	-12.1 0	-4.9 0	-1.6 0	-9.2 0	-4.0 0	-1.7 0
ACRES	6047			172079			151228			144974			36736			8467		

within historical range of variability
less than historical range of variability
more than historical range of variability

Figure 32. The cumulative change in upland forest ecosystems for the M332B-West ecoregion. Historical range of variability represents the 95% confidence interval for historical amounts of each ecosystem so current conditions less or more than the historical range of variability fall outside of 95% of historical values. Numbers in the second row of each cell represent the current representation of the mean historical amount of each ecosystem as a percentage of historical amounts. Amounts greater than 100 indicate an increased amount while numbers form 0-99 represent the remaining percentage representation of that ecosystem.

TREE SIZE CLASS	HOT-DRY Open	HOT-DRY Moderate	HOT-DRY Closed	WARM-DRY Open	WARM-DRY Moderate	WARM-DRY Closed	MOD WARM-DRY Open	MOD WARM-DRY Moderate	MOD WARM-DRY Closed	COOL-MOIST Open	COOL-MOIST Moderate	COOL-MOIST Closed	COOL-MOD DRY Open	COOL-MOD DRY Moderate	COOL-MOD DRY Closed	COLD Open	COLD Moderate	COLD Closed	TIMBERLINE Open	TIMBERLINE Moderate	TIMBERLINE Closed
GRASS-FORB-SHRUB-SEEDLING	28.8 >1000			1.1 >1000			4.6 >1000			-13.6 0			-11.9 40			-6.6 0			-6.8 0		
SAPLING - SMALL TREE	9.8 >1000		0.4 >1000	3.8 >1000		0.3 100	14.0 >1000		1.1 >1000	3.2 122		4.0 463	10.0 161		3.6 266	-0.5 95		3.4 100	5.7 157		3.2 958
MEDIUM TREE	9.5 >1000	7.8 >1000	3.3 >1000	18.0 >1000	13.5 >1000	6.4 >1000	4.4 >1000	25.1 >1000	18.6 >1000	2.6 195	26.2 >1000	29.9 >1000	1.9 165	17.2 834	25.8 >1000	13.9 614	21.7 >1000	45.7 >1000	7.2 364	33.2 >1000	32.2 >1000
LARGE TREE	-50.4 20	-26.6 28	_-0.03 91	15.7 >1000	18.8 >1000	2.0 >1000	5.8 >1000	12.9 >1000	1.9 >1000	-4.0 24	2.2 178	-0.1 97	-5.1 21	-0.5 87	-0.6 70	-38.3 1	-16.6 2	-2.8 30	-37.7 1	-14.4 10	-2.6 35
VERY LARGE TREE	10.8 >1000	6.3 >1000	0.2 364	-78.6 14	-0.9 89	0.0 92	-77.2 7	-9.6 30	-1.6 7	-30.1 2	-12.1 18	-8.5 9	-19.9 4	-11.0 13	-9.5 3	-13.9 0	-4.4 0	-1.6 0	-14.8 2	-3.9 5	-1.3 0
ACRES	668			4919			80110			75932			98562			21025			4157		

Canopy Cover: Open, Moderate, Closed

- within historical range of variability
- less than historical range of variability
- more than historical range of variability

Figure 33. The cumulative change in upland forest ecosystems for the M332B-East ecoregion. Historical range of variability represents the 95% confidence interval for historical amounts of each ecosystem so current conditions less or more than the historical range of variability fall outside of 95% of historical values. Numbers in the second row of each cell represent the current representation of the mean historical amount of each ecosystem as a percentage of historical amounts. Amounts greater than 100 indicate an increased amount while numbers form 0-99 represent the remaining percentage representation of that ecosystem.

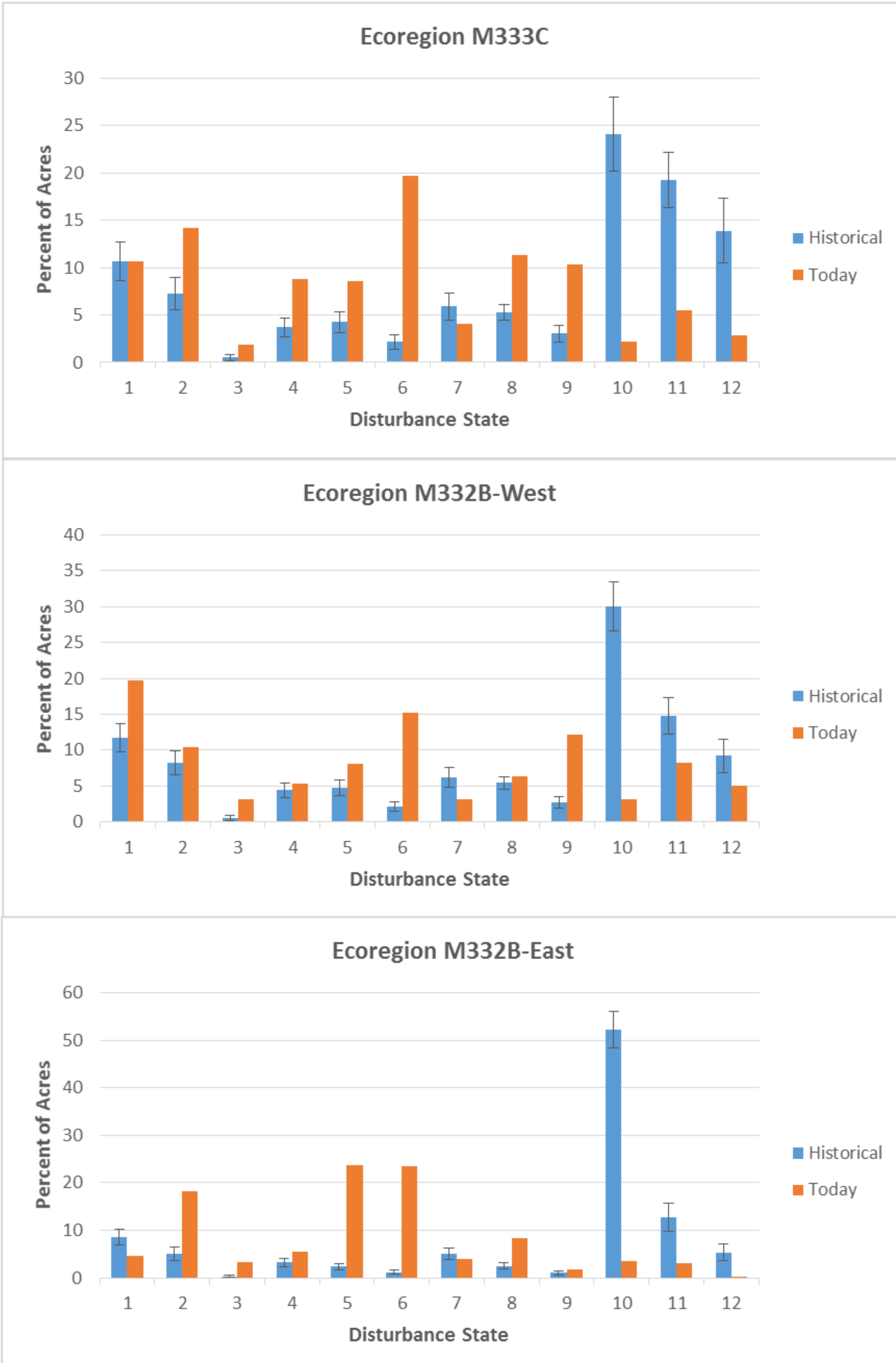


Figure 34. Comparisons current versus historical amounts of disturbance states occurring across all ecological sites in each of the 3 ecoregions for the BLSRP area.

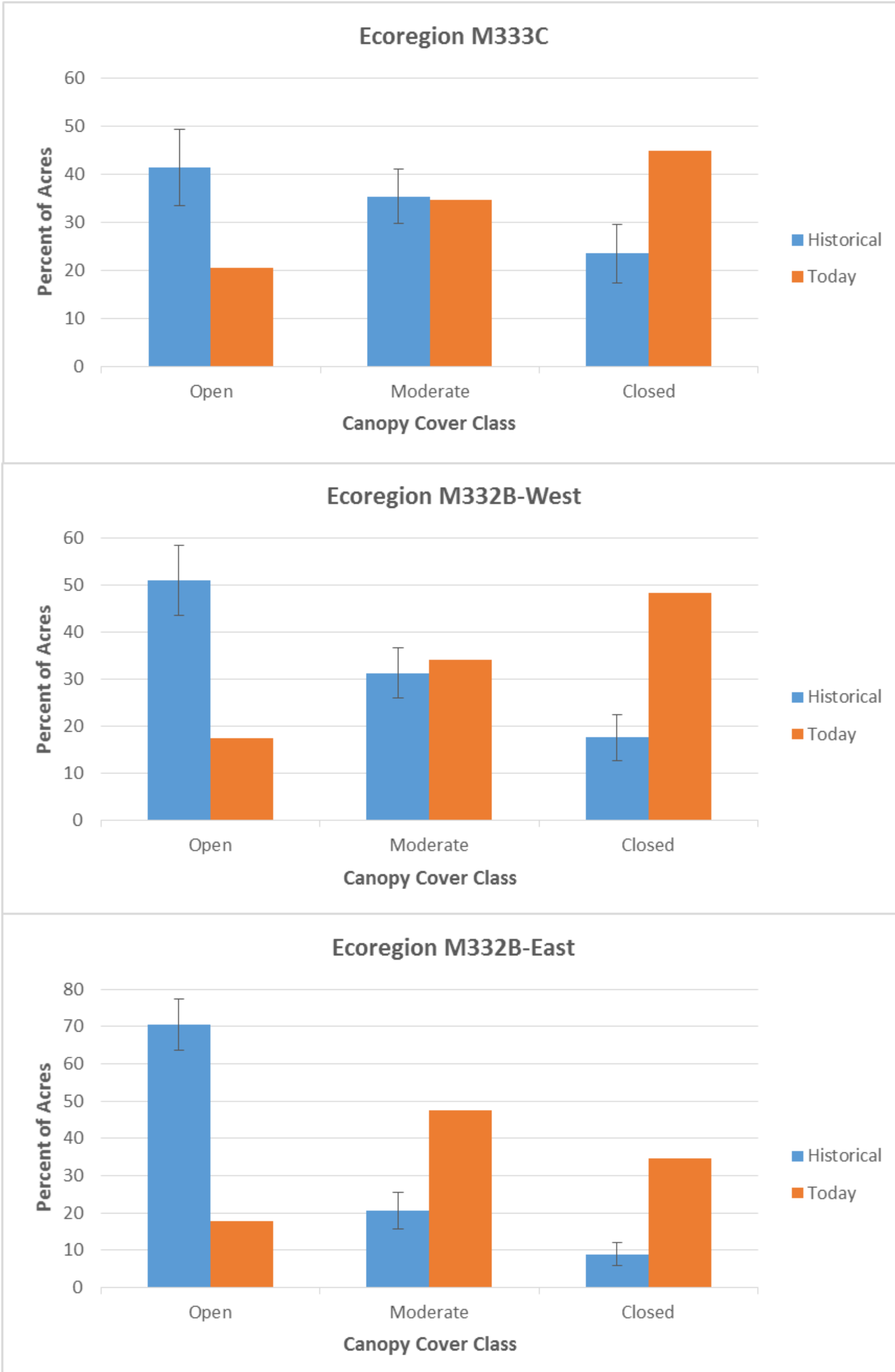


Figure 35. Comparison of percentage of forested area within each ecoregion that was historically in open, moderate, or closed canopy conditions compared to the percentages occurring today.

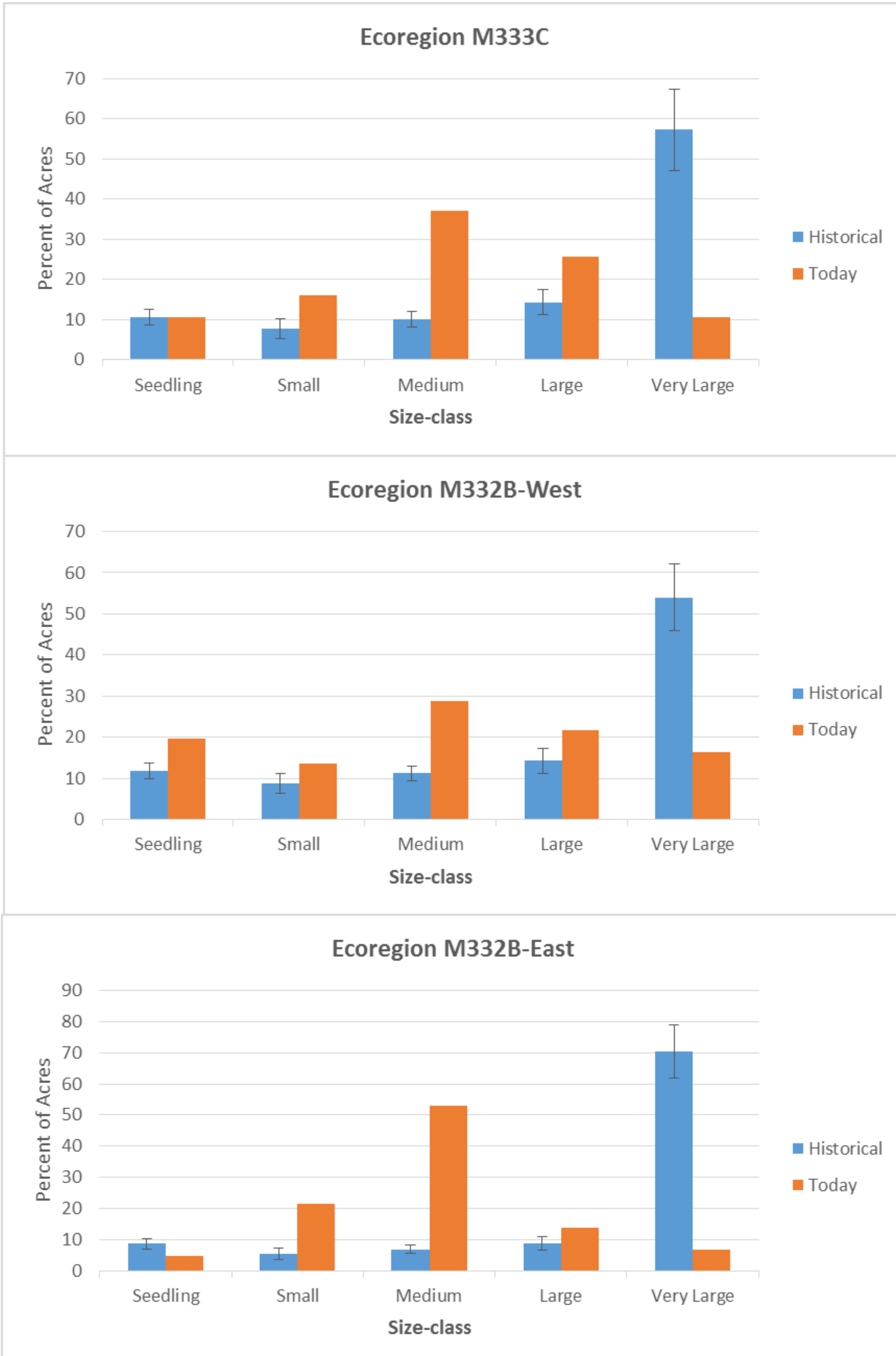


Figure 36. Comparison of the percentage of forested area within each of the three ecoregions in the project area that was historically comprised of ecosystems in each size class of trees compared to the size class distribution occurring today.

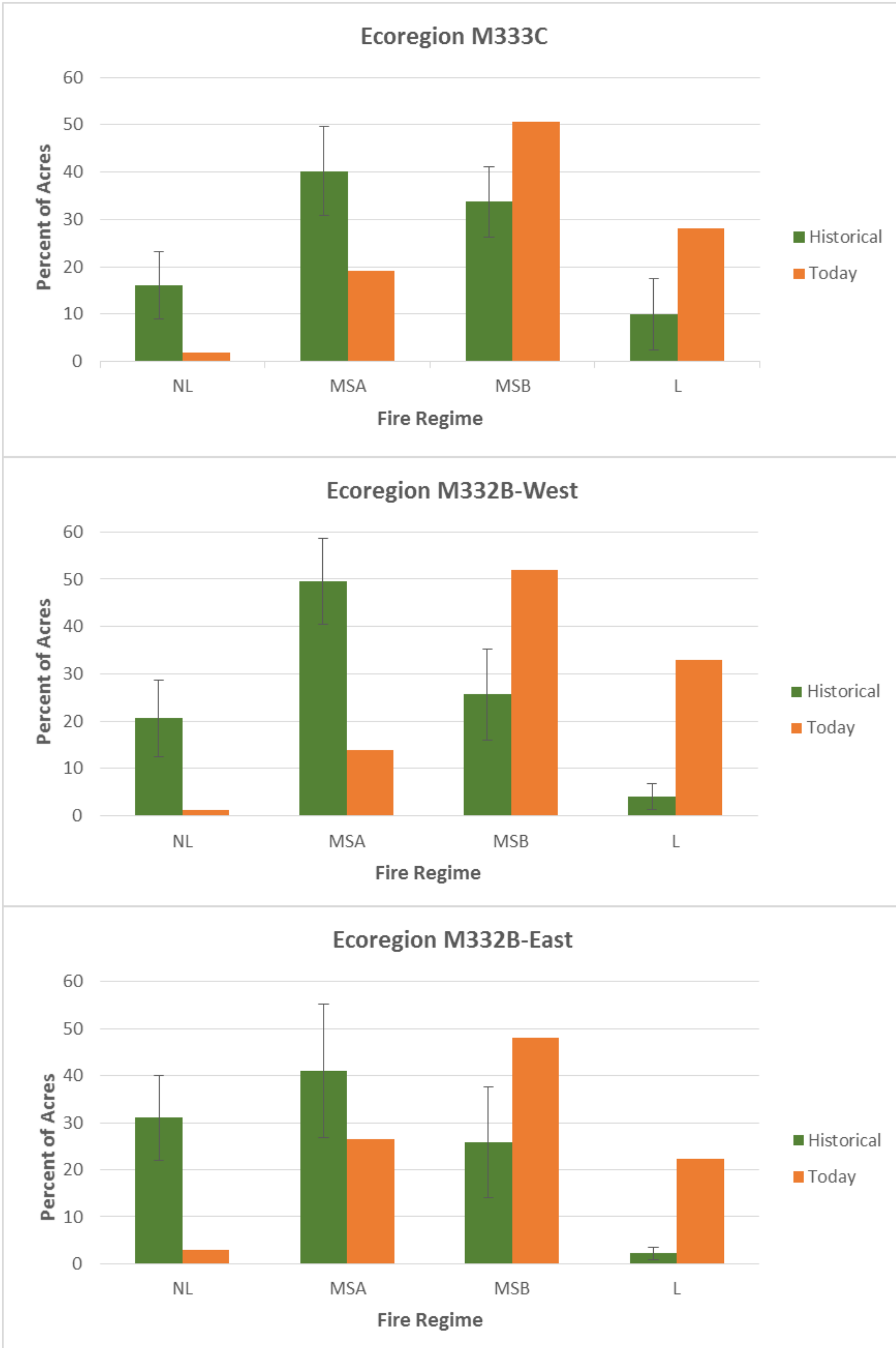


Figure 37. Historical fire regimes based on estimated fire severities of stand conditions within the 3 ecoregions of the project area compared to current conditions.

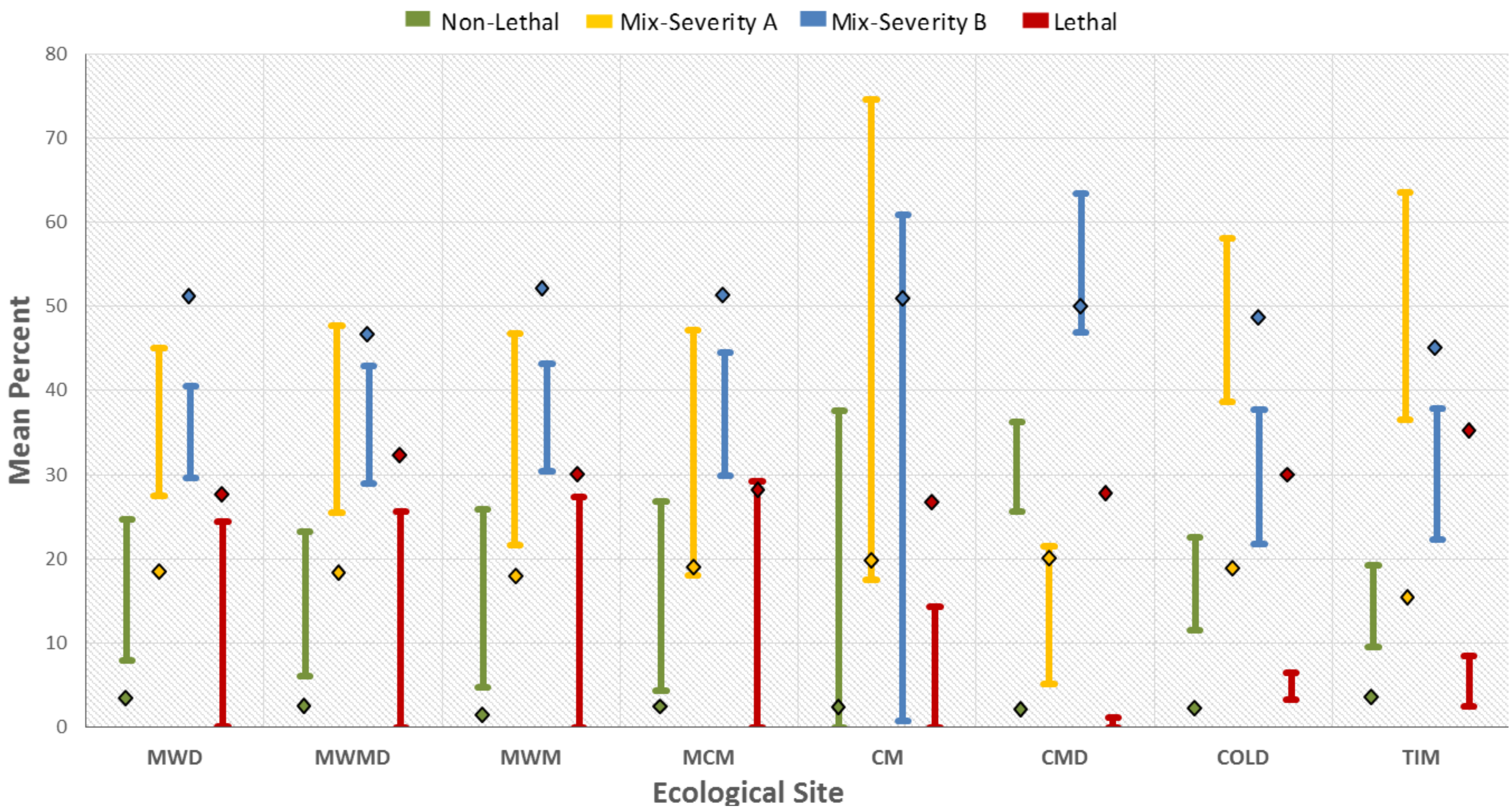


Figure 38. Historical fire regimes (95% confidence interval range) compared to today's fire regime occurrence (single point) for each ecological site in ecoregion M333C.

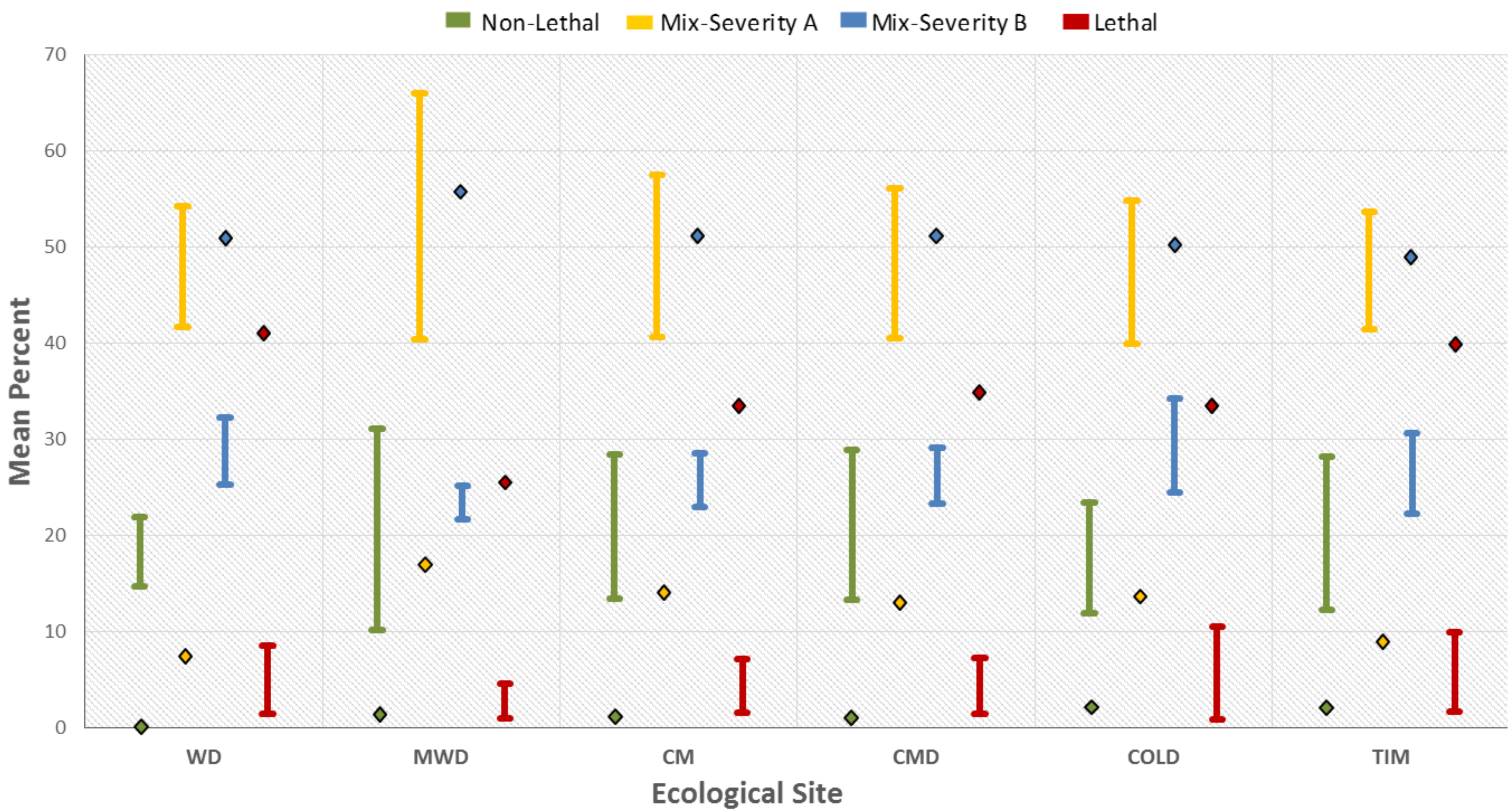


Figure 39. Historical fire regimes (95% confidence interval range) compared to today's fire regime occurrence (single point) for each ecological site in ecoregion M332B-West.

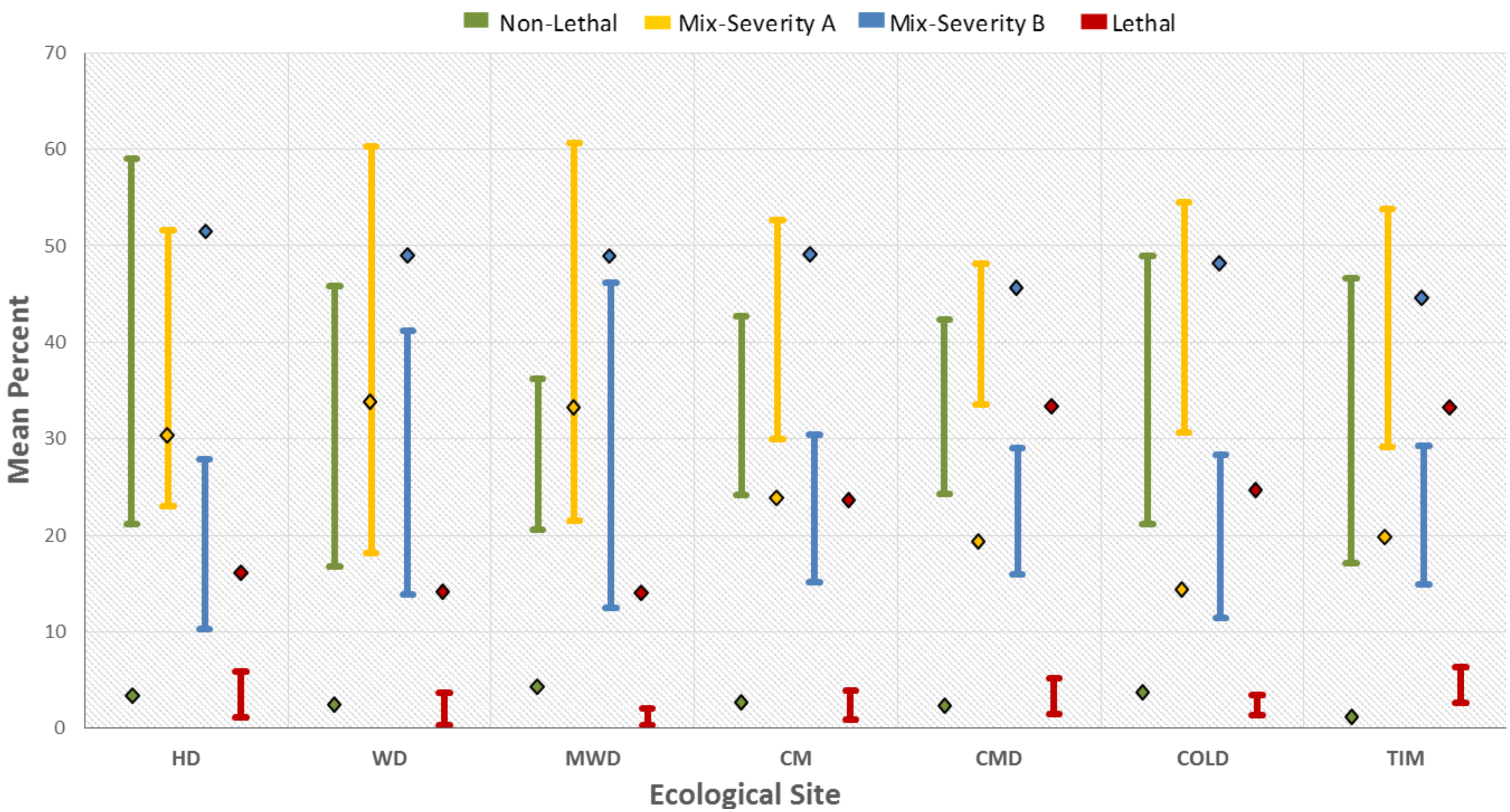


Figure 40. Historical fire regimes (95% confidence interval range) compared to today's fire regime occurrence (single point) for each ecological site in ecoregion M332B-East.

Discussion of Cumulative Changes to Forest Ecosystem Diversity

Direct conversion of forest ecosystems to other land uses has not been a major factor within the BSLRP area. While the impacts of direct conversion to other resources, such as the effects of roads on streams or the effects of the human population on grizzly bears may be much more significant than indicated by the footprint, the impacts to forest ecosystem diversity have been relatively minor. However, the cumulative changes to forest ecosystem diversity resulting from indirect human causes has been significant. Primarily these include changes to forest ecosystem diversity resulting from fire exclusion practices, past logging activity, introduction of exotic insects and diseases, and introduction of exotic species of weeds. This assessment did not evaluate the effects of exotic weeds on ecosystem diversity but it should be noted they could have significant influences on species composition in some locations, particularly in the understory.

Currently, forest ecosystems are for the most part no longer influenced by the non-lethal fire regime and the extent and distribution of the mixed-severity A fire regime has been greatly reduced (Figures 37-40). This has shifted current forest conditions to being influenced by significantly greater amounts of lethal and mixed severity B fire regimes (Figures 37-40). While this is true for all 3 ecoregions, these effects become greater as one moves from the M333C ecoregion through the M332B-West ecoregion and into the M332B-East ecoregion.

The M333C ecoregion included higher percentages of moister ecological sites due to climate influences. These sites, historically, had longer fire return intervals and greater percentages of closed canopy conditions than drier ecological sites. This ecoregion had slightly less deviation from historical conditions in terms of changes to ecosystem diversity and fire regimes than the other 2 ecoregions. Combining non-lethal and mixed-severity A fire regimes, both M332B-east and M332B-west had over 70% of their area in these combined fire regimes, on average, while M333C had only about 45% in these two fire regimes combined. Nonetheless, there are still clear needs for restoration across all 3 ecoregions, especially for the mixed severity A fire regime and associated open to moderate canopy cover disturbance states.

Examination of changes to fire regimes by ecological sites (Figures 38-40) highlights the restoration needs discussed above. Ecoregion M333C showed today's conditions are still within the historical 95% confidence intervals for some ecological sites, although even for these, most were near the outer limits of the confidence interval. Many other ecological sites were outside the historical confidence intervals, especially for the non-lethal fire regime. These differences were even more pronounced in the M3332B-West and M332B-East ecoregions, which were quite similar in their departures between current and historical conditions. An exception was the mixed severity A fire regime in M332B-East for the hot dry, warm dry, and moderately warm dry ecological sites where existing conditions were still estimated to be within the historical confidence interval for this fire regime. However, most of the non-lethal fire regimes for this ecoregion showed greater departures from historical than for the M3332B-West ecoregion.

Restoring today's fire regimes to more characteristic historical conditions will require shifting canopy cover to more open or moderate conditions. This should shift the percentage of area likely to burn at high severity to low or moderate severity. Re-establishing historical fire regimes will require a consideration of current vegetation patterns, particularly canopy cover and size classes, and should be integrated with specific ecosystem diversity objectives such as addressing an overabundance of medium tree structural conditons in the BSLRP area. Identifying where there is good potential for transitioning to larger size classes, particularly in open to moderate canopy conditions and the fire tolerant species associated with these ecosystems should be a priority. Alternatively, determining where early successional conditions should occur is particularly important from a fire regime standpoint.

The analysis of cumulative changes to disturbance states revealed two significant trends. The first being the increase in canopy cover in today's forests when compared to historical conditions (Figure 35). This is explained by the reduction in the frequently occurring low and moderate severity fire on the landscape today that historically functioned to produce ecosystems characterized by more open canopies of fire tolerant/shade intolerant species, particularly for the mid to low elevation ecological sites. The result is an increase in the amount of infrequently occurring high severity fires that support the establishment of higher canopy covers and the associated shade tolerant/fire intolerant species, where they would have been relatively rare historically.

The second significant trend was the reduction in stands containing very large trees in the project area (Figure 36). This trend was shown by both the VMAP classification as well as by the limited FIA plot data analysis. As mentioned previously, the limitations of satellite imagery classification may not accurately show the full distribution of large and very large trees in the landscape. Based on the VMAP analysis, Figure 36 reveals, on average, there were 53-70% of each landscape classified as the very large tree size-class historically, while currently there is only 5-18% classified as very large tree. The FIA plot data, while showing a range of 9-34% of plots in the 3 ecoregions classified as the very large tree size class using the sequential stand analysis method, overall only 15% of the available plots contained enough very large trees to be classified as such. Even with the limitations of the existing data sources, it is clear significantly fewer very large tree ecosystems occur today in the landscape compared to those occurring historically. This is not necessarily surprising given past logging practices that targeted large diameter trees along with the long timeframes required for their growth. The remnants of many of these very large trees are still visible today where their stumps remain providing evidence of the springboard notches used when they were harvested in the early 20th century. Thus, the cumulative change analysis reveals the need for restoration of very large tree ecosystems where they were common historically, across all three ecoregions. Hessburg et al. (2016) similarly reported on this need.

The cumulative change analysis quantified the greatest needs for ecosystem diversity restoration. Ecoregion M333C exhibited a number of ecosystems having very low rates of representation (or high departures) when compared to their estimated historical amounts. As discussed, very large tree ecosystems have the lowest overall level of representation in the landscape today and the greatest departure from historical. The moderately warm and dry ecological site had only 3% representation of its mean estimated historical amount of the open canopy, very large tree ecosystem, a decline of 69%. While this ecological site only represents 23,609 acres in this ecoregion, this still indicates a high priority for restoration. The moderate canopy cover - very large tree ecosystem for this same ecological site only had 6% representation today, also making it a priority for restoration. The moderately warm and moderately dry ecological site had even lower levels of very large tree ecosystems represented today, although this ecological site only occurred across 3404 acres in this ecoregion. The moderately warm and moist ecological site, occurring on 37,270 acres, had 5%, 10%, and 4% representation of the open, moderate, and closed canopy very large tree ecosystems respectively, while the moderately cool-moist ecological site had 3%, 6%, and 6% of these same disturbance states for this site. The cold and timberline ecological sites had no representation of very large tree structures, but for these ecological sites, the large tree ecosystems were more abundant historically due to species characteristics and the limitations of the extreme environment on growth potential. However, these large tree ecosystems were found to be poorly represented as well. In addition to the reduction in large and very large trees, losses of whitebark pine in these ecosystems from the combined effects of fire suppression, white pine blister rust, and bark beetles have further altered these ecosystems. An additional ecosystem restoration need in ecoregion M333C is for those ecosystems historically dominated by western white pine. These ecosystems did not occupy large areas historically; Ayres (1900) estimated its occurrence at 0.5% of sawlogs in the area he surveyed. However, it was still an important ecosystem that has been heavily impacted today through the combination of past logging practices and white pine blister rust.

Results of the cumulative change analysis for ecoregion M332B-West identifies a major restoration priority is the open canopy, very large tree ecosystem in the moderately warm and dry ecological site, having only an estimated 7% representation of its historical amounts and thus experiencing an average departure of 71%, for a loss of over 122,000 acres. The open, very large tree disturbance state in the warm and dry ecological site has only 4% estimated representation of historical conditions, but the relatively small amount of this site in the ecoregion reduces the impact of this change. Other ecosystem restoration priorities include the open canopy, large and very large tree size classes in the cold and timberline ecological sites. As with M333C, these types have been impacted by lack of fire, white pine blister rust, and beetle infestations of whitebark pine. Very large tree ecosytems, especially in the open canopy conditions, are also needed in the cool and moist and cool and moderately dry ecological sites, as these have only an estimated representation of 14 and 21% respectively.

Results of the cumulative change analysis for ecoregion M3332B-East is similar to M332B-West in that the open canopy-very large tree ecosystem in the warm and dry, moderately warm and dry, cool and moist, and cool and moderately dry ecological sites have been greatly reduced from historical amounts, having only an estimated 14%, 7%, 2%, and 4%, respectively, level of representation today. This is a high restoration priority for the latter three ecological sites, as they represent a large amount of acres in the ecoregion. Further, for all three of these ecological sites, the moderate and closed canopy very large tree ecosystems also had low levels of representation compared to historical amounts. Similar to the other two ecoregions, the cold and timberline ecological sites have significant reductions in the open, large and very large tree ecosystems, and also in the moderate canopy, large tree ecosystems. Restoration of these ecosystems, particularly emphasizing whitebark pine, is a priority.

Accordingly, the greatest overall restoration priorities are for open canopy, very large tree ecosystems and in particular those occurring on the moderately warm and dry, cool and moist, and cool and moderately dry ecological sites. Restoration of these conditions can occur in two ways. One method is to identify stands with enough very large trees present to meet this structural requirement but that are currently in higher canopy cover classes. The restoration goal for these stands would be to re-establish reference conditions for the open canopy disturbance state on that ecological site, using the most appropriate treatment options. For example, this might include examining the species composition of the very large tree component. Where more closed canopy conditions contain very large trees dominated by western larch and ponderosa pine, or Douglas-fir in some cases, this would indicate the appropriate structures and species compositions are present for restoration of the desired ecosystems. If the very large tree component is dominated by subalpine-fir and/or Engelmann spruce, these stands would not require restoration and would be left to represent the closed canopy, very large tree ecosystems. The second method for restoration is to find large and medium tree ecosystems that can be treated to enhance growth of the desired open canopy species (western larch, ponderosa pine, and to a lesser extent Douglas-fir), and plan for these ecosystems to eventually grow into the desired ecosystems, exhibiting the appropriate reference conditions for each ecological site. While this is a long-term restoration approach, it will be necessary to provide needed levels of representation into the future. Hessburg et al. (2016) recommended similar restoration approaches for areas that historically supported primarily mixed severity fire regimes.

Restoration of whitebark pine and western white pine ecosystems is challenged at this time by the continuing influence of white pine blister rust, and the increasing influence of bark beetles attributed to climate change. Restoring fire to these ecological sites, especially where whitebark pine still occurs, can help maintain this species and the needed open and moderate canopy cover ecosystems. In other locations, more intensive management, including planting of the desired species, may be required.

Species Assessment

The primary component of the BSLRP conservation strategy to meet sustainability and biodiversity objectives is to provide a coarse-filter representation of forest ecosystem diversity at both landscape and ecosystem levels. The secondary fine-filter or species assessment component of this strategy involves understanding how changes to ecosystem diversity have affected species of interest or concern. In addition, conducting a species assessment relative to desired future conditions provides an opportunity to evaluate and quantify the expected response of a species of concern to planned levels of ecosystem representation. The selection of species to include in the assessment will depend on the specific application of the strategy.

The BSLRP species assessment was conducted at the completion of the ecosystem diversity assessment and accomplished several things. First, the range of native ecosystem diversity can be used to assess and evaluate the inherent capability of the landscape to provide historical habitat conditions for a target species. This is an important consideration as some species may have never had high probabilities of persistence or viability in a landscape. Efforts to achieve viable and persistent populations for such species over the long-term may not be feasible in these landscapes, or such efforts may shift ecosystem diversity substantially away from what occurred historically, and ultimately undermine the scientific foundation of the conservation strategy for biodiversity. Thus, understanding the likely historical status of species of concern or interest in a landscape is important information for developing future management decisions. Second, species assessments provide information on the effects of cumulative changes to native ecosystem diversity on a selected species' habitat. While the current status of a species may be influenced by many factors including direct human impacts on populations, comparing historical habitat conditions to today's conditions provides a better understanding of the current status, both in terms of quality and amounts of habitat required by the species. Finally, assessing how changes in ecosystem diversity have affected species of interest can help evaluate planned restoration activities, and ensure that sufficient representation of ecosystem diversity is planned in order to support those species with high probabilities of persistence under historical conditions.

U.S. Forest Service wildlife biologists representing the Lolo, Flathead, and Helena National Forests, and the Swan and Seeley Lake Ranger Districts, identified 5 species of interest to include in the species assessment. These species were fisher, flammulated owl, northern goshawk, black-backed woodpecker, and pileated woodpecker.

Methods

Habitat suitability indices (HSI) where developed for each species based on review of the available literature. Several of these models had been previously developed for a SWCC wildlife monitoring project and were reviewed by the SWCC Wildlife Monitoring Working Group. The HSI models were used to assess both existing and historical conditions. For historical analyses, the ecosystem diversity outputs from the SIMPPLLE modeling were used. For current conditions, the VMAP layer provided by the BSLRP team and used in the existing condition analyses was used.

Each ecosystem in the ecosystem diversity framework was assigned HSI scores for each variable included in the HSI model for each species. Each ecosystem within each ecoregion was assigned values for each HSI variable based on available FIA plot data and TSMRS stand data to generate habitat values for each ecosystem. The mean value for each habitat variable was calculated using a Microsoft Access database. A complete list of habitat variables is included within the modeling description for each wildlife species (Appendix I). For existing conditions, all available plot data were used to calculate mean values. For historical conditions, only FIA plots that represented historical stands were included. These were typically stands that occurred in wilderness areas or had only experienced natural disturbances and not been harvested. Where needed, historical ecosystem conditions were adjusted as indicated in Table 13 to represent reported historical conditions rather than the existing conditions. Ecosystems that lacked empirical data from either source of plot data were assigned values based on extrapolation from adjoining ecosystems or from similar ecosystems in adjacent ecoregions.

For existing conditions two habitat variables were derived from VMAP data. These were percent tree canopy cover and tree size (dbh in inches). For historical conditions the values for tree canopy and tree size where taken from the SIMPPLLE analysis results.

Once each cell in the BSLRP landscape was assigned a value for each habitat variable from all the species models it was possible to calculate the HSI scores for each species. This was done in ESRI® ArcMap 10.4 using the Spatial Analysis extension to apply the HSI habitat formulas found in each species' model description. After calculating the HSI grid for an individual habitat variable it was possible to combine all the habitat HSI grids into a final HSI grid for each species. This process was the same for both existing and historical conditions.

The resulting habitat suitability map from either current conditions or the 5 historical periods based on the outputs of SIMPPLLE, revealed a range of expected high to low quality habitat based on the input of the key vegetation data. The habitat suitability maps were then used to estimate the number and quality of potential home ranges using a home range assessment method. HOMEGROWER is a program developed to automate the home range assessment method by aggregating the required elements into appropriate sized home ranges for each species within the planning landscape. Each species has minimum and maximum home range sizes that it will utilize. This process has been described by Roloff and Haufler (1997, 2002).

HOMEGROWER builds home ranges by evaluating the cells around a starting point of the highest quality habitat not already contained in a home range, and growing a new home range using the neighboring cells of highest quality. Cells are accumulated until the growth target, expressed as total HSI scores for that species has been met. HSI scores are tallied based on area multiplied by the habitat quality for each pixel that is added to the home range. The target for each species is based on a multiplier of its allometric home range. Allometric home ranges are the estimated minimum area that a species could occur in based on its estimated metabolic requirements. For consistency a rate of 5x the allometric home range was used to calculate the target home range size.

For example, if a bird has an allometric home range of 1 acre, its targeted home range requirements would be 5 acres or 5 HSI units. This could be met with a home range of 5 acres if all acres in that home range contributed 1.0 in HSI value, and would receive an overall home range quality of 1.0, and then be designated a high quality home range. However, this rarely occurs in the real world. Home ranges are typically comprised of patches of habitat for the species of varying quality. HOMEGROWER builds home ranges for a species by starting with a single cell of the highest quality in the landscape that has not already been included in another home range. It then grows by aggregating cells of the next highest quality until it has acquired the HSI units desired for the species, in this case, 5 HSI units. An upper threshold of size is set at 10 times the target size, or in this example 50 acres, beyond which HOMEGROWER ceases attempting to build a home range if the area becomes too large to be provide the necessary density of habitat required by the species. If in this example, HOMEGROWER identified a potential home range that took 8 acres to reach its target of 5 HSI units, it would be mapped as a home range, assigned an HSI value of 0.63, and would be designated a medium quality home range.

For this assessment, home ranges with total HSI values >0.75 were considered high quality home ranges, HSI values of 0.5-0.74 were considered medium quality home ranges, and HSI values of 0.1-0.49 were considered low quality. Roloff and Haufler (2002) discussed the implications of these ratings to their support of a species population. High quality home ranges are assumed to have high rates of occupancy, support high reproductive rates, and have high survival rates, thus providing good demographic support of the population of the species (Roloff and Haufler 2002). Lower quality home ranges have lower occurpancy rates and lower overall productivity. Kroll and Haufler (2010) documented these relationships to occur for occupancy rates and reproductive rates using empirical analysis of dusky flycatcher habitat in Idaho.

If enough high quality home ranges followed by medium quality home ranges occur, the species should do well in the landscape. If only low quality home ranges exist for the species, then the species viability is expected to have a lower probability in the landscape. While exact probability estimates for each species in the landscape are not computed, comparisons of amounts of high, medium, and low quality home ranges can be done between historical and current landscape conditions. This comparative approach to viability assessments, as opposed to efforts to directly estimate probabilities has been recommended as the most supportable way of using viability assessments (Beissinger et al. 2009, Ralls et al. 2002, Beissinger and Westphal 1998).

Because HOMEGROWER uses a random selection of the highest quality pixels available, it has a stochastic component. For existing conditions we ran 3 separate iterations of HOMEGROWER for each species, and calculated the mean values generated for numbers of home ranges. There was very little difference among the 3 runs for any species, so we determined that additional runs were not warranted. For historical conditions, HSI grids were created for 5 different time steps (every 200 years of the historical analysis) of the SIMPPLLE analysis.

For consistency a rate of 5x the allometric home range was used to calculate the target home range size. This resulted in the following targeted minimum possible home range sizes in HOMEGROWER:

- fisher – 3039 acres
- black-backed woodpecker – 60 acres
- flammulated owl – 42 acres
- northern goshawk – 825 acres
- pileated woodpecker – 305 acres

Results

SIMPPLLE modeling outputs were evaluated every 200 years of the 1000 year simulation for 5 separate habitat quality determinations for each species to capture a range of variability for species assessments. Figures 41, 42, 44, 45, 47, 48, 49, 52, 53, 55, and 56 show examples of the HSI maps for each species over several time steps as well as example home range estimates for these same example maps. For each species, the estimated number of high, medium and low quality home ranges from each time step as well as the 95% confidence interval for the mean values are presented. For each species, habitat suitability maps and home range maps are shown for two of the 5 modeled time steps. Decade 46 (400 years into the simulation modeling) was a time step that followed a normal to warmer and drier period. While decade 66 (600 years into the simulation modeling) followed a normal to cooler and moister period. These times captured some of the variability in conditions produced by disturbances in response to variable climate conditions. Tables 29, 31, 33, 35, and 37 present the estimated potential number of high, moderate, and low quality home ranges for the 5 modeled time steps for each species. Figures 43, 46, 50, 51, 54, 55,and 57 show maps of the current habitat qualities and estimated potential home ranges of varying quality for each species, while Tables 30, 32, 34, 36, and 38 show the comparison of current estimates of potential home ranges to the historical estimates.

Fisher

Historical Habitat Conditions

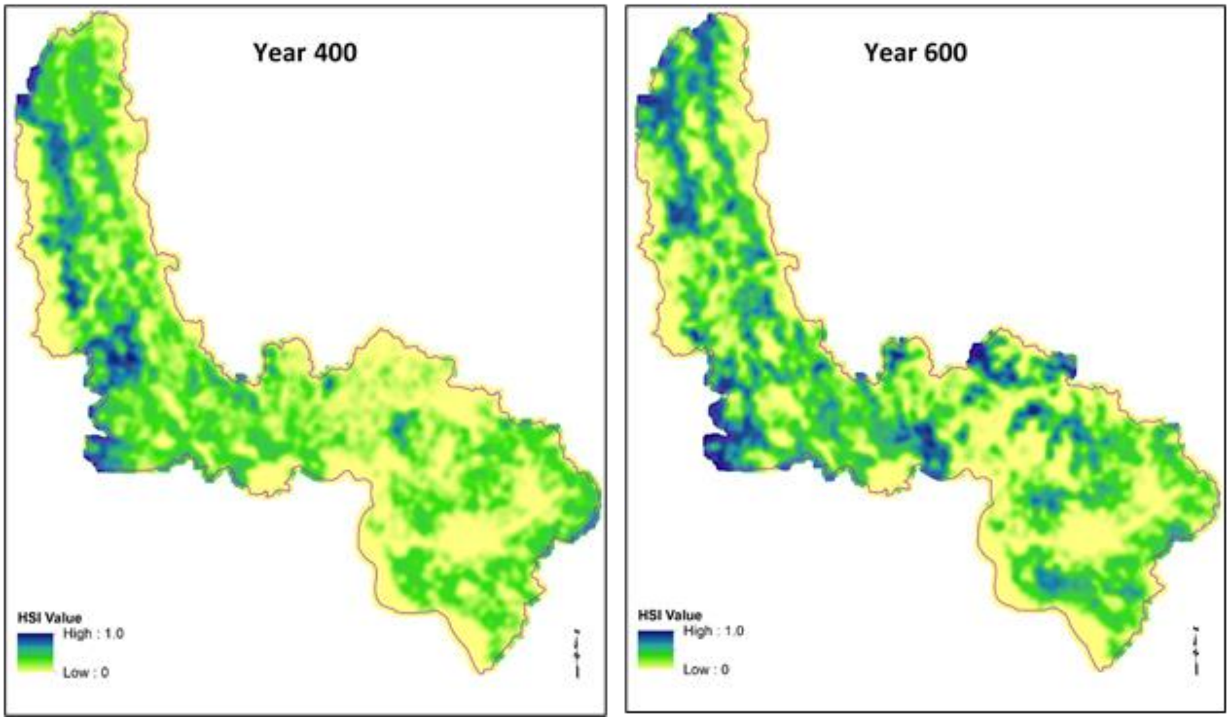


Figure 41. Maps of estimated habitat suitable for fisher showing outputs from two time steps produced through SIMPPLLE modeling of historical ecosystem conditions for the BSLRP area.

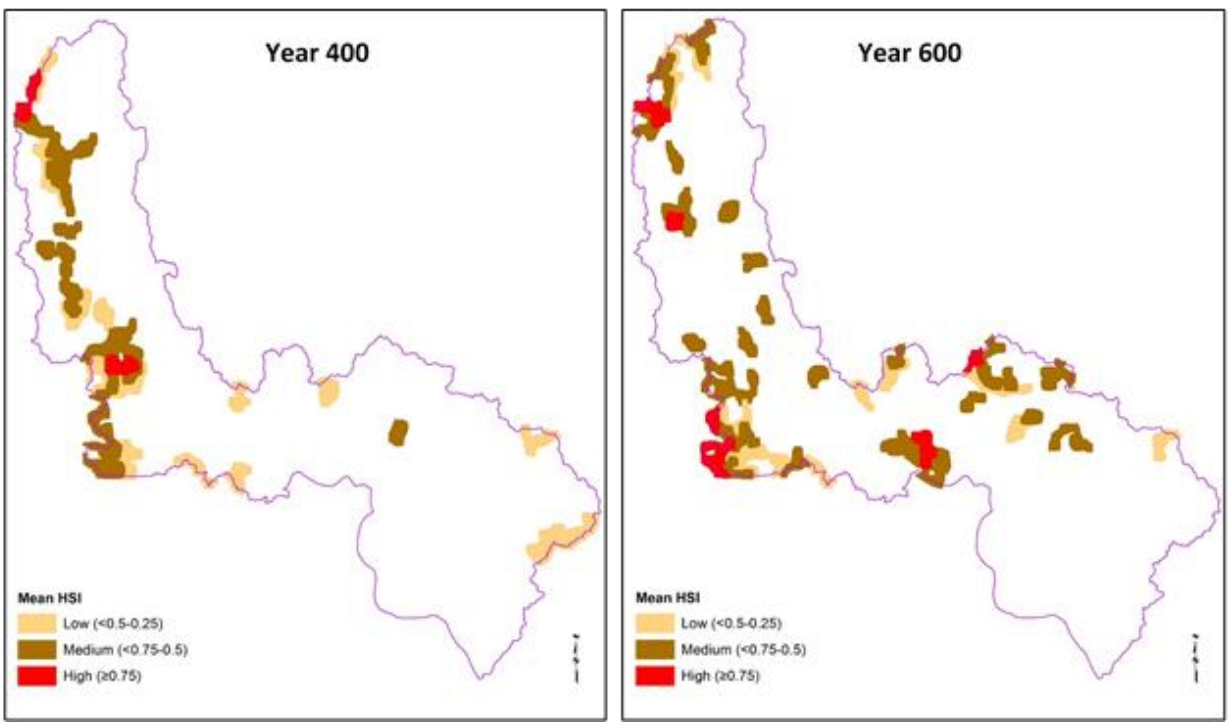


Figure 42. Maps of estimated potential home ranges of high, medium, and low quality for fisher produced from habitat suitability maps from outputs of two time steps of SIMPPLLE modeling for historical ecosystem conditions for the BSLRP area.

Table 29. Estimated numbers of potential home ranges of high, moderate and low quality for fisher from 5 time steps of SIMPPLLE historical modeling of ecosystem diversity for the BSLRP area.

Time Period	Home Range Quality		
	High	Moderate	Low
200	1	11	23
400	4	22	15
600	10	36	11
800	2	17	34
1000	1	11	22
Mean	**4**	**19**	**21**
CI (95%)	**1-7**	**11-28**	**14-28**

Current Habitat Conditions

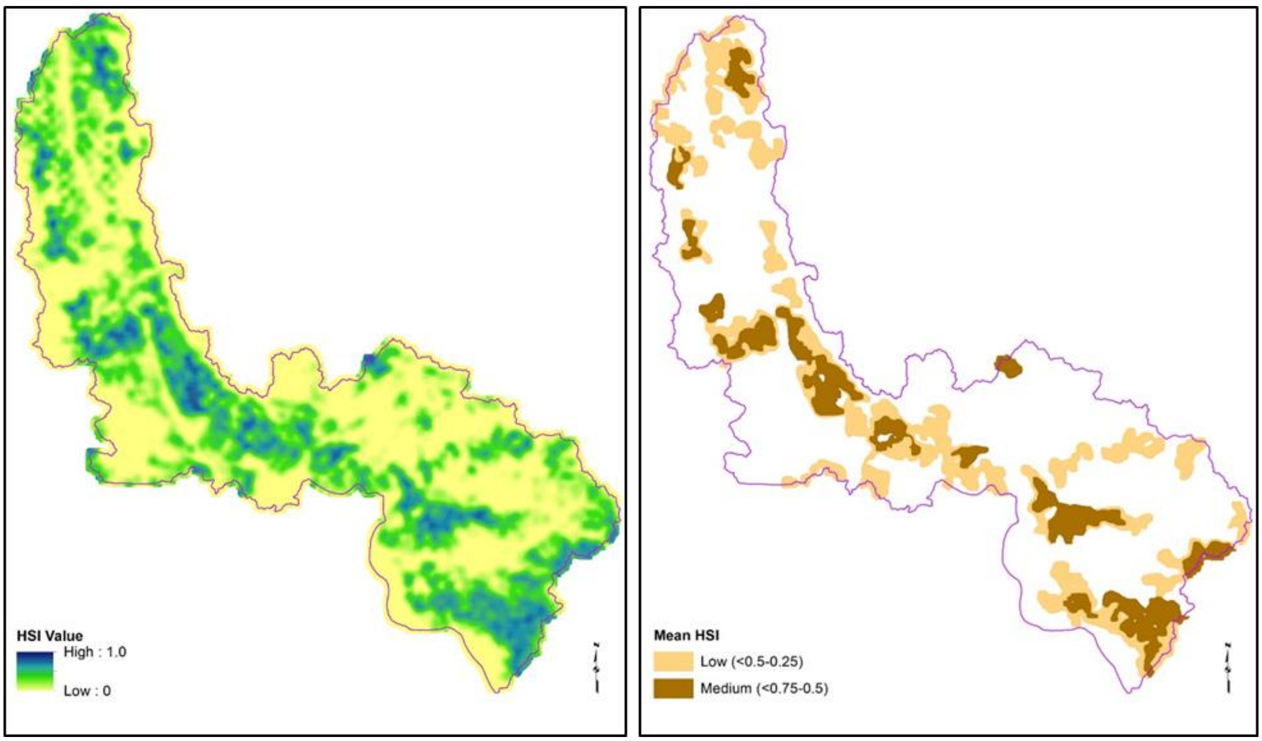


Figure 43. Current modeled habitat suitability and estimated potential home ranges of high, medium, or low quality for fisher for the BSLRP project area based on VMAP mapping of ecosystem diversity.

Table 30. Estimated numbers of home ranges of varying quality for fisher from historical modeling of ecosystem diversity compared to current conditions from VMAP mapping of ecosystem diversity for the BSLRP area.

Time Period		Home Range Quality		
		High	Moderate	Low
Historical	**Mean**	4	19	21
	CI (95%)	1-7	11-28	14-28
Today	**Mean**	0	32	5

Flammulated Owl

Historical

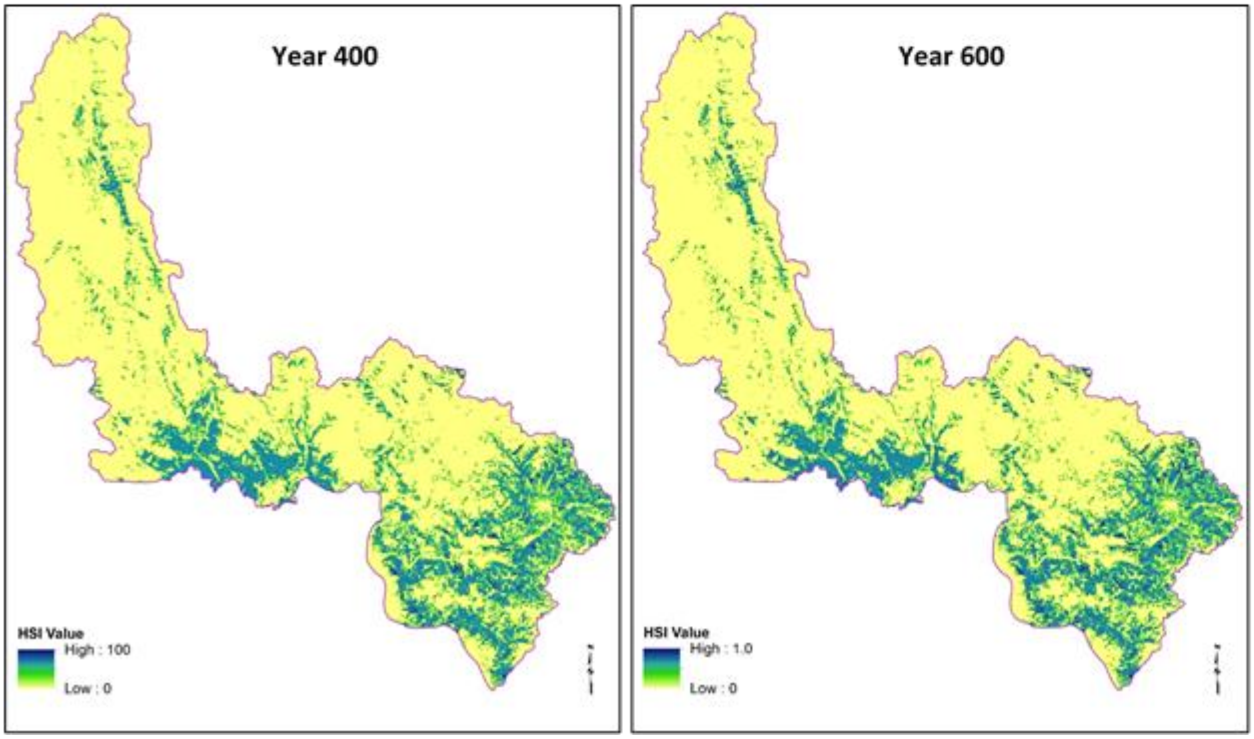


Figure 44. Maps of estimated habitat suitable for flammulated owl showing outputs from two time steps produced through SIMPPLLE modeling of historical ecosystem conditions for the BSLRP area.

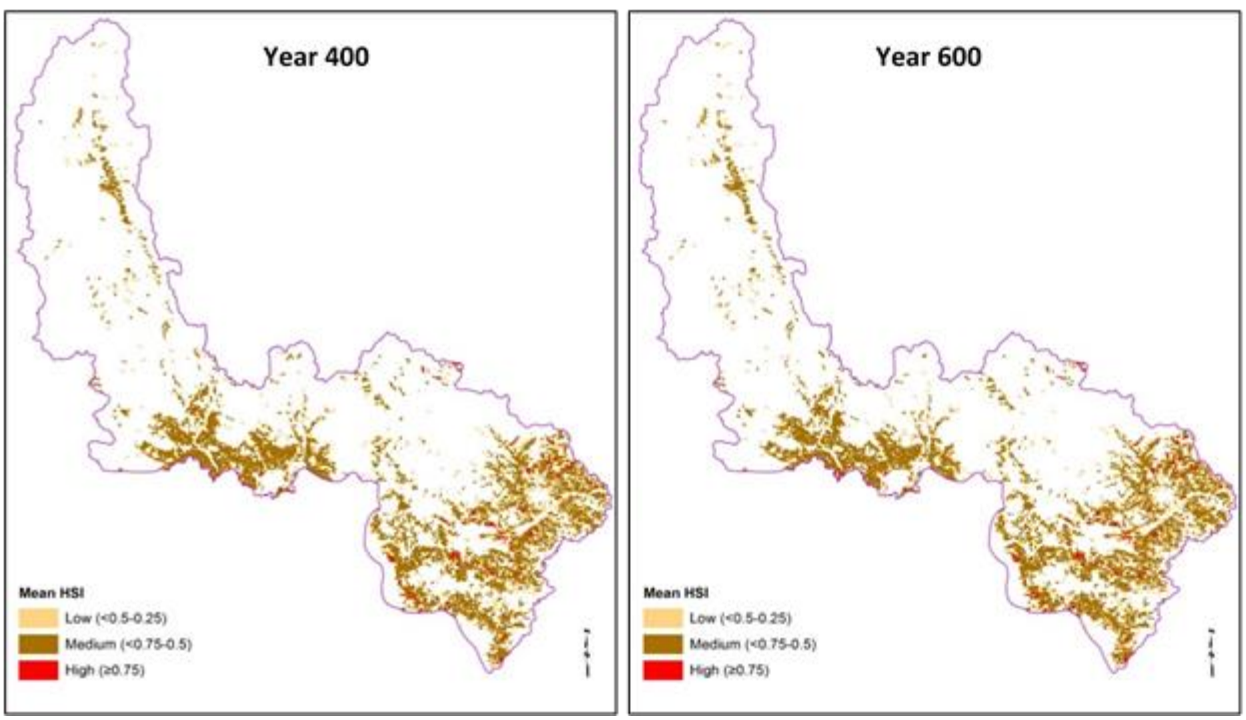


Figure 45. Maps of estimated potential home ranges of high, medium, and low quality for flammulated owl produced from habitat suitability maps from outputs of two time steps of SIMPPLLE modeling for historical ecosystem conditions for the BSLRP area.

Table 31. Estimated numbers of potential home ranges of high, moderate and low quality for flammulated owl for 5 time steps of SIMPPLLE historical modeling of ecosystem diversity for the BSLRP area.

Time Period	Home Range Quality High	Moderate	Low
200	148	3042	1142
400	145	3059	1109
600	159	3072	1109
800	151	3071	1114
1000	149	3065	1132
Mean	**150**	**3062**	**1121**
CI (95%)	**146-154**	**3052-3072**	**1109-1133**

Current

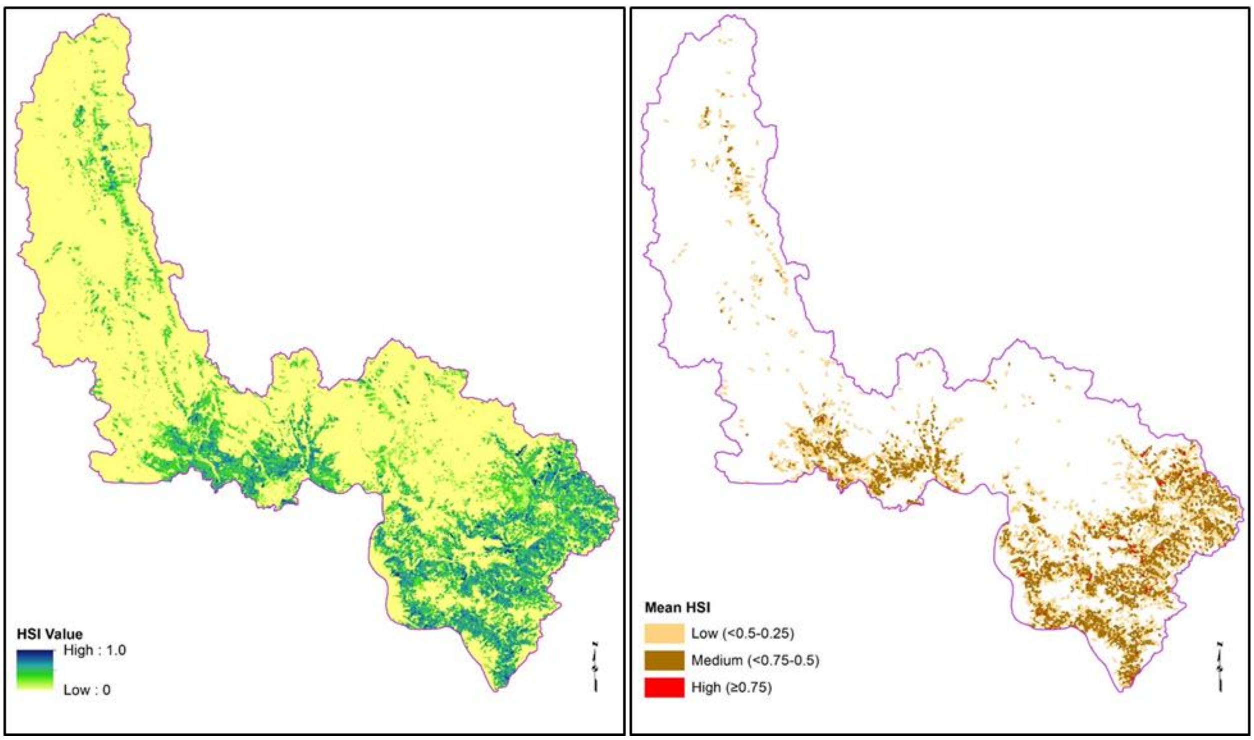


Figure 46. Current modeled habitat suitability and estimated potential home ranges of high, medium, or low quality for flammulated owl for the BSLRP project area based on VMAP mapping of ecosystem diversity.

Table 32. Estimated flammulated owl home range numbers of varying quality from historical modeling of ecosystem diversity as compared to current conditions developed from VMAP mapping for the BSLRP area.

Time Period		Home Range Quality High	Moderate	Low
Historical	**Mean**	150	3062	1121
	CI (95%)	146-154	3052-3072	1109-1133
Today	**Mean**	59	1538	319

Northern Goshawk

Historical

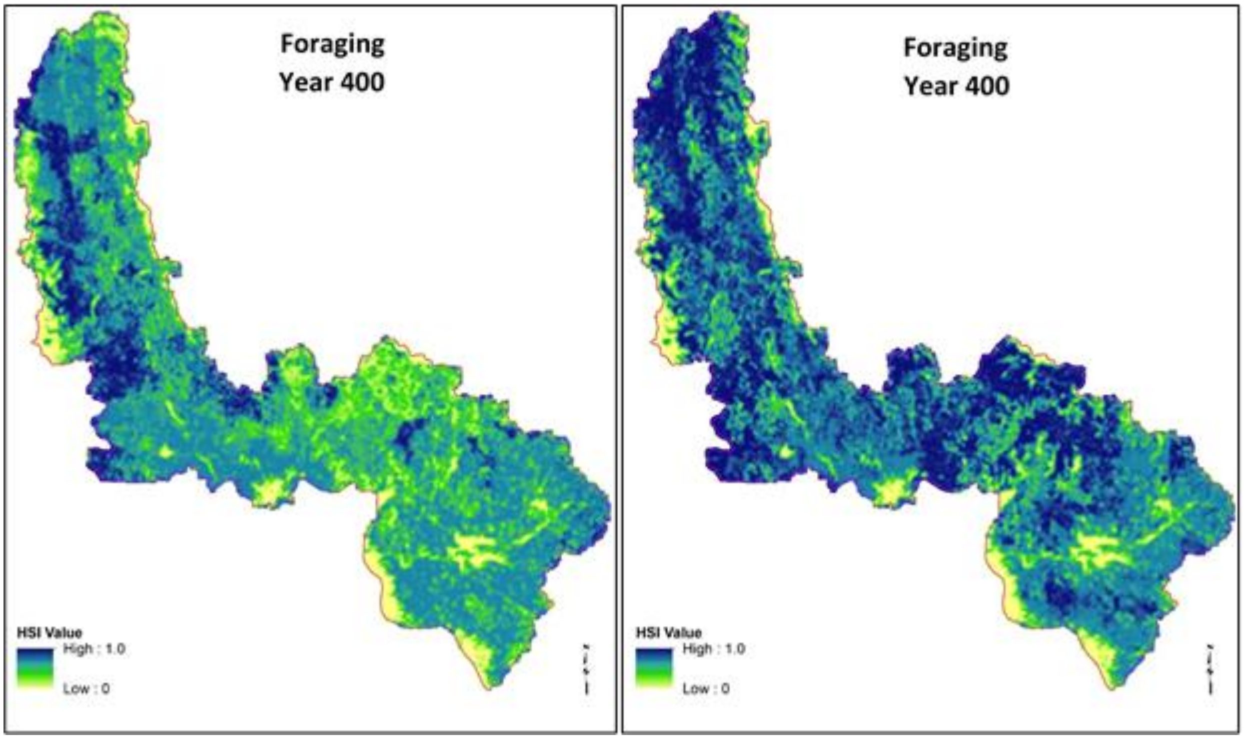


Figure 47. Maps of estimated habitat suitable for northern goshawk foraging areas showing outputs from two time steps produced through SIMPPLLE modeling of historical ecosystem conditions for the BSLRP area.

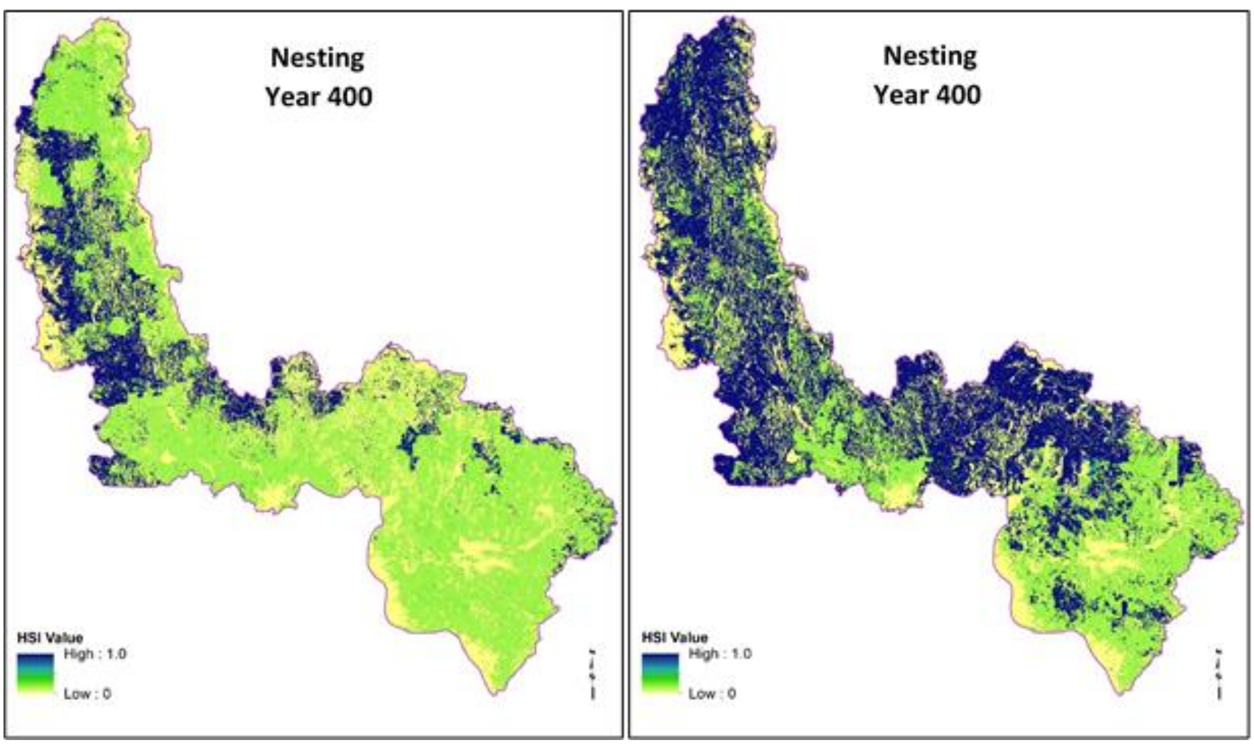


Figure 48. Figure 41. Maps of estimated habitat suitable for northern goshawk nesting areas showing outputs from two time steps produced through SIMPPLLE modeling of historical ecosystem conditions for the BSLRP area.

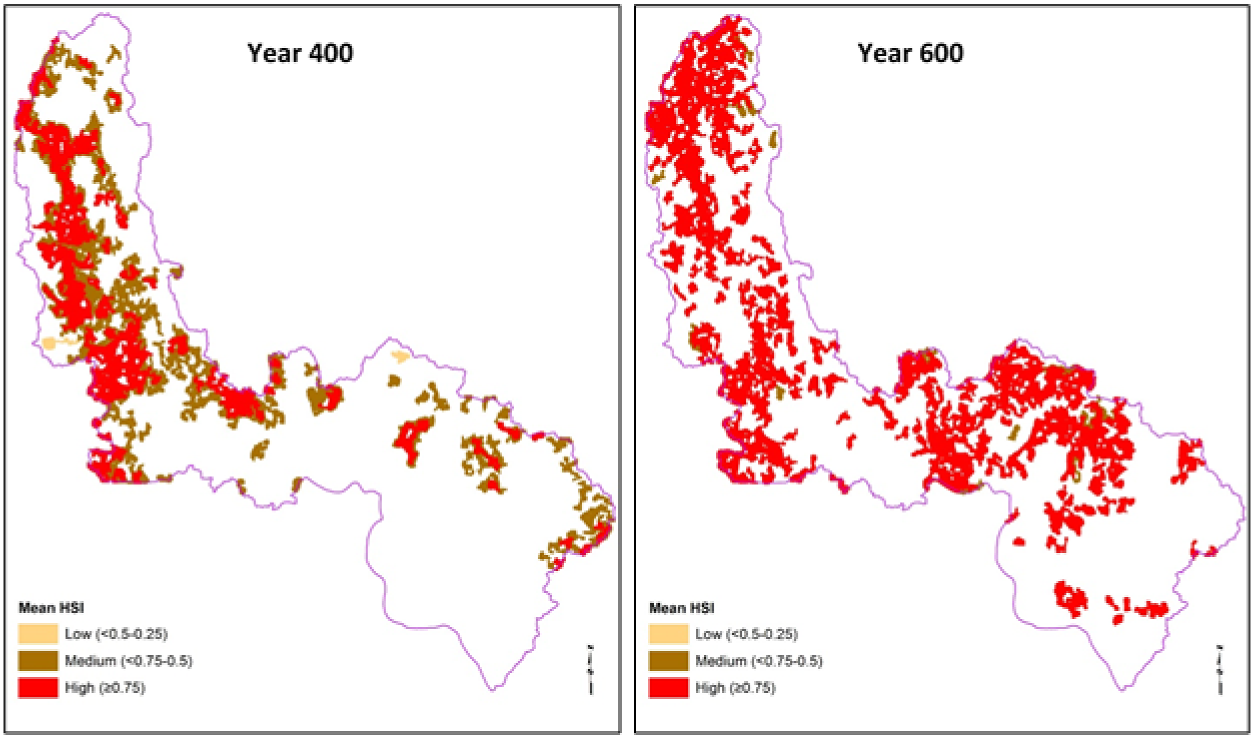


Figure 49. Maps of estimated potential home ranges of high, medium, and low quality for northern goshawk produced from habitat suitability maps from outputs of two time steps of SIMPPLLE modeling for historical ecosystem conditions for the BSLRP area.

Table 33. Estimated numbers of potential home ranges of high, moderate and low quality for northern goshawk from 5 time steps of SIMPPLLE historical modeling of ecosystem diversity for the BSLRP area.

Time	Home Range Quality		
Period	**High**	**Moderate**	**Low**
200	86	615	5
400	190	160	2
600	515	14	0
800	403	113	2
1000	168	244	7
Mean	**272**	**229**	**3**
CI (95%)	**132-412**	**48-410**	**3-5**

Current

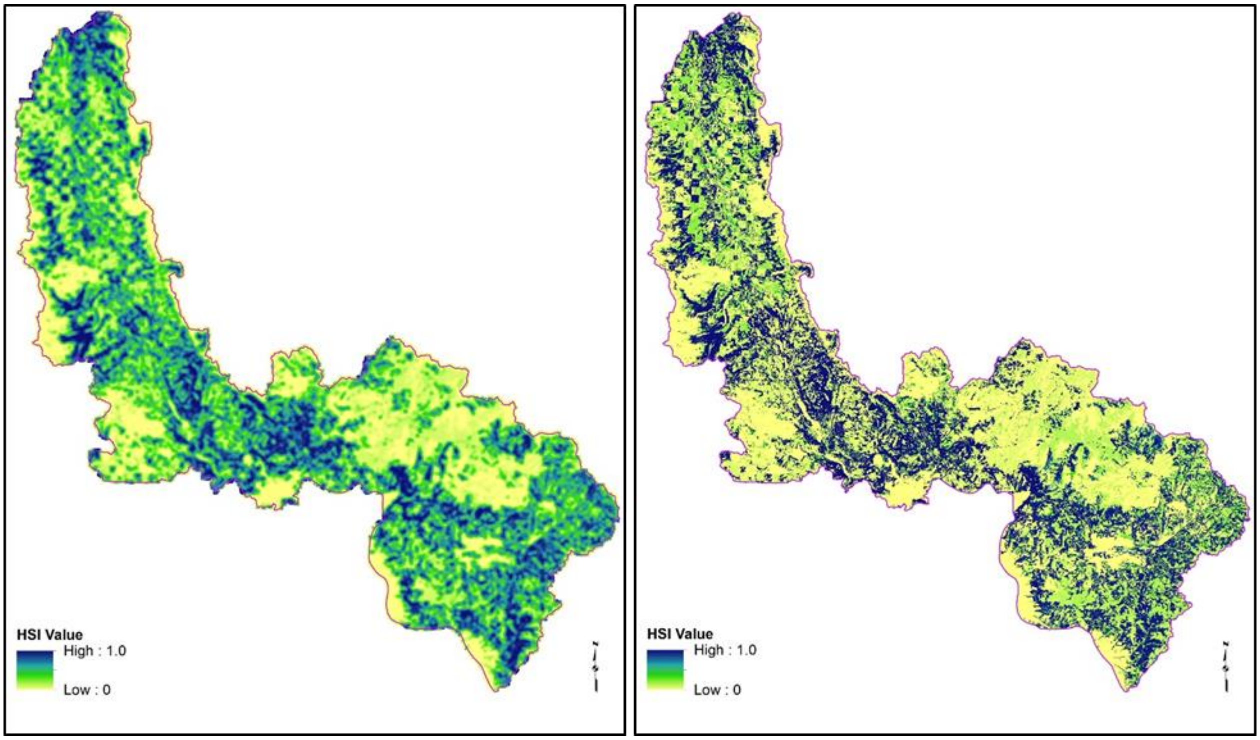


Figure 50. Current modeled habitat suitability for foraging and nesting for northern goshawk for the BSLRP project area based on VMAP mapping of ecosystem diversity.

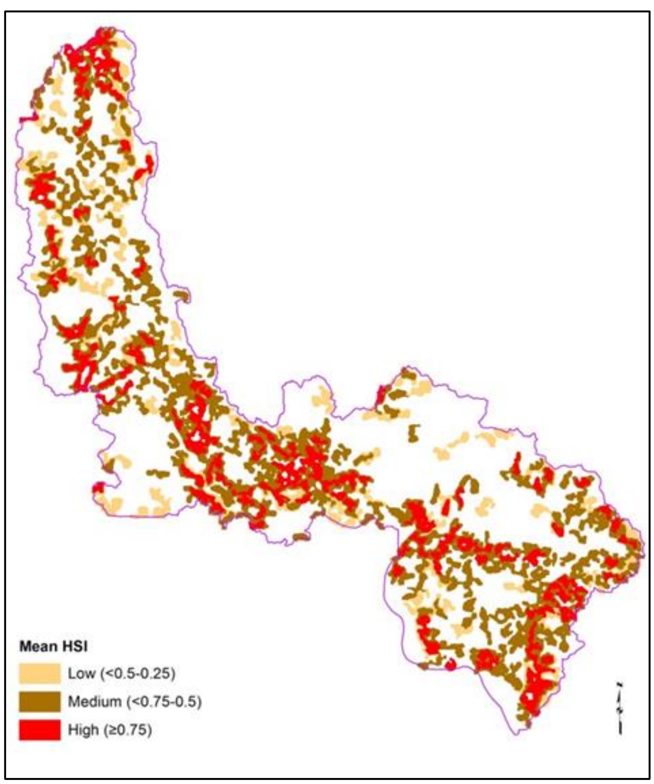


Figure 51. Current estimated potential home ranges of high, medium, or low quality for northern goshawk for the BSLRP project area based on VMAP mapping of ecosystem diversity.

Table 34. Estimated numbers of home ranges of varying quality for northern goshawk from historical modeling of ecosystem diversity compared to current conditions from VMAP mapping of ecosystem diversity for the BSLRP area.

		Home Range Quality		
Time Period		**High**	**Moderate**	**Low**
Historical	**Mean**	272	229	3
	CI (95%)	132-412	48-410	3-5
Today	**Mean**	166	281	37

Black-backed Woodpecker

Historical

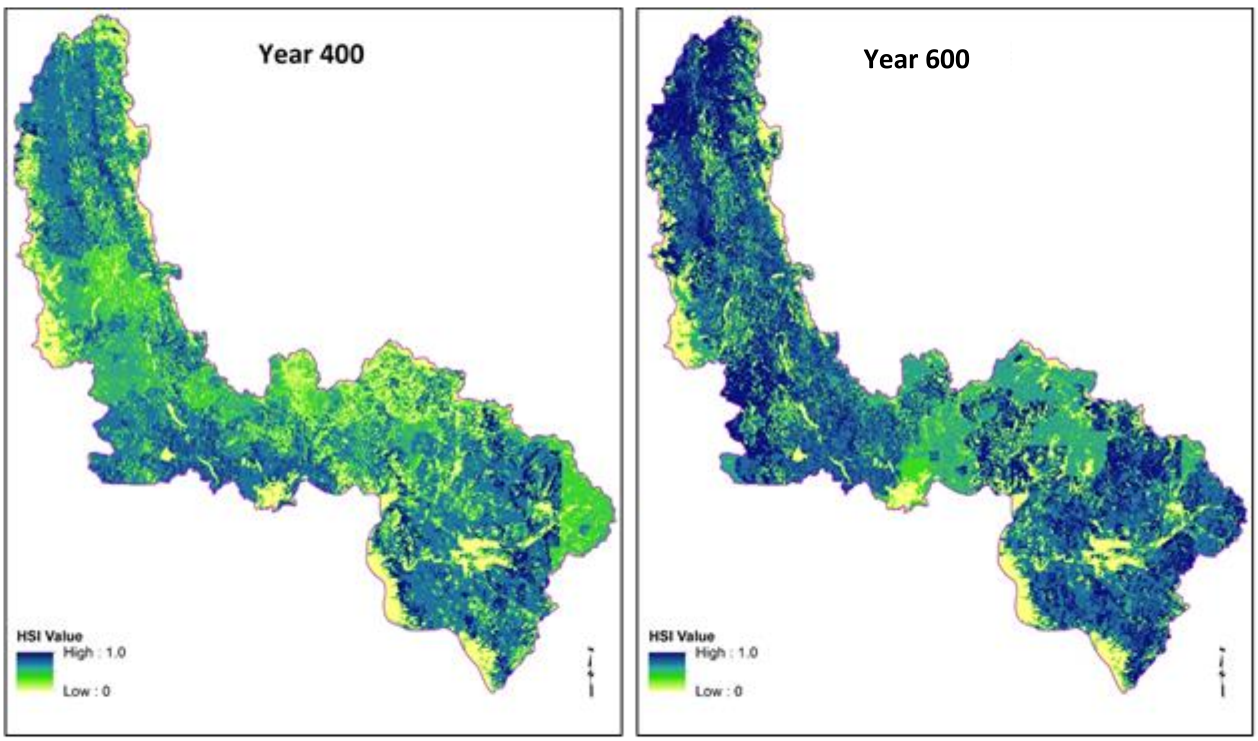


Figure 52. Maps of estimated habitat suitable for black-backed woodpeckers showing outputs from two time steps produced through SIMPPLLE modeling of historical ecosystem conditions for the BSLRP area.

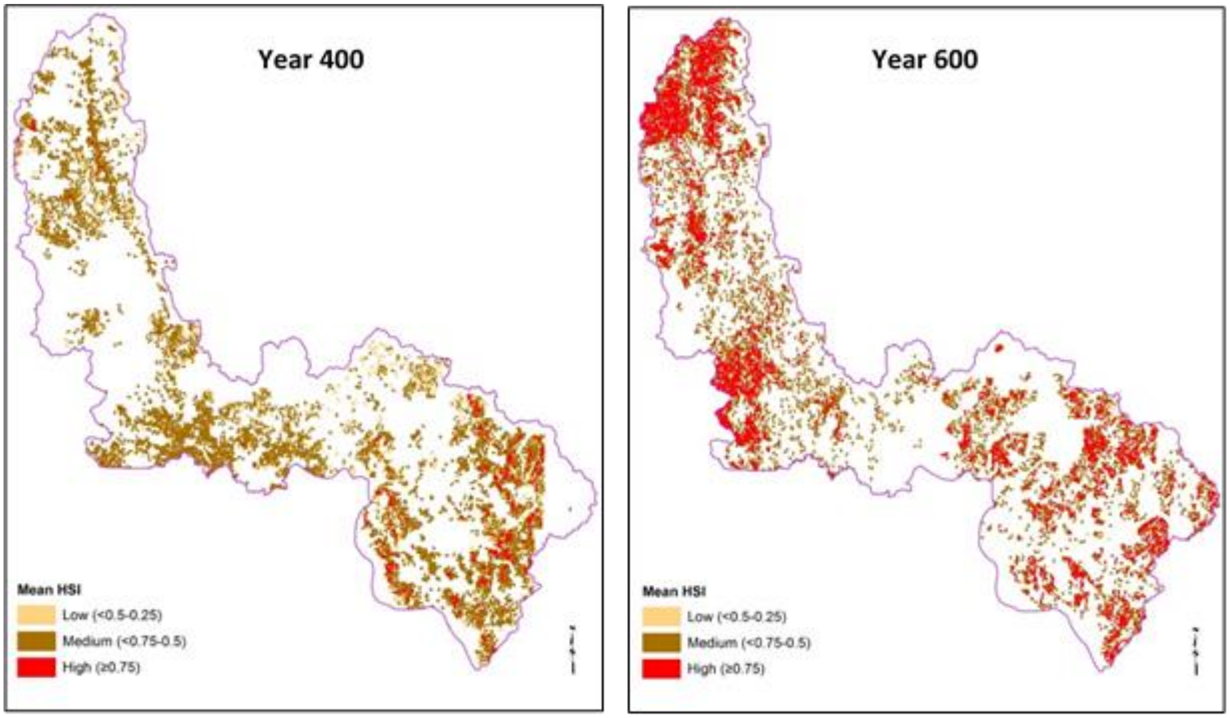


Figure 53. Maps of estimated potential home ranges of high, medium, and low quality for black-backed woodpecker produced from habitat suitability maps from outputs of two time steps of SIMPPLLE modeling for historical ecosystem conditions for the BSLRP area.

Table 35. Estimated numbers of potential home ranges of high, moderate and low quality for black-backed woodpecker from 5 time steps of SIMPPLLE historical modeling of ecosystem diversity for the BSLRP area.

Time	Home Range Quality		
Period	High	Moderate	Low
200	363	3986	522
400	306	3377	501
600	2431	2663	109
800	0	1393	463
1000	361	3989	502
Mean	**692**	**3082**	**419**
CI (95%)	**0-1463**	**2227-3937**	**282-556**

Current

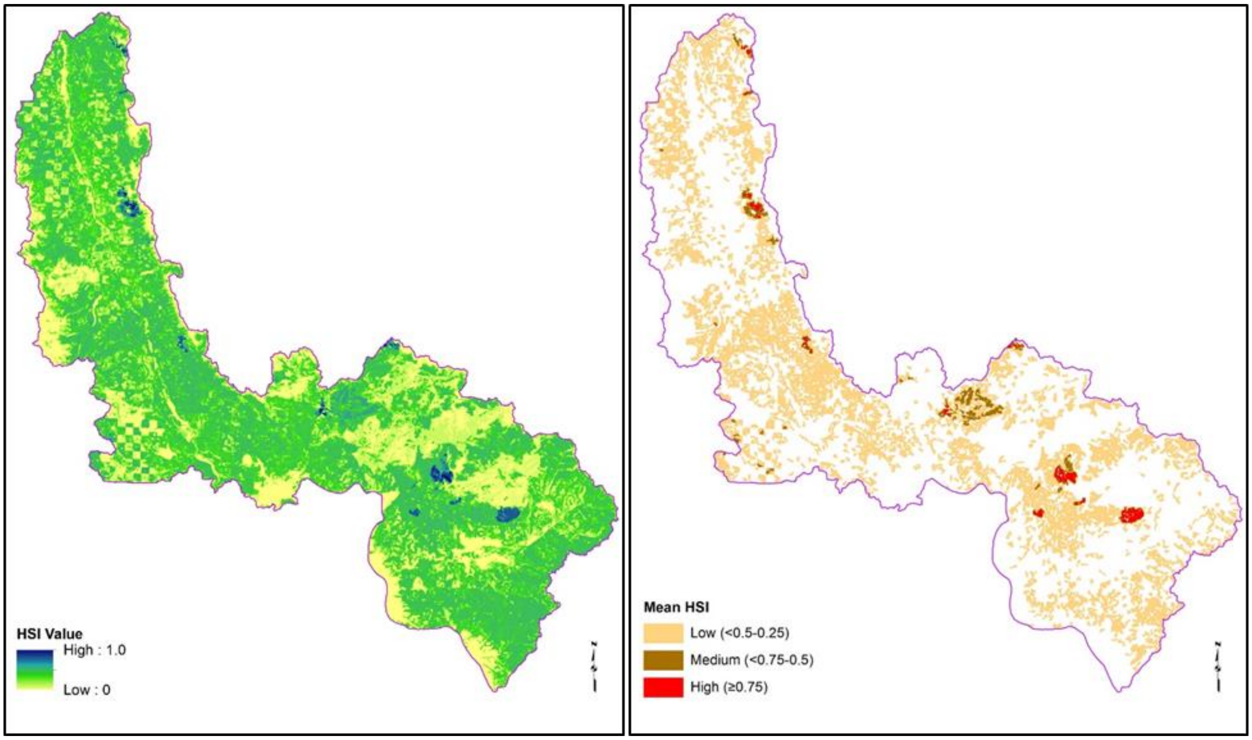


Figure 54. Current modeled habitat suitability and estimated potential home ranges of high, medium, or low quality for black-backed woodpecker for the BSLRP project area based on VMAP mapping of ecosystem diversity.

Table 36. Estimated numbers of home ranges of varying quality for black-backed woodpecker from historical modeling of ecosystem diversity compared to current conditions from VMAP mapping of ecosystem diversity for the BSLRP area.

		Home Range Quality		
Time Period		**High**	**Moderate**	**Low**
Historical	**Mean**	692	3082	419
	CI (95%)	0-1463	2227-3937	282-556
Today	**Mean**	99	177	1439

Pileated Woodpecker

Historical

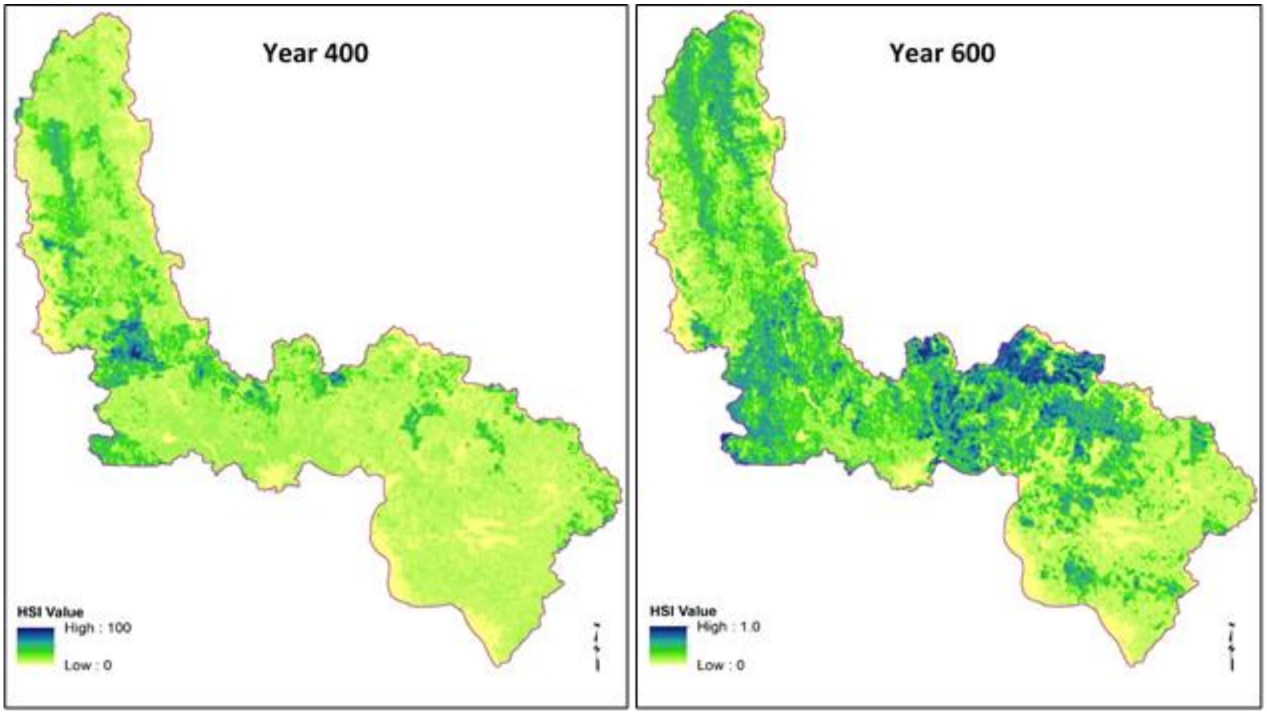


Figure 55. Maps of estimated habitat suitable for pileated woodpecker showing outputs from two time steps produced through SIMPPLLE modeling of historical ecosystem conditions for the BSLRP area.

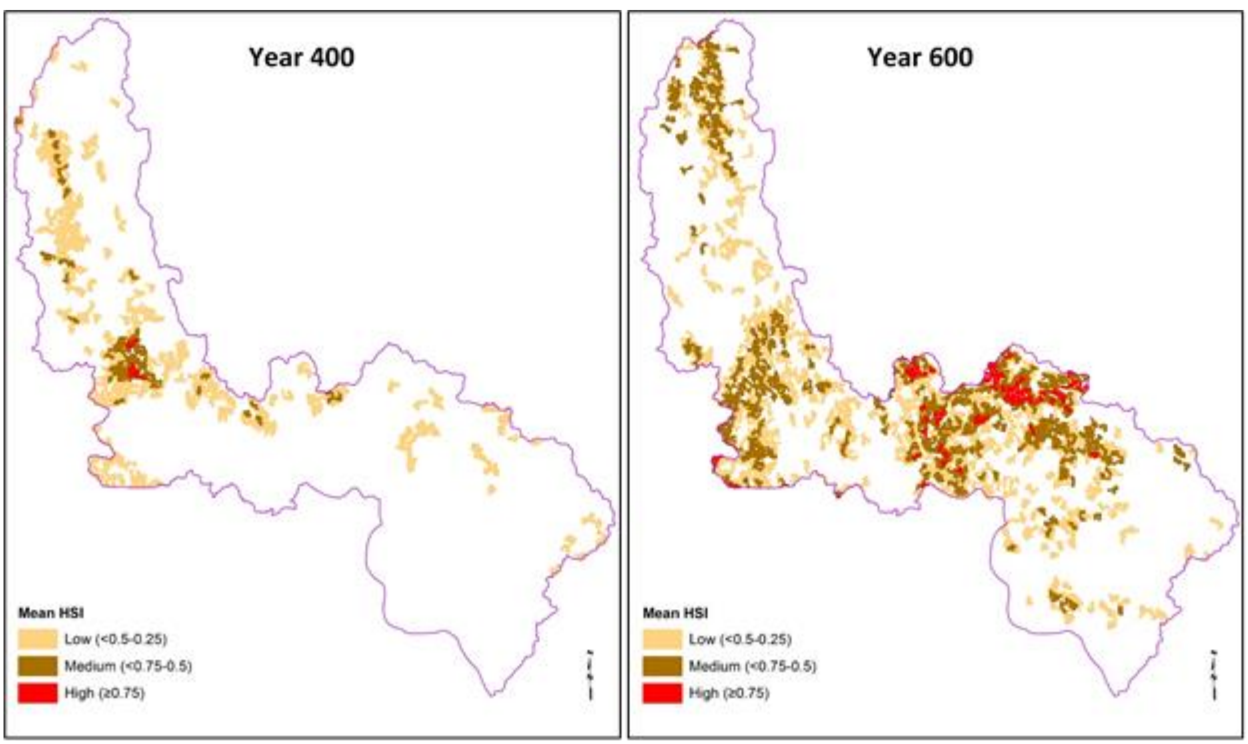


Figure 56. Maps of estimated potential home ranges of high, medium, and low quality for pileated woodpecker produced from habitat suitability maps from outputs of two time steps of SIMPPLLE modeling for historical ecosystem conditions for the BSLRP area.

Table 37. Estimated numbers of potential home ranges of high, moderate and low quality for pileated woodpecker from 5 time steps of SIMPPLLE historical modeling of ecosystem diversity for the BSLRP area.

Time	Home Range Quality		
Period	High	Moderate	Low
200	7	47	55
400	6	52	213
600	63	366	368
800	23	237	469
1000	13	52	185
Mean	**22**	**151**	**258**
CI (95%)	**3-41**	**37-264**	**145-385**

Current

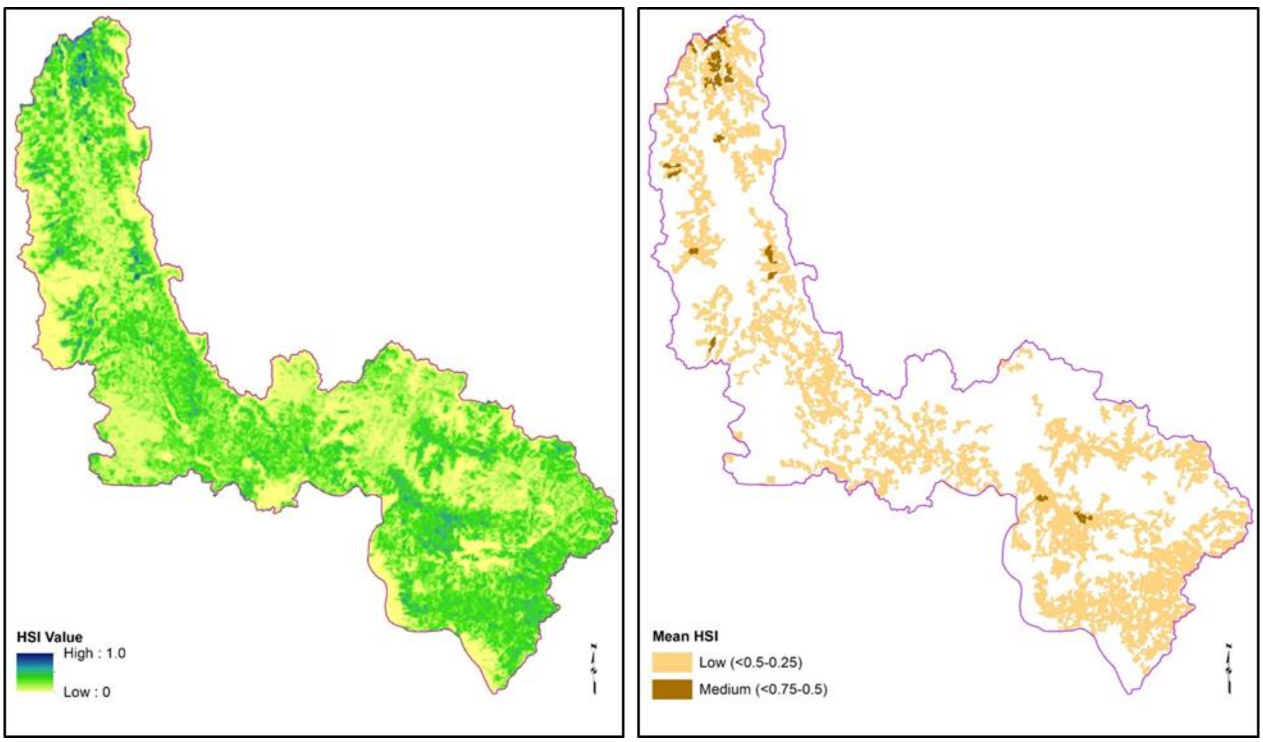


Figure 57. Current modeled habitat suitability and estimated potential home ranges of high, medium, or low quality for pileated woodpecker for the BSLRP project area based on VMAP mapping of ecosystem diversity.

Table 38. Estimated numbers of home ranges of varying quality for pileated woodpecker from historical modeling of ecosystem diversity compared to current conditions from VMAP mapping of ecosystem diversity for the BSLRP area.

		Home Range Quality		
Time Period		High	Moderate	Low
Historical	**Mean**	22	151	258
	CI (95%)	3-41	37-264	145-385
Today	**Mean**	0	25	41

Cumulative Change Discussion

The habitat and home range quality assessments for the 5 selected species showed some interesting results. Habitat suitability modeling has been used since the 1980's as a means of synthesizing the results of studies on the habitat requirements of a species, and using this knowledge to make informed estimates of habitat quality comparisons. The home range analysis takes habitat suitability outputs and interprets them at scales applicable to the species. This is particularly important for species with larger home ranges who use a diversity of ecosystems and thus more variable habitat qualities. The home range analysis uses allometric home range sizes as a starting point, recognizing that these are theoretical minimum sizes of areas that could, in the most optimal settings, support the species, but that in reality may never be achieved. We used 5X the allometric home range size in assessing potential home range qualities to produce more realistic targets of optimum home range sizes. This may still underestimate the actual minimum home range size that could be supported in this landscape, but it provides a scientifically formulated and consistent methodology for evaluating habitat quality over time and for comparisons of historical to current conditions.

Species results were varied in terms of habitat quality. Fisher (Table 29) had very low numbers of potential high quality (mean of 4, 95% confidence interval (CI) of 1-7) and low numbers of potential moderate quality (mean of 19, with CI of 11-28) home ranges as estimated from modeling of historical ecosystem diversity. So, while this landscape historically supported some fisher habitat, it had low inherent capacity to support fisher populations. Fisher were likely to have been consistently present in low numbers, but would have had a relatively low probability of viable populations based on the habitat quality in this landscape alone. Populations in the project area likely received demographic support from populations in nearby landscapes that had higher inherent capability to support viable populations, such as reported in northern Idaho. This means population goals for fisher in the BSLRP area should recognize the limitiations of the landscape and not plan for habitat quality beyond what occurred historically. High expectations for long-term viability would be unrealistic and would likely compromise other biodiversity objectives. Current estimates of fisher habitat quality (Table 30) showed slightly lower qualities than estimated to have occurred historically. High quality home ranges do not occur but some moderate quality home ranges (32) were estimated to be present. Restoration of priority ecosystems discussed previously, particularly the large to very large tree size classes with moderate to closed canopies in the higher elevation and moister ecological sites, will provide higher quality habitat conditions for this species more consistent with historical amounts.

Flammulated owls had very consistent historical habitat quality (Table 31), having a 95% confidence interval of 146-154 for high quality home ranges with similar low variability in amounts of moderate quality home ranges. This is not surprising, as the low elevation, warmer and drier ecological sites used by this species were historically maintained in more consistent conditions by frequent low to moderate severity fires. This historical analysis reveals the BSLRP area had good inherent capability for flammulated owls and would be expected to have supported good populations with reasonable probabilities of persistence. Estimates of current conditions (Table 32) reveal an approximate 2/3 reduction in habitat capability for this species. This reduction is easily explained by the loss of very large trees in the lower elevation ecological sites, particularly reductions in large ponderosa pine from early logging. Restoration of open canopy, very large tree ecosystems that have been prioritized for the warmer and drier ecological sites will significantly enhance habitat quality for flammulated owls.

Northern goshawks showed some variability over time in numbers of different quality home ranges (Table 33), but overall had good amounts of potential habitat. It was interesting that at times when high quality home ranges were lower, more moderate quality home ranges were mapped and vice versa. Overall, the analysis showed the BSLRP area supports good habitat for northern goshawks, and the population of this species would be expected to have high probabilities of persistence indicating the landscape's high inherent capability for this species. Current habitat conditions (Table 34) showed a slightly lower number of high quality home ranges than the historical mean, but still well within the historical confidence intervals for both high quality and moderate quality home ranges. In general, these results indicate this species had good habitat conditions in this landscape both historically and under current conditions. The priority restoration goals for ecosystem diversity should maintain this species habitat quality in the landscape.

Black-backed woodpeckers showed the greatest variability in historical habitat quality (Table 35). High quality home ranges varied from 0 to 2431 with a mean of 692. Moderate quality home ranges showed less variation, and maintained good amounts throughout the 1000 year historical analysis. The variation in the habitat for this species is not surprising. High quality habitat is provided by high amounts of high severity fire. The historical analysis (Figures 21-23) showed how amounts of the different fire severities varied over time with associated variations in climate conditions. In decades with large amounts of high severity fire, there was an abundance of high quality black-backed woodpecker habitat. In periods following cooler and moister conditions, low amounts of high severity fire occurred, reducing the high quality habitat for this species. However, while there was some variation in the amounts of moderate quality habitat, there was always generally good amounts of this habitat available. This was provided by older high severity burns as well as stands with high tree mortality from disturbances such as insect infestations. While this landscape had good inherent capability for providing black-backed woodpecker habitat, populations would be expected to show substantial fluctuations depending on climate patterns and resulting amounts of high severity fire. Current habitat conditions (Table 36) estimated 99 high quality home ranges, which was within the historical confidence interval but on the low end. A low number of moderate quality home ranges (less than the historical confidence interval) was estimated. This is surprising given the amounts of insect infestations currently present in the BLSRP landscape. Our use of the VMAP mapping of ecosystem diversity may not have captured some of these more recent changes to current conditions. It is very likely that the future of the landscape will include significant amounts of high severity fire, given current ecosystem conditions and projected effects of climate change on fire regimes. High severity fire has been an important disturbance in this landscape, although in smaller amounts than would occur today without management intervention. High severity fire should continue to play a role in the BSLRP landscape but restoration actions should target locations where high severity fire was an historically important process, while reducing its occurrence on sites that were only very rarely influenced by this process historically, or where human settlement occurs today.

Pileated woodpeckers showed relatively low quality habitat both historically (Table 37) and under current conditions (Table 38). Historical estimates of a mean of 22 high quality and 151 moderate quality home ranges seem low for this species, as do current conditions of 0 high quality and 25 moderate quality home ranges. The substantial reduction in very large trees explains the lower quality habitat conditions occurring today compared to historical. More open canopy cover conditions historically could have reduced amounts of high quality habitat for this species in some lower elevation and dryer ecological sites. The model outputs show that the landscape had inherent capability to support this species over time, but did not support high numbers of high quality home ranges. Restoration of ecosystem diversity, particularly increasing the very large tree component of the landscape will improve habitat for this species, even with some reduction in canopy cover. As with fisher, identifying areas where large to very large tree ecosystems are desired particularly in cooler and moister ecological sites will improve habitat quality for pileated woodpeckers.

With the exception of fisher and to a lesser extent pileated woodpecker, the BSLRP landscape had good to excellent inherent capability to support the modeled species. Fisher habitat was present, but historical conditions may not have provided a high probability of viability. Pileated woodpeckers would have had a reasonable probability of persistence, but may not have been in high numbers. Flammulated owls would have had good habitat conditions in limited portions of the landscape. Black-backed woodpeckers would have had persistent populations but shown considerable variability in numbers in response to climate conditions that temporally increased amounts of high severity fire. Northern goshawk habitat appeared to have consistently good amounts. Restoration of priority ecosystems will enhance the habitat quality of fisher, flammulated owls, northern goshawks, and pileated woodpeckers. Planning should include consideration of where high severity fire will be allowed to occur, and with this provision, habitat will be provided for black-backed woodpeckers.

Key Findings and Recommendations

This landscape assessment is based on the conservation strategy for ecological sustainability as described in the Forest Planning Rule and the Ecological Restoration Policy of the Forest Service. The scientific foundation of this strategy is based on maintaining the diversity of ecosystems occurring historically and that by ensuring adequate representation of these ecosystems, the habitat requirements of all species will be provided where consistent with the inherent capability of each landscape. The Ecosystem Restoration Policy emphasizes restoration should be evaluated within the context of the historical or natural range of variability (NRV), in support of the scientific foundation of the strategy. While adjustments to historical conditions are expected due to current social and economic considerations as well as the challenges of climate change, to maintain the integrity of the conservation strategy, ecosystem restoration founded on a thorough understanding of historical conditions is a primary requirement. The Ecosystem Restoration Policy acknowledges it may not be feasible to restore all ecosystems when it stated: “when an ecosystem has been so degraded such that it is impossible or impractical to return conditions to those within the NRV, then functional restoration may be appropriate to restore ecological processes but achieve the essential functions of the ecosystem with different species compositions and structure than pre-European settlement conditions.” However, the intent of the policy is clear - historical conditions are the restoration goal - while recognizing forest management must also incorporate social and economic considerations.

This landscape assessment has classified and quantified the ecosystem diversity for the BSLRP landscape at both landscape and ecosystem levels. Further, it has provided estimates of historical ecosystem diversity compared to current conditions, and quantified the cumulative changes that have occurred to this diversity due to direct and indirect effects from modern settlement. The cumulative change analysis highlights those specific ecosystems having low levels of representation today compared to historical amounts and those that should receive high priority for restoration. The assessment has also provided the data needed to describe and quantify the reference conditions to use as stand level goals for restoration of specific ecosystems in terms of vegetation compositions, structures, and disturbance processes.

Identified restoration priorities include increasing the amounts of open and moderate canopy cover conditions and increasing the percentage of the landscape containing very large trees. Ecosystems with the greatest departure from historical conditions occur in some of the drier ecological sites, specifically the moderately warm dry type in the M332B-West and M332B-East ecoregions. In addition, whitebark pine ecosystems in the higher elevation cold and timberline ecological sites are a priority for restoration because of the combined effects of fire exclusion, infestation by the exotic white pine blister rust, and recent increases in effects of pine beetles attributed to climate change. In the M333C ecoregion, in addition to the need to restore very large tree ecosystems in dry ecological sites, some of the moister sites also have ecosystems that are at very low representation levels. The moderately warm moist ecological site had 5%, 10%, and 4% representation of the open, moderate, and closed canopy very large tree ecosystems respectively, while the moderately cool-moist ecological site had 3%, 6%, and 6% representation of historical amounts of these same disturbance states. As with the other ecoregions, high elevation whitebark pine is a restoration priority, as well as western white pine ecosystems that have experienced nearly complete loss as functional ecosystems due to past logging, fire exclusions, and especially the influence of white pine blister rust.

Fire regimes were also assessed and quantified for historical and current conditions. This analysis demonstrated the important role of mixed severity fire regimes in the BSLRP landscape, and also quantified how the amounts of all 4 fire regimes have shifted from historical to current conditions. These shifts were shown to be greater in ecoregions M332B-East and West, as drier ecological sites are a greater proportion of these areas and historically had more low to moderate severity fire.

This assessment did not model historical fire sizes. Because the SIMPPLLE model uses decadal time steps, the size of individual fires may be inflated by the 10 year time increments. As an alternative, managers designing restoration treatments should look at the existing vegetation patterns in targeted areas, and use interpretation of the known disturbance processes to guide the determination of patterns for restoration (Hessburg et al. 2016). Residual very large trees, especially larch and ponderosa pine, provide good evidence of where low to moderate severity fire occurred historically, creating the stand conditions described in reference conditions for these ecosystems from the empirical and observational historical data. In many other locations, residual large stumps are indicators of where very large tree stands occurred, particularly on drier ecological sites. In contrast, patches of 100+ year old stands of lodgepole pine are a good indicator of where high severity fire occurred historically, and where early successional conditions may be warranted for restoration.

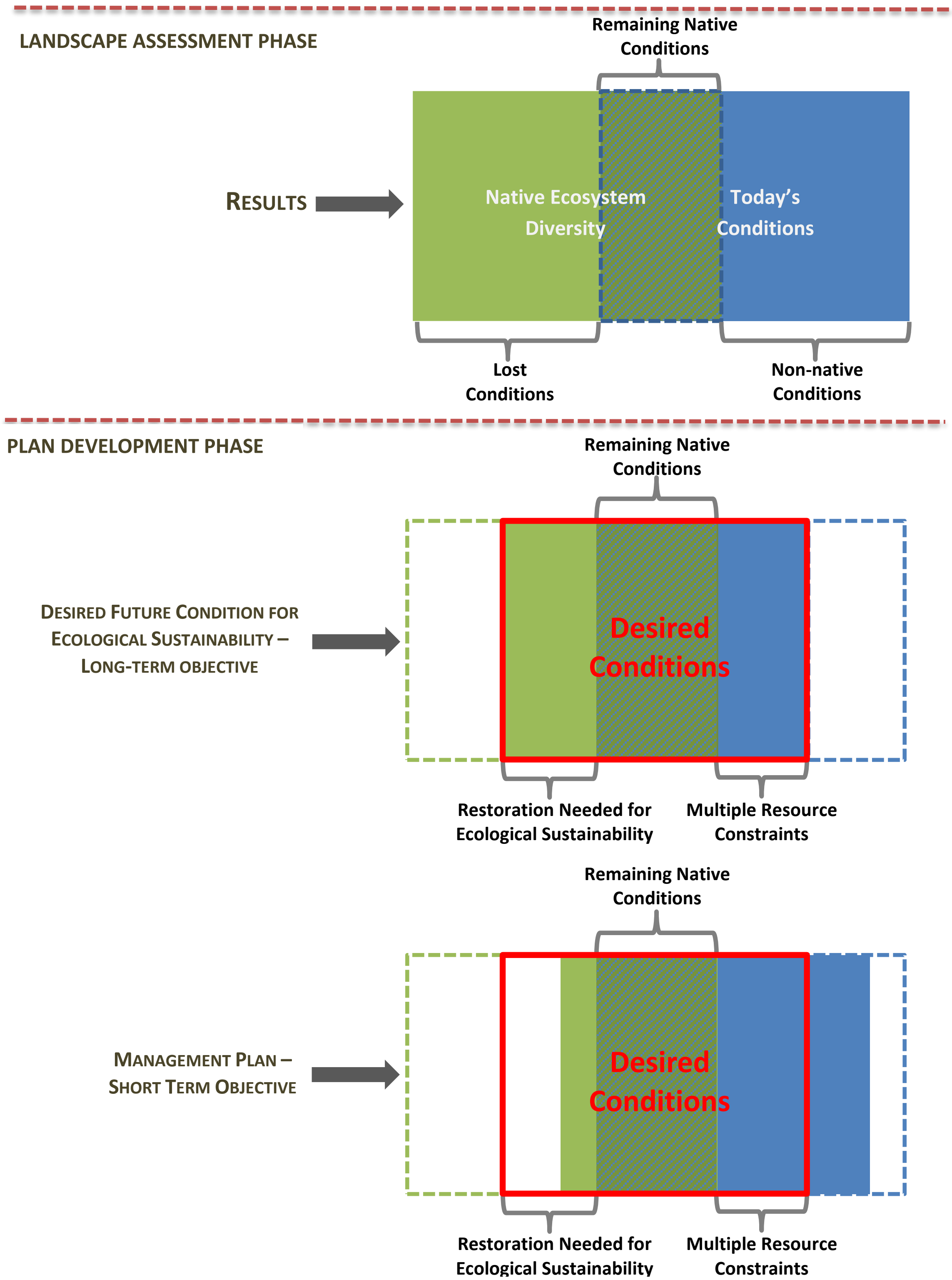
Of the 5 wildlife species assessed in relation to historical ecosystem diversity, the landscape was found to have low inherent capability of fisher habitat. While fisher habitat was present, populations of this species were likely to have been low, and would have had a low probability of longterm persistence without potential for demographic support from neighboring ecoregions. The landscape did not appear to support large amounts of high quality pileated woodpecker habitat, but would have been expected to have the inherent capability to maintain habitat for this species. Flammulated owls had good inherent habitat quality in this landscape but were limited in their distributions, while northern goshawks showed some variation in amounts of habitat but had good numbers, well distributed throughout the landscape. Black-backed woodpeckers showed the greatest variability in habitat quality in response to the amount of high severity fire, but displayed good habitat persistence within the landscape. Restoration of priority ecosystems should enhance the status of all of these species within the project area, with the exception of black-backed woodpeckers which will require a fire management strategy that allows high severity fires to occur in appropriate locations.

Restoration planning should also consider the likely influences of climate change to ensure that restoration of ecosystems will be sustainable and resilient under predicted future conditions (Saxon 2003). For example, a specific native ecosystem may be identified as in need of restoration within a landscape. However, if this ecosystem, when evaluated against downscaled climate change predictions, is found to be unsustainable into the future, then the future desired species compositions may need to be adjusted through functional restoration to make the ecosystem more resilient under predicted future climate conditions, but kept as functionally similar to the historical plant community as possible.

Some authors (e.g., Hanberry et al. 2015, Dumroese et al. 2015) have suggested restoration should look to the future and plan to produce ecosystems that will be resilient under future predicted conditions. There is certainly merit in considering the future sustainability of all ecosystems in a landscape. However, what some of these discussions fail to capture is the importance of maintaining adequate representation of historical ecosystems as the scientific foundation of the biodiversity conservation strategy. Restoration is far more than having forest stands that will persist or be reslilient into the future; restoration must attempt to provide for the biodiverisity of the landscape, which by definition in North America is all of the ecosystems, species, and genetic diversity that occurred in a landscape prior to major alterations by modern anthropogenic activities. While it would not make sense to try to restore or maintain ecosystem conditions that will have low likelihood of resilience or persistence under future predicted climate conditions, the importance of such ecosystems in the landscape should still be recognized. If a specific ecosystem cannot be sustained or restored, then functional restoration objectives should become the target instead, by developing conditions as close to historical as possible but that will be sustainable into the future. As Hart et al. (2015) noted: “if forest restoration is no longer primarily concerned with the recovery of ecosystem conditions within the HRV, the term “restoration” is a misnomer. Incorporating resiliency in management goals is wise forest stewardship, but labeling this objective as restoration can lead to a disconnect between restoration scientists and forest managers.” The Ecosystem Restoration Policy of the U.S. Forest Service makes clear the objectives of ecosystem restoration, and identifies where functional restoration may be appropriate because ecological restoration is not feasible or sustainable.

BSLRP, as with all restoration projects, will have limited budgets and other constraints. To be the most effective in meeting the project objectives, treatments and actions should target those ecosystems with the greatest restoration needs for better representation in the landscape. This assessment has identified the restoration needs at the landscape level for the BSLRP project area, as well as the reference conditions at the ecosystem level to help guide desired conditions for individual stands receiving treatment. Figure 58 depicts how restoration can be envisioned to shift the ecosystem diversity from its current conditions to better represent the historical ecosystem diversity needed to support the biodiversity conservation strategy. While many additional factors will limit the ability and even desirability of restoring historical ecosystem conditions in all locations, the overall restoration goals should emphasize providing key ecosystem conditions that are currently underrepresented in the landscape in order to meet sustainability and biodiversity objectives for the long-term. Deviations from these goals should be identified relative to balancing social and economic needs, or ecological constraints that would interfere with achieving a targeted reference condition.

Figure 58. The decision process used to restore historical native ecosystem diversity to levels needed to support the biodiversity conservation strategy, while also recognizing current constraints and social needs.



LITERATURE CITED

Abella, S. R., W. W. Covington, P. Z. Fulé, L. B. Lentile, A. J. Sánchez Meador, and P. Morgan. 2007. Past, present, and future old growth in frequent-fire conifer forests of the western United States. Ecology and Society 12(2):16.

Abella, S. R., and C. W. Denton. 2009. Spatial variation in reference conditions: historical tree density and pattern on a *Pinus ponderosa* landscape. Canadian Journal of Forest Research 39:2391-2403.

Agee, J. K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, DC.

Agee, J. K. 1998. Fire and pine ecosystems. Cambridge University Press, New York, New York

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

Agee, J. K. 2004. The complex nature of mixed-severity fire regimes.*in* Mixed-severity fire regimes: ecology and management. Association for Fire Ecology and Washington State University Extension, Spokane, WA.

Alexander, M. E., and F. G. Hawksworth. 1976. Fire and dwarf mistletoes in North American coniferous forests. Journal of Forestry 74:446-449.

Alexander, R. R., R. C. Shearer, and W. D. Shepperd. 1990. Subalpine fir. Pages 60-70 *in* R. M. Burns and B. H. Honkala, editors. Silvics of North America. USDA Agricultural Handbook.

Alig, R. J. 2007. A United States view on changes in land use and land values affecting sustainable forest management. Journal of Sustainable Forestry 24:209-227.

Amoroso, M. M., L. D. Daniels, M. Bataineh, and D. W. Andison. 2011. Evidence of mixed-severity fires in the foothills of the Rocky Mountains of west-central Alberta, Canada. Forest Ecology and Management 262:2240-2249.

Anderson, L., C. E. Carlson, and R. H. Wakimoto. 1987. Forest fire frequency and western spruce budworm outbreaks in western Montana. Forest Ecology and Management 22:251-260.

Andison, D. W. 2012. The influence of wildfire boundary delineation on our understanding of burning patterns in the Alberta foothills. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 42:1253-1263.

Antos, J. A., and J. R. Habeck. 1981. Successional development in Abies grandis (Dougl.) forbes forests in the Swan Valley, Western Montana. Northwest Science 55:26-39.

Aplet, G. H., and W. S. Keeton. 1999. Application of historical range of variability concepts to biodiversity conservation. Pages 71-86 *in* R. K. Baydack, H. Campa, and J. B. Haufler, editors. Practical approaches to the conservation of biological diversity. Island Press, Washington, D.C.

Arno, S. F. 1980. Forest fire history in the Northern Rockies. Journal of Forestry 78:460-465.

Arno, S. F., and R. J. Hoff. 1989. Silvics of whitebark pine (*Pinus albicaulis*). USDA Forest Service, General Technical Report INT-GTR-253. Ogden, UT. 13pp.

Arno, S. F., D. J. Parsons, and R. E. Keane. 2000. Mixed-severity fire regimes in the Northern Rocky Mountains: consequences of fire exclusion and options for the future.*in* D. N. Cole, S. F. McCool, W. T. Borrie, and J. O'Loughlin, editors. Wilderness science in a time of climate change - Volume 5: Wilderness, ecosystems, threats, and management; 1999 May 23-27. USDA Forest Service, Missoula, MT.

Arno, S. F., J. H. Scott, and M. G. Hartwell. 1995. Age-class structure of old growth ponderosa pine/Douglas-fir stands and its relationship to fire history. USDA Forest Service Report INT-RP-481, Ogden, UT.

Arno, S. F., D. G. Simmerman, and R. E. Keane. 1985. Forest succession of four habitat types in western Montana. USDA Forest Service, General Technical Report INT-GTR-177. Ogden, UT. 74 pp.

Arno, S. F., H. Y. Smith, and M. A. Krebs. 1997. Old growth ponderosa pine and western larch stand structures: influences of pre-1900 fires and fire exclusion. USDA Forest Service Report, INT-RP-495, Ogden, UT.

Ayres, H. B. 1900. The Lewis and Clark Forest Reserve, Montana. Part V, Forest Reserves Edition. The 21st Annual Report of the Survey, 1899-1900, Department of the Interior - US Geological Society. Government Printing Office. Washington, DC.

Baker, W. L. 2015. Are high-severity fires burning at much higher rates recently than historically in dry-forest landscapes of the western USA? Plos One 10:26.

Baker, W. L., and D. Ehle. 2001. Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. Canadian Journal of Forest Research 31:1205-1226.

Barrett, S. and J.Jones. 2001. Fire regimes database for Region 1 USDA Forest Service (unpublished).

Barrett, S. 2012. Fire regimes assessment for the Dalton Analysis Area. Unpublished report produced for Lincoln Ranger District, Helena National Forest. 43pp.

Barrett, S. 2013. Fire regimes assessment: Stemple-Flesher Analysis Area. Unpublished report produced for Lincoln Ranger District, Helena National Forest. 49 pp.

Barrett, S., T. W. Swetnam, and W. L. Baker. 2005. Indian fire use: deflating the legend. Fire Management Today 65:31-34.

Barrett, S. W. 1981. Indian fires in the pre-settlement forests of western Montana. Pages 35-41 in M.A. Stokes and J.H. Dieterich, eds, Proceedings of a Fire History Workshop, October 20-24, 1980, Tucson, Arizona. USDA Forest Service, General Technical Report RM-GTR-035.

Barrett, S. W. 2002. Fire history and fire regimes - Upper Swan Valley, Montana. Unpublished report prepared for Swan Ecosystem Center. Condon, MT.

Barrett, S. W., and S. F. Arno. 1982. Indian fires as an ecological influence in the Northern Rockies. Journal of Forestry 80:647-651.

Barrett, S. W., S. F. Arno, and C. H. Key. 1991. Fire regimes of western larch - lodgepole pine forests in Glacier National Park, Montana. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 21:1711-1720.

Barrett, S. W., S. F. Arno, and J. P. Menakis. 1997. Fire episodes in the inland northwest (1540-1940) based on fire history data. USDA Forest Service, General Technical Report INT-GTR-370. Ogden, UT.

Bassman, J. H., B. Gillespie, and C. Fisher. 2015. Disturbance factors driving ecological pattern and process in the Blackfoot Swan Landscape Restoration Project Area: historical conditions and regional trends. USDA Forest Service, Region 1 - unpublished draft report

Bassman, J. H., J. D. Johnson, L. Fins, and J. P. Dobrowolski. 2003. Rocky Mountain ecosystems: diversity, complexity and interactions. Tree Physiology 23:1081-1089.

Bebi, P., D. Kulakowski, and C. Rixen. 2009. Snow avalanche disturbances in forest ecosystems - state of research and implications for management. Forest Ecology and Management 257:1883-1892.

Bebi, P., D. Kulakowski, and T. T. Veblen. 2003. Interactions between fire and spruce beetles in a subalpine Rocky Mountain forest landscape. Ecology 84:362-371.

Beissinger, S. R., E. Nicholson, and H. P. Possingham. 2009. Application of population viability analysis to landscape conservation planning. Pages 33-50 *in* J. J. Millspaugh and F. R. Thompson III, editors. Models for planning wildlife conservation in large landscapes. Academic Press, Burlington, MA.

Beissinger, S. R., and M. I. Westphal. 1998. On the use of demographic models of population viability in endangered species management. Journal of Wildlife Management 62:810-822.

Bestelmeyer, B. T., A. J. Tugel, G. L. Peacock, D. G. Robinett, P. L. Shaver, J. R. Brown, J. E. Herrick, H. Sanchez, and K. M. Havstad. 2009. State-and-transition models for heterogeneous landscapes: a strategy for development and application. Rangeland Ecology & Management 62:1-15.

Bigler, C., D. Kulakowski, and T. T. Veblen. 2005. Multiple disturbance interactions and drought influence fire severity in Rocky Mountain subalpine forests. Ecology 86:3018-3029.

Birch, D. S., P. Morgan, C. A. Kolden, J. T. Abatzoglou, G. K. Dillon, A. T. Hudak, and A. M. S. Smith. 2015. Vegetation, topography and daily weather influenced burn severity in central Idaho and western Montana forests. Ecosphere 6:17.

Blocker, L., S. K. Hagle, R. Lasko, R. E. Keane, B. Bollenbacher, B. Fox, F. Samson, R. Gay, and C. Manning. 2001. Understanding the connection between historic range of variation, current social values, and developing desired conditions. Pages 51-59 *in* S. J. Barras, ed., Proceedings of the National Silvicultural Workshop. USDA Forest Service, Proceedings RMRS-P-19, Ogden, UT.

Brown, R. T., J. K. Agee, and J. F. Franklin. 2004. Forest restoration and fire: principles in the context of place. Conservation Biology 18:903-912.

Brunelle, A., G. E. Rehfeldt, B. Bentz, and A. S. Munson. 2008. Holocene records of Dendroctonus bark beetles in high elevation pine forests of Idaho and Montana, USA. Forest Ecology and Management 255:836-846.

Byler, J., and S. K. Hagle. 2000. Successional functions of pathogens and insects - ecoregional sections M332a and M333d northern Idaho and western Montana. Forest Health Protection Report No. 00-09.

Cansler, C. A., and D. McKenzie. 2014. Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. Ecological Applications 24:1037-1056.

Carlson, C. E., D. G. Fellin, and W. C. Schmidt. 1983. The western spruce budworm in Northern Rocky Mountain forests: a review of the ecology, past insecticdal treatments and silvicultural practices. Pages 76-103 *in* J. O'Loughlin and R. D. Pfister, editors. Proceedings of Management of Second-growth Forests: The State of Knowledge and Research Needs. Montana Forest and Conservation Experiment Station, University of Montana, Missoula, MT.

Carr, C. A. 2007. An evaluation of understory vegetation dynamics, ecosystem resilience and state and transition ecological theory in an Eastern Oregon ponderosa pine forest. PhD Thesis (3295615), Rangeland Ecology and Management. Oregon State University. 260pp.

Chew, J. C., C. M. Stalling, and K. Moeller. 2004. Integrating knowledge for simulating vegetation changes at landscape scales. Western Journal of Applied Forestry 19:102-108.

Cleland, D. T., P. E. Avers, W. H. McNab, M. E. Jensen, R. G. Bailey, T. King, and W. E. Russell. 1997. National hierarchical framework for ecological units. Pages 181-200 *in* M. S. Boyce and A. Haney, editors. Ecosystem management applications for sustainable forest and wildlife resources. Yale University, New Haven, CT.

Clewell, A. F., and J. Aronson. 2013. Ecological restoration: principles, values, and structure of an emerging profession. Island Press, Washington, D.C.

Conklin, D. A., and W. A. Armstrong. 2001. Effects of three prescribed fires on dwarf mistletoe infection in Southwestern ponderosa pine. USDA Forest Service, Southwest Region. Forest Health Report R3-01-02.

Covington, W., R. Everett, R. Steele, L. L. Irwin, T. A. Daer, and A. Auclair. 1994. Historical and anticipated changes in forest ecosystems of the Inland West of the United States. Journal of Sustainable Forestry 2:13-63.

Covington, W., P. Z. Fule, S. C. Hart, and R. P. Weaver. 2001. Modeling ecological restoration effects on ponderosa pine forest structure. Restoration Ecology 9:421-431.

Covington, W. W., P. Z. Fule, M. M. Moore, S. C. Hart, T. E. Kolb, J. N. Mast, S. S. Sackett, and M. R. Wagner. 1997. Restoring ecosystem health in ponderosa pine forests of the southwest. Journal of Forestry 95:23-29.

Daubenmire, R. 1968. Plant communities: a textbook of plant synecology. Harper and Row, New York, NY.

Davis, K. M. 1980. Fire history of a western larch/Douglas-fir forest type in northwestern Montana. Pages 69-74 *in* Fire History Workshop. USDA Forest Service, General Technical Resport RM-GTR-81, Tucson, Arizona.

Dumroese, R. K., B. J. Palik, and J. A. Stanturf. 2015. Forest restoration is forward thinking. Journal of Forestry 113:430-432.

Egan, D., and E. A. Howell. 2001. The historical ecology handbook: a restorationist's guide to reference ecosystems. Island Press, Washington, DC.

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

Fischer, W. C., and A. F. Bradley. 1987. Fire ecology of western Montana forest habitat types. USDA Forest Service General Technical Report, GTR-INT-223. Ogden, UT. 95pp.

Fischer, W. C., and B. D. Clayton. 1983. Fire ecology of Montana forest habitat types east of the Continental Divide. USDA Forest Service, General Technical Report INT-GTR-141. Ogden, UT. 83 pp.

Fitzgerald, S. A. 2004. Fire ecology of ponderosa pine and the rebuilding of fire-resilient ponderosa pine ecosystems. Pages 197-225 *in* Symposium on ponderosa pine: issues, trends, and management; October 18-21, 2004, Klamath, OR. USDA Forest Service, General Technical Report PSW-GTR-198, Portland, OR.

Flint, H. R. 1925. Fire resistance of northern Rocky Mountain conifers. Idaho Forester 7:7-10; 41-43.

Franklin, J. F., R. K. Hagmann, and L. S. Urgenson. 2014. Interactions between societal goals and restoration of dry forest landscapes in western North America. Landscape Ecology 29:1645-1655.

Franklin, J. F., T. A. Spies, R. Van Pelt, A. B. Carey, D. A. Thornburgh, D. R. Berg, D. B. Lindenmayer, M. E. Harmon, W. S. Keeton, D. C. Shaw, K. Bible, and J. Q. Chen. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management 155:399-423.

Freedman, J. D., and J. R. Habeck. 1985. Fire, logging, and white-tailed deer interrelationships in the Swan Valley, Northwestern Montana. Pages 23-35 *in* Proceedings: Fire's Effects on Wildlife Habitat, 1984. USDA Forest Service, General Technical Report INT-GTR-186, Ogden, UT.

Fule, P. Z., W. W. Covington, and M. M. Moore. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7:895-908.

Fule, P. Z., T. Swetnam, P. M. Brown, D. A. Falk, D. Peterson, C. D. Allen, G. H. Aplet, M. A. Battaglia, D. Binkley, C. Farris, R. E. Keane, E. Q. Margolis, H. D. Grissino-Mayer, C. Miller, C. H. Sieg, C. N. Skinner, S. L. Stephens, and A. Taylor. 2013. Unsupported inferences of high-severity fire in historical dry forests of the western United States: response to Williams and Baker. Global Ecology & Biogeography (doi:10.1111/geb.12136).

Garrison-Johnston, M. T., J. A. Morre, S. P. Cook, and G. J. Niehoff. 2003. Douglas-fir beetle infestations are associated with certain rock and stand types in the Inland Northwestern United States. Environmental Entomology 32:1354-1363.

Graham, R. T., S. McCaffery, and T. B. Jain. 2004. Science basis for changing forest structure to modify wildfire behavior and severity. General Technical Report RMRS-GTR-120, USDA Forest Service, General Technical Report RMRS-GTR-120, Fort Collins, CO.

Grissino-Mayer, H. D., C. M. Gentry, S. Croy, J. Hiatt, B. Osbrone, A. Stan, and G. DeWeese Wight. 2003. Fire history of western larch forested landscapes via tree-ring analyses. Professional Paper No. 23.

Grossman, D. H., P. Bourgeron, W. N. Busch, D. T. Cleland, W. Platts, G. C. Ray, D. R. Robins, and G. J. Roloff. 1999. Principles for ecological classification. Pages 353-394 *in* R. C. Szaro, C. J. Johnson, W. T. Sexton, and A. J. Malk, editors. Ecological stewardship: a common reference for ecosystem management. Elsevier Science, Oxford, U.K.

Gruell, G. E., W. C. Schmidt, S. F. Arno, and W. J. Reich. 1982. Seventy years of vegetative change in a managed ponderosa pine forest in western Montana: implications for resource management. USDA Forest Service, General Technical Report INT-GTR-130, Ogden, UT.

Habeck, J. R. 1990. Old-growth ponderosa pine-western larch forests in western Montana: ecology and management. Northwest Environmental Journal 6:271-292.

Habeck, J. R. 1994. Using General Land Office records to assess forest succession in ponderosa pine/Douglas-fir forests in western Montana. Northwest Science 68:69-78.

Hadley, K. S., and T. T. Veblen. 1993. Stand response to western spruce budworm and Douglas-fir bark beetle outbreaks, Colorado Front Range. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 23:479-491.

Hagle, S. K., K. E. Gibson, and S. Tunnock. 2003. Field guide to diseases and insect pests of Northern Central Rocky Mountain conifers. USDA Forest Service, Resource Note RN-03-08, Missoula, Montana. 197 pp.

Hagle, S. K., J. W. Schwandt, T. L. Johnson, S. J. Kegley, C. B. Randall, J. E. Taylor, B. Lockman, and N. J. Sturdevant. 2000. Successional functions of pathogens and insects - ecoregional section M332a and M333d in northern Idaho and western Montana; Volume 2. Results and conclusions. USDA Forest Service Health Protection Report 00-11.

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

Hagmann, R. K., J. F. Franklin, and K. N. Johnson. 2014. Historical conditions in mixed-conifer forests on the eastern slopes of the northern Oregon Cascade Range, USA. Forest Ecology and Management 330:158-170.

Hanberry, B. B. 2014. Compositional changes in selected forest ecosystems of the western United States. Applied Geography 52:90-98.

Hanberry, B. B., R. F. Noss, H. D. Safford, S. K. Allison, and D. C. Dey. 2015. Restoration is preparation for the future. Journal of Forestry 113:425-429.

Harrington, M. G., and F. G. Hawksworth. 1990. Interaction of fire and dwarf mistletoe on mortality of southwestern ponderosa pine. Pages 234-240 *in* Effects of Fire Management of Southwestern Natural Resources: Symposium Proceedings, Fort Collins, CO. USDA Forest Service, General Technical Report RMRS-GTR-191, Fort Collins, CO.

Hart, J. L., M. L. Buchanan, and L. E. Cox. 2015. Has forest restoration been freed from the bonds of history? Journal of Forestry 113:429-430.

Hartwell, M. G., P. Alaback, and S. F. Arno. 2000. Comparing historic and modern fires on the Bitterroot front. Pages 11-16 *in* The Bitterroot Ecosystem Management Research Project: what have we learned? USDA Forest Service, Proceedings RMRS-P-17, Denver, CO.

Harvey, A. E. 1994. Integrated roles for insects, diseases, and decomposers in fire-dominated forests of the Inland Western United States: past, present, and future forest health. Journal of Sustainable Forestry 2:211-220.

Harvey, A. E. 1998. Fire as an ecological factor in forest health. Pages 93-98 *in* K. Close and E. Bartlette, eds., Fire management under fire (adapting to change): proceedings of the 1994 Interior West Fire Council meeting

and program. International Association of Wildland Fire, Fairfield, WA.
Haufler, J. B., R. K. Baydack, H. Campa, B. J. Kernohan, L. J. O'Neil, L. Waits, and C. Miller. 2002. Performance measures for ecosystem management and ecological sustainability. The Wildlife Society, Technical Review 02-1, Bethesda, Maryland
Haufler, J. B., C. A. Mehl, and G. J. Roloff. 1999a. Conserving biological diversity using a coarse filter approach with a species assessment. Pages 107-116 *in* R. K. Baydack, H. Campa, and J. B. Haufler, editors. Practical approaches to the conservation of biological diversity. Island Press, Washington, D.C.
Henderson, E. 2008. Development of state and transition model assumptions used in National Forest Plan revision. Pages 89-97 *in* Third Forest Vegetation Simulator Conference. USDA Forest Service, Proceedings RMRS-P-54, Fort Collins, CO.
Hermoso, V., S. R. Januchowski-Hartley, and R. L. Pressey. 2013. When the suit does not fit biodiversity: loose surrogates compromise the achievement of conservation goals. Biological Conservation 159:197-205.
Hessburg, P., R. G. Mitchell, and G. M. Filip. 1994. Historical and current roles of insects and pathogens in eastern Oregon and Washington forested landscapes. USDA Forest Service, General Technical Report PNWR-GTR-327. Portland, Oregon.
Hessburg, P. F., and J. K. Agee. 2003. An environmental narrative of Inland Northwest United States forests, 1800-2000. Forest Ecology and Management 178:23-59.
Hessburg, P. F., J. K. Agee, and J. F. Franklin. 2005. Dry forests and wildland fires of the inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management 211:117-139.
Hessburg, P. F., R. B. Salter, and K. M. James. 2007. Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology 22:5-24.
Hessburg, P. F., R. B. Salter, M. B. Richmond, and B. G. Smith. 2000. Ecological subregions of the Interior Columbia Basin, USA. Applied Vegetation Science 3:163-180.
Hessburg, P. F., B. G. Smith, R. B. Salter, R. D. Ottmar, and E. Alvarado. 2000. Recent changes (1930s-1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management 136:53-83.
Hessburg, P. F., T. A. Spies, D. A. Perry, C. N. Skinner, A. H. Taylor, P. M. Brown, S. L. Stephens, A. J. Larson, D. J. Churchill, N. A. Povak, P. H. Singleton, B. McComb, W. J. Zielinski, B. M. Collins, R. B. Salter, J. J. Keane, J. F. Franklin, and G. Riegel. 2016. Tamm review: management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. Forest Ecology and Management 366:221-250.
Heyerdahl, E. K., K. Lertzman, and C. M. Wong. 2012. Mixed-severity fire regimes in dry forests of southern interior British Columbia, Canada. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 42:88-98.
Heyerdahl, E. K., P. Morgan, and J. P. Riser, II. 2008. Cross-dated fire histories (1650-1900) from ponderosa pine-dominated forests of Idaho and western Montana. USDA Forest Service, General Technical Report RMRS-GTR-214, Fort Collins, CO. 83 pp.
Hjort, J., J. E. Gordon, M. Gray, and M. L. Hunter. 2015. Why geodiversity matters in valuing nature's stage. Conservation Biology 29:630-639.
Holland, D. G. 1986. The role of forest insects and diseases in the Yellowstone ecosystem. Western Wildlands 12:19-23.
Holling, C. S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics 4:1-23.
Howe, E., and W. L. Baker. 2003. Landscape heterogeneity and disturbance interactions in a subalpine watershed in northern Colorado, USA. Annals of the Association of American Geographers 93:797-813.
Hutto, R. L., R. E. Keane, R. L. Sherriff, C. T. Rota, L. A. Eby, and V. A. Saab. 2016. Toward a more ecologically informed view of severe forest fires. Ecosphere 7:e01255.01210.01002/ecs01252.01255.
Kaufmann, M. R., D. Binkley, P. Z. Fulé, M. L. Johnson, S. L. Stephens, and T. W. Swetnam. 2007. Defining old growth for fire-adapted forests of the western United States. Ecology and Society 12(2):15.
Keane, R. E., S. F. Arno, and L. J. Dickinson. 2006. The complexity of managing fire-dependent ecosystems in wilderness: relict ponderosa pine in the Bob Marshal Wilderness. Ecological Restoration 24:71-78.
Keane, R. E., G. J. Cary, I. D. Davies, M. D. Flannigan, R. H. Gardner, S. Lavorel, J. M. Lenihan, C. Li, and T. S. Rupp. 2004. A classification of landscape fire succession models: spatial simulations of fire and vegetation dynamics. Ecological Modelling 179:3-27.
Keane, R. E., G. Dillon, T. B. Jain, A. T. Hudak, P. Morgan, E. Karau, and R. S. Sikkink. 2012. The problems with fire

severity and its application in fire management. USDA Forest Service, Rocky Mountain Research Station unpublished report. 39 pp.

Keane, R. E., P. F. Hessburg, P. B. Landres, and F. J. Swanson. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management 258:1025-1037.

Keane, R. E., K. C. Ryan, T. T. Veblen, C. D. Allen, J. Logan, and B. Hawkes. 2002. Cascading effects of fire exclusion in Rocky Mountain ecosystems. USDA Forest Service, General Technical Report RMRS-GTR-91, Fort Collins, CO. 24 pp.

Keeling, E. G., A. Sala, and T. H. DeLuca. 2006. Effects of fire exclusion on forest structure and composition in unlogged ponderosa pine/Douglas-fir forests. Forest Ecology and Management 237:418-428.

Kilgore, B. M. 1981. Fire in ecosystem distribution and structure: western forests and scrublands. Pages 58-89 *in* H. A. Mooney, T. M. Bonnicksen, and N. L. Christensen, editors. Proceedings of the Conference: Fire Regimes and Ecosystem Properties. USDA Forest Service, General Technical Report WO-GTR-26.

Kipfmueller, K. F., and J. A. Kupfer. 2005. Complexity of successional pathways in subalpine forests of the Selway-Bitterroot Wilderness Area. Annals of the Association of American Geographers 95:495-510.

Kolb, T. E., K. M. Holmberg, M. R. Wagner, and J. E. Stone. 1998. Regulation of ponderosa pine foliar physiology and insect resistance mechanisms by basal area treatments. Tree Physiology 18:375-381.

Korb, J. E., P. Z. Fule, and R. Wu. 2013. Variability of warm/dry mixed conifer forests in southwestern Colorado, USA: Implications for ecological restoration. Forest Ecology and Management 304:182-191.

Kroll, A. J., and J. B. Haufler. 2009. Development and evaluation of habitat models at multiple spatial scales: a case study with the dusky flycatcher. Forest Ecology and Management 229:161-169.

Landres, P. B., P. Morgan, and F. J. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179-1188.

Larson, A. J., and D. Churchill. 2012. Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management 267:74-92.

Larson, E. R., S. L. Van De Gevel, and H. D. Grissino-Mayer. 2009. Variability in fire regimes of high-elevation whitebark pine communities, western Montana, USA. Ecoscience 16:282-298.

Laternser, M., and M. Schneebeli. 2002. Temporal trend and spatial distribution of avalanche activity during the last 50 years in Switzerland. Natural Hazards 27:201-230.

Lecina-Diaz, J., A. Alvarez, and J. Retana. 2014. Extreme fire severity patterns in topographic, convective and wind-driven historical wildfires of Mediterranean pine forests. Plos One 9:e85127.

Lehmkuhl, J. F., P. Hessburg, E. K. Everett, D. E. Huff, and R. D. Ottmar. 1994. Historical and current forest landscape of eastern Oregon and Washington. Part 1. Vegetation pattern and insect and disease hazards. USDA Forest Service, Portland, Oregon.

Lentile, L. B., F. W. Smith, and W. D. Shepperd. 2005. Patch structure, fire-scar formation, and tree regeneration in a large mixed-severity fire in the South Dakota Black Hills, USA. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 35:2875-2885.

Lentile, L. B., F. W. Smith, and W. D. Shepperd. 2006. Influence of topography and forest structure on patterns of mixed severity fire in ponderosa pine forests of the South Dakota Black Hills, USA. International Journal of Wildland Fire 15:557-566.

Long, J. N. 2003. Diversity, complexity and interactions: an overview of Rocky Mountain forest ecosystems. Tree Physiology 23:10911-11099.

Long, J. N. 2009. Emulating natural disturbance regimes as a basis for forest management: A North American view. Forest Ecology and Management 257:1868-1873.

Losensky, B. J. 1994. Historical vegetation types of the interior Columbia River Basin. USDA Forest Service, Intermountain Research Station unpublished report. 108 pp.

Lydersen, J., and M. North. 2012. Topographic variation in structure of mixed-conifer forests. Ecosystems 15:1134-1146.

Marcoux, H. M., L. D. Daniels, S. E. Gergel, E. Da Silva, Z. Gedalof, and P. Hessburg. 2015. Differentiating mixed- and high-severity fire regimes in mixed-conifer forests of the Canadian Cordillera. Forest Ecology and Management 341:45-58.

Marcoux, H. M., S. E. Gergel, and L. D. Daniels. 2013. Mixed-severity fire regimes: how well are they represented by existing fire-regime classification systems? Canadian Journal of Forest Research-Revue Canadienne De

Recherche Forestiere 43:658-668.
Mast, J. N. 1993. Climatic and disturbance factors influencing *Pinus ponderosa* stand structure near the forest/grassland ecotone in the Colorado Front Range. PhD Thesis, University of Colorado.
Mast, N. J., P. Z. Fule, M. M. Moore, W. Covington, and A. E. M. Waltz. 1999. Restoration of presettlement age structure of an Arizona ponderosa pine forest. Ecological Applications 9:228-239.
Mast, N. J., T. T. Veblen, and Y. B. Linhart. 1998. Disturbance and climate influences on age structure of ponderosa pine at the pine/grassland ecotone, Colorado Front Range. Journal of Biogeography 25:743-755.
McAlpine, C., C.P. Catterall, R. MacNally, D. Lindenmayer, J.L. Reid, K.D. Holl, A.F. Bennett, R.K. Runting, K. Wilson, R.J. Hobbs, L. Seabrook, S. Cunningham, A. Moilanen, M. Maron, L. Shoo, I. Lunt, P. Vesk, L. Rumpff, T.G. Martin, J. Thomson, and H. Possingham. 2016. Integrating plant- and animal-based perspectives for more effective restoration of biodiversity. Frontiers in Ecology and the Environment 14:37-45.
McGaughey, R. J. 2004. Stand Visualization System, Version 3.3. USDA Forest Service, Pacific Northwest Research Station unpublished report. 141pp.
Merschel, A. G., T. A. Spies, and E. K. Heyerdahl. 2014. Mixed-conifer forests of central Oregon: effects of logging and fire exclusion vary with environment. Ecological Applications 24:1670-1688.
Meyn, A., and M. C. Feller. 2006. Fire history of forest remnants in wetter lodgepole pine dominated forests in southern British Columbia, Canada. Northwest Science 80:86-94.
Milburn, A., B. Bollenbacher, and M. Manning. 2015. Region 1 - Existing and potential Vegetation groupings used for broad-level analysis and monitoring. USDA Forest Service internal report.
Morgan, P., G. H. Aplet, J. B. Haufler, H. C. Humphries, M. M. Moore, and W. D. Wilson. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. Journal of Sustainable Forestry 2:87-112.
Morgan, P., S. C. Bunting, A. E. Black, T. Merrill, and S. Barrett. 1996. Fire regimes in the interior Columbia River Basin: past and present. USDA Forest Service, Intermountain Research Station unpublished internal report RJVA-INT-94913.
Murray, M. P., S. C. Bunting, and P. Morgan. 1998. Fire history of an isolated subalpine mountain range of the Intermountain Region, United States. Journal of Biogeography 25:1071-1080.
Negron, J. F., W. C. Schaupp Jr., K. E. Gibson, J. Anhold, D. Hansen, R. W. Thier, and P. Mocettini. 1999. Estimating the extent of mortality associated with the Douglas-fir beetle in the central and northern Rockies. Western Journal Applied Forestry 14:121-127.
Nichols, W. F., K. T. Killingbeck, and P. V. August. 1998. The influence of geomorphological heterogeneity on biodiversity II. A landscape perspective. Conservation Biology 12(2):371-379.
NRCS Soil Survey Staff. Web Soil Survey. Available online at http://websoilsurvey.nrcs.usda.gov., USDA Natural Resources Conservation Service.
Odion, D. C., C. T. Hanson, A. Arsenault, W. L. Baker, D. A. DellaSala, R. L. Hutto, W. Klenner, M. A. Moritz, R. L. Sherriff, T. T. Veblen, and M. A. Williams. 2014. Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. Plos One 9:14.
O'Laughlin, J. 1998. Forest health concepts and their application to managing Idaho forests. Pages 25-39 *in* Proceedings of the symposium: ecosystem management in Western Interior Forests, May 3-5, 1994. Washington State University, Department of Natural Resource Sciences, Pullman, WA.
Olsen, W. K., J. M. Schmid, and S. A. Mata. 1996. Stand characteristics associated with mountain pine beetle infestations in ponderosa pine. Forest Science 42:310-327.
Parker, T. J., K. M. Clancy, and R. L. Mathiasen. 2006. Interactions among fire, insects and pathogens in coniferous forests of the interior western United States and Canada. Agricultural and Forest Entomology 8:167-189.
Pfister, R. D., B. L. Kovalchik, S. F. Arno, and R. C. Presby. 1977. Forest habitat types of Montana. USDA Forest Service, General Technical Report INT-GTR-34,, Ogden, UT. 174 pp.
Powell, D. C. 1999. Suggested stocking levels for forest stands in Northeastern Oregon and Southeastern Washington: an implementation guide for the Umatilla National Forest. USDA Forest Service Pacific Northwest Region, Umatilla National Forest.
Ralls, K., S. R. Beissinger, and J. F. Cochrane. 2002. Guidelines for using population viability analysis in endangered species management. Pages 521-555 *in* S. R. Beissinger and D. R. McCullough, editors. Population viability analysis. University of Chicago Press, Chicago, IL, USA.
Randall, C. B., B. Steed, and R. Bush. 2011. Revised R1 forest health rating system user guide for use with inventory data stored in FSVeg and/or analyzed with the forest vegetation simulator. USDA Forest Service Forest

Health Protection Report 11-06, Missoula, MT.
Rollins, M. G., P. Morgan, and T. Swetnam. 2002. Landscape-scale controls over 20(th) century fire occurrence in two large Rocky Mountain (USA) wilderness areas. Landscape Ecology 17:539-557.
Roloff, G. J., and J. B. Haufler. 1997. Establishing population viability planning objectives based on habitat potentials. Wildlife Society Bulletin 25:895-904.
Roloff, G. J., and J. B. Haufler. 2002. Modeling habitat-based viability from organism to population.*in* J. M. Scott, P. J. Heglund, M. L. Morrison, J. B. Haufler, M. G. Raphael, W. A. Wall, and F. B. Samson, editors. Predicting species occurrence. Island Press, Washington, D.C., USA.
Romme, W. H., and D. G. Despain. 1989. Historical perspective on the Yellowstone fires of 1988. BioScience 39:695-699.
Saxon, E. C. 2003. Adapting ecoregional plans to anticipate the impact of climate change. Pages 345-365 *in* C. Groves, editor. Drafting a conservation blueprint - a practitioner's guide to planning for biodiversity: The Nature Conservancy. Island Press, Washington, D.C.
Schoennagel, T., T. T. Veblen, and W. H. Romme. 2004. The interaction of fire, fuels, and climate across Rocky Mountain forests. BioScience 54:661-676.
Sherriff, R. L., and T. T. Veblen. 2006. Ecological effects of changes in fire regimes in *Pinus ponderosa* ecosystems in the Colorado Front Range. Journal of Vegetation Science 17:705-718.
Stevens, J. T., H. D. Safford, M. P. North, J. S. Fried, A. N. Gray, P. M. Brown, C. R. Dolanc, S. Z. Dobrowski, D. A. Falk, C. A. Farris, J. F. Franklin, P. Z. Fule, R. K. Hagmann, E. E. Knapp, J. D. Miller, D. F. Smith, T. W. Swetnam, and A. H. Taylor. 2016. Average stand age from forest inventory plots does not describe historical fire regimes in ponderosa pine and mixed-conifer forests of western North America. Plos One 11:20.
Stine, P.A., P. Hessburg, T.A. Spies, M.G. Kramer, C.J. Fettig, J.F. Lehmkuhl, K.L. O'Hara, K. Polivka, P.H. Singleton, S. Charnley, A.G. Merschel, and R. White. 2014. 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. USDA Forest Service, General Technical Report, Portland, OR. 254 pp.
Swanson, F. J., J. A. Jones, D. A. Wallin, and J. H. Cissel. 1993. Natural variability-implications for ecosystem management. Pages 85-99 *in* M. E. Jensen and P. Bourgeron, editors. Eastside forest ecosystem health assessment: Vol. II, Ecosystem management: principles and applications. USDA Forest Service, Northern Region, Missoula, MT.
Swanson, M. E., J. F. Franklin, R. L. Beschta, C. M. Crisafulli, D. A. DellaSala, R. L. Hutto, D. Lindenmayer, and F. J. Swanson. 2010. The forgotton stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and Environment doi:10.1890/090157.
Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage for the future. Ecological Applications 9(4):1189-1206.
Swetnam, T. W., and A. M. Lynch. 1993. Multi-century, regional-scale patterns of western spruce budworm outbreaks. Ecological Monographs 63:399-424.
Turner, M. G., W. H. Romme, and R. H. Gardner. 1999. Prefire heterogeneity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. International Journal of Wildland Fire 9:21-36.
USDA Forest Service. 2002. Region One - Potential Natural Vegetation (PNV) for Western and Central Montana and Northern Idaho (unpublished GIS Data).
USDA Forest Service. 2004. Region One - Potential Vegetation Type (PVT) classification of Western Montana and Northern Idaho (unpublished GIS data).
Veblen, T. T., K. S. Hadley, E. M. Nel, T. Kitzberger, M. Reid, and R. Villalba. 1994. Disturbance regime and disturbance interactions in a Rocky Mountain subalpine forest. Journal of Ecology 82:125-135.
Wellner, C. A. 1970. Fire history in the Northern Rocky Mountains: the role of fire in the Intermountain West. . Pages 42-64 *in* Intermountain Fire Resource Council Symposium. Intermountain Fire Resource Council, Missoula, MT.
Williams, G. W. 2003. References on the American Indian use of fire in ecosystems. USDA Forest Service Washington Office unpublished paper. Washington, DC.
Williams, M. A., and W. L. Baker. 2011. Testing the accuracy of new methods for reconstructing historical structure of forest landscapes using GLO survey data. Ecological Monographs 81:63-88.

Williams, M. A., and W. L. Baker. 2012. Comparison of the higher-severity fire regime in historical (A.D. 1800s) and modern (A.D. 1984-2009) montane forests across 624,156 ha of the Colorado Front Range. Ecosystems 15:832-847.

Williams, M. A., and W. L. Baker. 2012. Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Global Ecology and Biogeography 21:1042-1052.

Williams, M. A., and W. L. Baker. 2014. High-severity fire corroborated in historical dry forests of the western United States: response to Fule et al. Global Ecology and Biogeography 23:831-835.

Wilson, J. L., and B. M. Tkacz. 1996. Historical perspective on forest insects and pathogens in the Southwest: implication for restoration of ponderosa pine and mixed conifer forests. Pages 26-31 *in* W. Covington and P. K. Wagner, editors. Conference on adaptive ecosystem restoration and management: restoration of Cordilleran conifer landscapes in North America. USDA Forest Service, General Technical Report RM-GTR-278, Flagstaff, AZ.

Winter, S. L., S. D. Fuhlendorf, C. L. Goad, C. A. Davis, and K. R. Hickman. 2011. Topoedaphic variability and patch burning in sand sagebrush shrubland. Rangeland Ecology & Management 64:633-640.

Woodall, C. W., and P. D. Miles. 2006. New method for determining the relative stand density of forest inventory plots. Proceeding of the sixth annual forest inventory and analysis symposium: 2004 September 21-24; Denver, CO. USDA Forest Service, Washington Office Report, Washington, DC.

Zimmerman, G. T., and R. D. Laven. 1994. Ecological interrelationships of dwarf mistletoe and fire in lodgepole pine forests. Pages 123-131 *in* F. G. Hawksworth and R. F. Scharpf, editors. Biology of Dwarf Mistletoes. USDA Forest Service, General Technical Report, Fort Collins, CO.