Ecology and Management of Fire-prone Forests of the Western United States

Society for Conservation Biology Scientific Panel on Fire in Western U.S. Forests

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EXECUTIVE SUMMARY

Fire is a primary natural disturbance in most forests of western North America and has shaped their plant and animal communities for millions of years. Native species and fundamental ecological processes are dependent on conditions created by fire. However, many western forests have experienced shifts in wildfire regimes and forest structure following a century or more of resource use and management, with some past and present management activities lacking a scientific basis. Changes in wildfire and fuel management policies are needed to address social and environmental problems that have arisen as a result of these activities.

Incorporation of current scientific knowledge into revised policies and practices is essential to insure that the productivity, biological diversity, and ecological values of western forests are sustained. As an example, implementation of the Healthy Forests Restoration Act (HFRA) of 2003 (Public Law 108-V148) will benefit from adaptive application of the dramatically expanding base of scientific knowledge. Our review addresses the ecological science relevant to developing and implementing forest restoration and fuel management policies, including activities conducted before, during, and after forest wildfires. An essential principle of ecological variability within and among forests underlies all of our findings.

In this summary and in the background report we use the term "characteristic" in referring to the dominant natural disturbance regime of a forest type or site. For example, some types of dry forests are described as being historically or naturally characterized by a low-severity fire regime while some coastal and subalpine forests are characterized by a high-severity fire regime. These are generalized characterizations of the regimes that these types experience and are not necessarily exclusive. For example, forests characterized by high-severity fire regimes may also experience low-severity events and vice versa. The term "uncharacteristic" refers to disturbances, forest structure, or fuel loads of a scale or type outside the historical range of variability based on site-specific vegetation reconstructions using tree rings, fire scars, pollen, charcoal, or early historical records.

Fire in Western Forests

Wildfire is inevitable and ecologically important in forests throughout much of the western United States, given the fuels, ignition sources, and variable climatic conditions. Nevertheless, characteristic fire regimes—especially the extent, frequency, and severity of the wildfires—are immensely variable. For example, fires historically recurred in western forests at intervals ranging from as frequently as a decade or less in some dry ponderosa pine forests to 250 to 800 years or more in forests at high elevations and along the Pacific Coast. Fires provide important services such as recycling nutrients, regulating the density and composition of young trees, and creating and shaping wildlife habitat at the stand level. At larger spatial scales wildfire influences landscape patterns and affects water and sediment delivery in watersheds. Many native plant and animal species are adapted to post-fire habitats and suffer population declines with fire exclusion.

Characteristic fire regimes differ markedly among forest types and regions—as well as within major forest types—and these differences need to be considered in fire and fuel management policies to assure that these policies are effective and sustain ecological values. Managers, stakeholders, and policy makers are challenged by the complexity created by this variability, which defies a simple, one-size-fits-all prescription. Fortunately, plant association groups (PAGs) provide a surrogate classification of this diversity in forest wildfire regimes that is effective and scientifically credible, since plant associations have predictable relationships to characteristic fuels and fire regimes.

Forest Management Before Wildfire

How could forests be managed prior to the inevitable wildfires they will experience, so as to insure that fires will play their characteristic roles in maintaining the composition, structure, and function of the forest ecosystem when they do occur? Appropriate management will vary greatly with the type of forest and its dominant fire regime. Determining the appropriate management and restoration goals requires that the effects of past land uses first be identified so that those effects can be specifically remedied. Then appropriate ecologically based restoration and management policies can be developed. Protected areas require particular management approaches that may differ from practices appropriate in managed forests. Each of these topics is addressed in turn below.

Variable Effects of Fire Exclusion, Logging, Livestock Grazing, and Plantations

The effects of fire exclusion, as well as other activities that affect fire regimes (e.g., logging, livestock grazing, plantations) on forest structure are not necessarily easy to identify or demonstrate scientifically; they also vary significantly among forest types and regions. In some forest types change has been dramatic since European settlement due, for example, to fire exclusion, logging, grazing, or tree planting (singly or in combination), and restoration is clearly needed. In other forest types major changes are not apparent and restoration is not needed. In many cases it has been inappropriately assumed that forests in general or all forests dominated by a particular tree species have been altered in the same way. In fact, these effects are known to vary, depending upon the forest type and whether fire was characteristically high, mixed, or low severity, each of which is discussed below.

Key Findings:

- **Fire exclusion and other human activities have led to significant deviations from historical variability in some forests but not in others.** Restoration treatments are warranted, sometimes urgently, in those cases where such activities have led to significant alterations in ecosystem structure, function, or composition, but cannot be justified ecologically in cases where such changes have not occurred. The following sections discuss this for forests with different fire regimes.

- **Land uses and fire exclusion do not universally increase fuel loads or fire risk.** Such activities may alter fuels in divergent or complex ways that lead to a need for decreases in particular fuels and increases in other fuels, if restoration to the historical range of variability is the goal. For example, fire exclusion can increase tree regeneration and ladder fuels in some cases and decrease tree regeneration and ladder fuels in other cases.

Forests Characterized by High-Severity Fires

Forests characterized by high-severity fires are found in several disparate locations: subalpine forests at higher elevations throughout the West (e.g., lodgepole pine and Engelmann spruce-subalpine fir); the moist and highly productive forests in marine-influenced regions of the Pacific Northwest; and certain semi-arid woodlands, including some dominated by pinyon-juniper and by oak-pine-chaparral. High-severity fires, which are usually infrequent, kill most or all of the trees in large portions of the burn, although such fires typically create a landscape mosaic that also includes some areas of unburned forest and of low- to moderate-severity burn. Forests subject to high-severity fires typically support high densities of trees and other woody plants and, consequently, large fuel loadings. When these dense fuels dry out and an ignition source is present, the resulting fires can spread rapidly and quickly become difficult or impossible to suppress. Many large, high-severity fires are probably associated with either infrequent, severe droughts or short-term synoptic weather patterns or both.

Key Findings:

- **Fire exclusion has had little to no effect on fuels or forest structure in forests characterized by high-severity fire regimes**—a fact that is especially relevant to fire policy. High-severity fires are relatively infrequent—coming at intervals of one to many centuries—while the period of active fire exclusion in these remote forests has been less than a century. Land uses, including logging, plantations, and grazing, may have extensively modified the structure of these forests in some areas, but evidence suggests that fire regimes have not been fundamentally modified.

- **Because fuel structures or tree densities are usually well within the historical range of variability, "restorative" treatments are ecologically inappropriate in forests characterized by stand-replacement fire.** Modifying stand densities and fuels to levels that would reduce the potential for stand-replacement fire would render these forests incapable of fulfilling their characteristic ecological roles, including provision of high densities of standing dead trees (snags) and other critical elements of fish and wildlife habitat that are created by fire. Restoration could address other needs, such as restoring native understory plant diversity, where land use is known to have caused changes.

Forests Characterized by Mixed-Severity Fires

Fire is quite variable in severity and frequency in many mid-elevation and some low-elevation forests of moderate to high productivity across variable topography in the interior west and some coastal regions, such as the Klamath-Siskiyou region. In these forests both low- and high-severity fires may occur, with the former often more frequent than the latter. Topographically complex western mountain landscapes may be especially prone to mixed-severity fire, because drier south-facing slopes with lower fuel loads can burn at low severity when adjacent, moister north-facing slopes that support higher tree densities experience high-severity fire. The inherent variability of mixed-severity fire regimes precludes easy detection and analysis of the effects of fire exclusion. Exclusion of fire may have allowed tree densities to increase in some areas but post-fire tree density is naturally high in patches killed by high-severity fire.

Key Finding:

- **Scientific understanding of mixed-severity forest landscapes is limited, making it difficult to provide ecologically appropriate guidelines for restorative treatments.** These are most often very complex landscape mosaics; hence, it is necessary to plan and conduct activities at larger spatial scales. In mixed-severity forest landscapes where sufficient ecological and fire-history information is available, a combination of thinning and prescribed fire may be useful in restoration. However, only portions of these landscapes will warrant treatment from an ecological perspective that recognizes the spatially complex patterns. More scientific research is needed to understand the dynamics of mixed-severity forest landscapes.

Forests Characterized by Low-Severity Fires

The consequences of many human activities—including fire exclusion, logging, tree planting, and livestock grazing—are most serious in forest types that historically were characterized primarily by low-severity fires. Low-severity fire regimes were typical of many (but not all) pine and dry mixed-conifer forests, which occurred on warm, dry sites prior to European settlement. These fires historically burned fine fuels (e.g., grasses and litter on the forest floor) at regular intervals. Such surface fires killed few large fire-resistant trees but killed many smaller trees of all species, helping to maintain open-canopied stands of large, old trees. Human activities since European settlement have dramatically modified the fuel structure in these forests. Logging of large fire-resistant trees has eliminated key ecological elements of these forests, including the large trees, snags, and logs essential to many ecological functions, such as provision of fish and wildlife habitat. Logging also has promoted higher stand densities in many dry forests by stimulating dense natural regeneration, even when it was not followed by aggressive replanting.

Key Findings:

- **Restoration of dry ponderosa pine and dry mixed-conifer forests—where low-severity fires were historically most common—is appropriate and desirable ecologically on many sites.** Mechanical thinning of small stems and prescribed fire are effective techniques for restoring stand densities to levels that existed prior to fire exclusion, livestock grazing, logging, and plantation establishment.

- **Retention of large and/or old live trees, large snags, and large down logs in restoration treatments, such as thinning, is critical to restoring and maintaining ecological function.** Also, other key components of these ecosystems, such as native understory plants, must be restored or protected for full restoration of natural conditions, including the potential for characteristic fire behavior.

Priorities and Principles of Ecologically-Based Forest Restoration

Forest restoration varies along a continuum from restoring structure (e.g., reducing densities of small trees and increasing the density of large trees) to restoring the processes (e.g., low-severity fire, competition between grasses and tree seedlings) that create and maintain that structure. The continuum also represents a gradient from symptoms (e.g., uncharacteristically high tree densities) to causes (e.g., exclusion of fire). A well-established principle in land health, as in human health, is that treating symptoms may be necessary in the short term, but that ultimately causes must be identified and treated to restore health.

Appropriate models for restoration will vary with current forest conditions, management objectives, and plant association groups, among other factors. An essential early step in a management program is to identify the Desired Future Condition (DFC) to which treatments are directed. DFCs are often based on conditions that are considered to be within the historical range of variability (HRV). Precisely achieving some past condition is not a reasonable goal, but conditions broadly representative of the historical range of variability can often be approximated through restorative activities. Restoration of processes (e.g., low-severity fire) may allow the re-structured forest to eventually equilibrate with contemporary environmental conditions. The level of threat to particular natural values—such as critical wildlife habitat, watershed and aquatic values, and existing populations of veteran old trees—should be considered in setting priorities for restoration treatments.

Areas in the wildland-urban interface (WUI) may require fuel reduction and fire management policies that are inconsistent with HRV or with maintaining the biodiversity of those sites, even though carefully tailored treatments can maintain some aspects of biodiversity. Growth-management policies could minimize adverse ecological impacts from the WUI.

We provide two case studies—the Klamath Reservation Forest and Rocky Mountain ponderosa pine–Douglas-fir forests—in the background report to illustrate the wide variety of ecological conditions and ecologically appropriate management and restoration practices in western forests.

Key Findings:

- **From an ecological perspective priorities for restoration need to be determined on the basis of ecological considerations and urgency outside of the wildland-urban interface (WUI).** High-priority cases are likely to include areas where significant ecological values are at risk of undesirable stand-replacement fire. Many of these are outside of the WUI.

- **On lands where ecological objectives dominate, the desired goal will often be a forest ecosystem with its fire regime, fuels, tree population structure, and other living organisms restored to within the historical range of variability.** Ideally, the conditions created must be consistent with the characteristic fire regime of the site—i.e., sustainable in the context of the probable fire regime. Deviation from historic conditions sometimes may be necessary, however, to accommodate an altered biota or environment, or appropriate social objectives. In such cases the highest conservation values are likely to be obtained by minimizing deviations from the historical range of variability.

- **Broader conception and implementation of restoration objectives, beyond fuel and fire mitigation, are necessary** to achieve comprehensive, scientifically based approaches to ecological restoration of western forests. An example is the restoration of understory plant communities.

- **Restoration plans must recognize and systematically incorporate fire management needed to maintain the restored forest**. Forests are dynamic; therefore, any restoration program has to provide for sustained fire management in order to maintain the desired condition. A common-sense goal consistent with ecological science is to achieve restored forests that are low maintenance, such as can be achieved through managed natural fire, and, where this is not possible, to use prescribed fire that seeks to mimic as closely as possible the characteristic fire regime.

- **Large trees of fire-resistant species and large snags and logs have high ecological importance and should be retained in restoration projects with ecological goals.**

Where present, large and old live trees are the most fire-resistant component of western forests and are essentially irreplaceable. Snags and logs on the forest floor are key wildlife features that are deficient in many western forests due to logging.

- **There are risks associated with restorative treatment of stands and landscapes including**: (1) Uncertainties associated with basing treatments on inadequate knowledge; and (2) Risks associated with not taking restorative actions, including the potential loss of significant ecological values. An example of the latter is potential loss of spotted owl habitat to stand-replacement fire, which is uncharacteristic in some landscapes, such as on the lands that previously constituted the Klamath Indian Reservation in the Eastside Cascades. Again, we emphasize the need to recognize variability, as portions of landscapes that are generally characterized as falling within a low-severity fire regime did experience high-severity fire, at least on occasion.

- **Adaptive management, including properly designed monitoring activities, needs to be a part of all major restoration programs.** Many proposed research and monitoring activities associated with restoration programs have lacked both sufficient and sustained funding. Creation of a dedicated funding mechanism to support these activities is imperative for proposals to provide critical feedback to managers and, secondarily, to have credibility with stakeholders.

- **Credible, third-party scientific reviews are critical when major controversies arise as to the scientific merits of proposed activities.** Regular processes or mechanisms for the initiation and nature of these scientific reviews need to be established along with appropriate funding mechanisms.

Protected Areas Are Essential for Managing Fire for Ecological Diversity

Not all conservation needs can be met in managed forests. Reserves of various kinds are a fundamental conservation tool whether they are congressionally recognized (e.g., national parks and wilderness), land allocations (e.g., Late Successional Reserves), or de facto reserves (e.g., roadless areas). They provide essential enclaves for species and serve as control or reference sites for lands managed for commodities. The question of how reserves in fire-prone landscapes should be managed cannot be addressed by application of a simplistic "one-size-fits-all" philosophy, but must be guided by consideration of the vegetation structure and composition of the area in question and its characteristic fire regime.

Key Findings:

- **Reserves may be required for species closely associated with late- or early-successional forests in fire-prone landscapes for a variety of reasons.** For example, unreserved forests are often fragmented by periodic logging or consist only of stands of trees too small or too open to meet the needs of late-successional species, such as spotted owls. Species typical of natural post-fire habitats (e.g., many woodpeckers), which contain abundant standing dead trees, require substantial areas reserved from post-fire logging.

- **The reserve concept does provide for appropriate kinds of management and ecologically compatible human use.** Restoring a natural fire regime is most compatible with the reserve concept, but in cases where fully restoring a natural fire regime is not feasible, ecologically appropriate management will likely be needed to restore and maintain biodiversity and the conditions for which reserves were set aside. Some types of management, such as prescribed burning, and some uses, such as ecological research and monitoring, are often essential to the persistence of populations, habitat features, and key ecological processes within reserves. The general goal would be to restore the reserve landscape to a condition within the historical range of variability (where restoration is necessary) and then to maintain it in that state with minimal human intervention, or allow it to equilibrate with contemporary natural conditions.

Management Activities During Wildfire

Fire management policies provide direction regarding responses to wildfire, including such basic issues as whether or not to suppress wildfires. A generalized policy regarding fire suppression is inappropriate as evidenced by the negative ecological (and other) impacts of a universal fire-suppression policy during the 20th century. Decisions regarding appropriate response to fire need to consider many ecological and social factors, beginning with the nature of the forest type and societal goals.

Key Findings:

- **From an ecological perspective, allowing fires to serve their natural role may be most beneficial ecologically.** Certainly, fire must be managed when close to human settlements and infrastructure and in some cases where economic resource values are high. Away from these areas—such as in many wilderness areas, national parks, and large areas of contiguous public lands—there is opportunity to increase the use of wildland fire, thus benefiting the range of species that require a diversity of natural fire regimes.

- **Fire suppression may be beneficial to ecological values in some forest landscapes, particularly where special values are at risk.** For example, fire suppression may be appropriate where rare or unique ecological values (including imperiled species habitat) could be lost, where uncharacteristic fuel accumulations have created the potential for a fire that is outside the historical range of variability, or where infrequent high-severity fires are characteristic but where such fires are not currently viewed as ecologically desirable (e.g., old-growth forests in Pacific Northwest).

Forest Management After Wildfire

Forest landscapes that have been affected by a major natural disturbance—such as a severe wildfire or windstorm event—are commonly viewed as devastated and biologically impoverished. Such perspectives are usually far from ecological reality. Overall species diversity measured as number of species—at least of higher plants and vertebrates—is often highest following a natural stand-replacement disturbance and before re-development of closed-canopy forest. Important reasons for this include an abundance of biological legacies, such as living organisms and dead tree structures, the migration and establishment of additional organisms adapted to the disturbed, early-successional environment, and temporary release of other plants on the site from dominance by trees.

Currently, natural, early-successional forest habitat—naturally disturbed areas with a full array of legacies (i.e., not subject to post-fire logging) and experiencing natural recovery processes (i.e., not seeded or planted)—are among the scarcest habitat condition in some regions, such as the Pacific Northwest.

Key Findings:

- **Research by both ecologists and foresters provides evidence that areas affected by large-scale natural disturbances often recover naturally.** Post-burn landscapes have substantial capacity for natural recovery. Reestablishment of closed forest following stand-replacement fire characteristically occurs at widely varying rates, providing temporary, but ecologically important and now rare early-successional habitat for a variety of native species and key ecological processes.

- **Post-fire (often called "salvage") logging does not contribute to ecological recovery; rather it negatively impacts recovery processes, with the intensity of such impacts depending upon the nature of the logging activity.** Post-fire logging in naturally-disturbed forest landscapes generally has no direct ecological benefits and many potential negative impacts from an ecological standpoint. Trees that survive the fire for even a short period of time are critical as seed sources and as habitat that will sustain many elements of biodiversity both above and below ground. The dead wood, including large snags and logs, is second only to live trees in overall ecological importance. Removal of these structural legacies—living and dead—is inconsistent with our scientific understanding of natural disturbance regimes and short- and long-term recovery processes.

- **Post-fire logging destroys much of whatever natural tree regeneration is occurring on a burned site.** This is a fundamental concern since these tree seedlings are derived from local seed sources, which are most likely the best adapted to the site. Furthermore, environmental variables, such as moisture and temperature conditions, are major selective factors in determining which natural tree seedlings survive, which favors genotypes more tolerant of environmental stresses than are nursery- or greenhouse-grown seedlings.

- **Evidence from empirical studies is that post-fire logging typically generates significant short- to mid-term increases in fine and medium fuels.** In some cases this may result in increased reburn potential rather than a decreased reburn potential, as is often claimed. In any case, from an ecological perspective large wood is of demonstrated importance in ecological recovery; removing this wood in an attempt to influence the behavior of a potential reburn event has little scientific support.

- **In forests subjected to severe fire and post-fire logging, streams and other aquatic ecosystems will take longer to return to historic conditions or may switch to a different (and often less desirable) state altogether.** Following a severe fire the biggest impacts on aquatic ecosystems are often increased sedimentation caused by runoff from roads. High sediment loads from roads may continue for years, greatly increasing the time for recovery.

- **Post-fire seeding of non-native plants generally damages natural ecological values, such as reducing the recovery of native plant cover and biodiversity, including tree regeneration.** Non-native plants typically compete with native species, reducing both native plant diversity and cover. Reductions in natural tree regeneration as a result of seeding of non-native plants have also been reported in numerous studies.

- **Post-fire seeding of non-native plants is often ineffective at reducing soil erosion.** Aerial seeding of grasses (primarily non-native) is common on federal lands following moderate- to high-severity fire to reduce post-fire erosion. The effectiveness of seeding in reducing erosion is mixed. Grass seeding generally does not mitigate erosion during the first winter following fire, when seeded grasses are not yet well established. Seeding may slow erosion during the second year following fire but is rarely effective during intense storms.

- **There is no scientific or operational linkage between reforestation and post-fire logging; potential ecological impacts of reforestation are varied and may be either positive or negative depending upon the specifics of activity, site conditions, and management objectives. On the other hand, ecological impacts of post-fire logging appear to be consistently negative.** Salvage and reforestation are often presented as though they are interdependent activities, which they are not from either a scientific or operational perspective. From a scientific perspective, policy and practice should consider each activity separately. As noted above, post-fire logging is a consistently negative practice from the standpoint of ecological recovery. Natural tree regeneration is ecologically most appropriate, but intentional reforestation could also be designed to provide significant ecological benefits in some cases.

- **Accelerated reestablishment of extensive closed forest conditions after fire is usually not an appropriate objective on sites managed with a major ecological focus**. Wildfires have been viewed historically as events that destroy valuable standing forest and create undesirable expanses of deforested (i.e., unproductive) landscape. Re-establishment of fully stocked stands of commercially important tree species on burned sites has been a fundamental forest management objective on most private and public forestlands; hence the historic commitment to intensive reforestation. However, timber production is no longer the primary objective on many federal lands, where the focus on provision of biodiversity and ecosystem services equals or exceeds wood production objectives. The ecological importance of biological legacies and of uncommon, structurally complex early-successional stands argues against actions to achieve rapid and complete reforestation except where the primary goal is wood production. In addition, it is also inappropriate to re-establish fully stocked stands on sites characterized by low-severity fire—the same sites where managers are trying to restore fuel loadings to their historical range of variability.

- **Where timber production, other societal management goals, or special ecological needs are the focus, planting or seeding some native trees and other plants using local seed sources may be appropriate.** Ecological assessments of the post-burn area and considerations of management objectives should be used to determine appropriate activity. Special ecological circumstances might include a need to restore an uncommon plant species or habitat for a threatened or endangered species. Innovative practices, such as low or variable density planting, will likely be more appropriate ecologically than traditional practices that involve dense tree plantations of one or a few commercial species. Dense uniform conifer plantations are always inappropriate on sites characterized by low-severity fire unless the intent is intensive management of such sites for wood production.

More Ecological Science is Needed in Fire Management

Despite the complexity of fire ecology in western forests and uncertainty over the effects of particular management actions, the scientific basis for rational decision-making about fire has improved dramatically in recent years. Some of this improvement is evident in law and policy. For example, there is explicit attention to old-growth and characteristic forest structure in the Healthy Forests Restoration Act (HFRA) of 2003:

> *"In carrying out a covered project, the Secretary shall fully maintain, or contribute toward the restoration of, the structure and composition of old growth stands according to the pre-fire suppression old growth conditions characteristic of the forest type, taking into account the contribution of the stand to landscape fire adaptation and watershed health, and retaining the large trees contributing to old growth structure."*

Nevertheless, current approaches to implementation of HFRA may be flawed; while attempts are being made to incorporate the variability of fire regimes and vegetation dynamics among forest types, there is heavy reliance on expert opinion and unvalidated, over-specified models. Critical review of the scientific basis for HFRA, FRCC (Fire Regime Condition Classes), and LANDFIRE from a credible independent source, such as the National Academy of Sciences, is needed.

More generally, principles of ecological science and the detailed existing knowledge of individual forest ecosystems need to be incorporated more systematically into the development of forest fire and fuel policies. A current example is the need to incorporate ecological principles into proposed legislation dealing with post-fire (salvage) logging and reforestation.

One barrier to better use of ecological science is that scientists involved in developing fire policies and practices have tended to be specialists in fire and fuel management, not ecologists, conservation biologists, or other broadly trained scientists. It is not surprising, then, that current forest law and policy, such as HFRA, does not adequately incorporate ecological science in its implementation and tends to promote a narrow definition of restoration that focuses almost exclusively on fuels.

True ecological restoration requires the maintenance of ecological processes, native species composition, and forest structure at both stand and landscape scales. Because ecological variability is great, few universal principles exist for integrating insights from ecology and conservation biology into fire management and conservation policies. Nevertheless, one principle that does seem to hold is that as forests are managed or restored, they should not only support the desired fire regime but also viable populations of native species in functional networks of habitat.

INTRODUCTION

The vegetation of North America has been shaped by recurring fires over millions of years. For example, fossils of pines (*Pinus* spp.), which are closely associated with fire, date from the Cretaceous Period more than 100 million years ago (Agee 1998a, Millar 1998). Fire remains the primary natural disturbance influencing the plant and animal communities of most western forests today (Habeck and Mutch 1973, Agee 1993). Without periodic fire, many of the characteristic landscapes of the West would change dramatically and many species that evolved in the presence of fire likely would decline or become extinct. Humans have used fire for various purposes for thousands of years, although the extent of their influence on fire regimes in the American West is controversial and was probably not as great as often assumed (Vale 2002 but see Anderson 2005). On the other hand, there is little disagreement that many western forests have been degraded over the past century by ill-informed management. Uncharacteristic fuel loads contribute to altered fire regimes in some forest types. Urbanization and increased human habitation of wildlands have intensified problems of managing fire in forested landscapes and the wildland-urban interface (see Dombeck et al 2004) and have created new problems such as concerns about smoke from prescribed fires.

The consequences of disrupted forest structures and fire regimes are most serious in forest types that were historically characterized by low-severity fire. Many of these forests, which are usually at low elevations and often close to human settlements, are currently experiencing forest-health problems due to disruption of their structure, particularly by logging, livestock grazing, and unusually high-severity fires (but see Baker and Ehle 2003), accompanied by continued suppression of low-severity fires. We note at this juncture that “characteristic” is a sometimes controversial concept among ecologists, even within this group of authors. Characteristic means the fire regime and associated structural characteristics that were apparently most common for a particular forest type historically, before significant disruption of the fire regime by EuroAmericans. Conversely, the term “uncharacteristic” refers to disturbances, forest structure, or fuel loads of a scale or type outside the historical range of variability based on site-specific vegetation reconstructions using tree rings, fire scars, pollen, charcoal, or early historical records.

Characteristic does not imply uniformity, because significant variability occurs within as well as among forest types in fire regime and structure. Hence, a forest type that was “historically characterized” by low-severity fire may, on many sites, can experience high or moderate severity fire (Baker and Ehle 2001). What is characteristic also changes over time, sometimes radically, for example with changing climate (Pierce et al. 2004). Despite difficulties with the concept, without some idea of the predominant or characteristic fire regime for a given forest type, little direction is provided for management strategies. Management then becomes highly laissez faire, with any kind of fire management considered acceptable and with no target or direction for restoration. Hence, we use the terms “characteristic” and “uncharacteristic” intentionally but with the caveats that they do not imply uniformity within forest types and that there are almost always exceptions to the characteristic pattern and process of a type.

In many forests, key structural elements (e.g., old "veteran" trees), terrestrial and aquatic biodiversity, and habitat for many threatened and endangered species are already substantially diminished and at continuing risk of loss. Hence, major public controversies have emerged over appropriate wildfire and fuel policies, especially on the public lands of the West. One recent response has been the Healthy Forests Restoration Act (HFRA) of 2003 (Public Law 108-V148), a law that has potentially wide-ranging consequences for forests and their biodiversity and, therefore, must be implemented on the basis of the best available scientific information and guidance.

Forest and fire policy must be scientifically defensible. Scientists do, and policy-makers should, recognize a diversity of ecological conditions and fire regimes among western North American forests as well as the presence of variability across the geographic ranges of widespread types, such as ponderosa pine (*Pinus ponderosa*) forest. Management policies also must explicitly recognize that forest landscapes are mosaics of different forest and non-forest (e.g., meadow) communities, each with its distinct fire regime and important ecological values.

In this paper we review the scientific literature that is relevant to conservation, restoration, and management of forests, including those characterized by either low-severity stand-maintenance or high-severity stand-replacement fire regimes. Our focus is on the forests of western North America, especially the United States (excluding Alaska). Our review addresses the ecological science relevant to developing and implementing forest fire and fuel management policies, including activities conducted before, during, and after occurrence of wildfires. Our focus is primarily on wildlands, rather than on the wildland-urban interface where ecological values may be secondary to fire-risk mitigation to protect people and their homes (DellaSala et al. 2004). Our purpose is to inform dialogue on the development of policies and management practices related to control and use of fire on public forestlands and appropriate management practices following such disturbances.

Although our goal is an objective review of scientific information relevant to policies and management practices, we suggest that ecologically sustainable practices should be based upon such principles as:

- Broad ecological objectives must be incorporated into definitions of desired future conditions;
- Restoration projects require a series of treatments. Restoration is generally not going to be achieved by a single treatment. Current conditions are the consequence of many decades and, often, multiple human intercessions; hence, restoration will require similar time and effort.
- Creation of restored forests does not necessarily require intensive and costly continuing treatments, if there is sustained commitment to science-based management.
- Restoration programs must be planned and implemented at a landscape level, even though prescriptions are developed and implemented at the stand (local) level. Furthermore, landscape-level planning and implementation is often the only way to reconcile conflicting objectives, such as the restoration of forest structure and protection of imperiled species habitat (Noss et al. 2006)
- Scientific approaches and adaptive management must guide restoration strategies. Untested approaches should be applied to small areas (i.e., individual stands) to identify problems (e.g., an increase in invasive species) and validate the approach scientifically before being widely applied. Credible monitoring programs should be associated with any landscape-level application.

- Postfire (salvage) logging and reforestation policies following severe (stand-replacement) wildfire should take into account the high biodiversity and ecological values of early-successional post-fire habitat, because such practices compromise the recovery and integrity of post-fire landscapes. Similar concerns may exist following uncharacteristic stand-replacement wildfire.
- Aquatic ecosystems, including the smallest streams, are of overwhelming ecological importance in planning restoration and post-fire treatments because of their exceptional sensitivity to many management activities.

FOREST TYPES AND FIRE REGIMES OF THE WESTERN UNITED STATES

Western forests assumed their present tree composition and structure after the end of the Pleistocene. Fire shaped these forests well before that, however, with fire frequency and severity varying with climate over millennia (Millspaugh et al. 2000, Pierce et al. 2004). A close linkage between climate and fire occurs for several reasons. Most western forest landscapes have sufficient fuels to readily ignite and sustain a fire. Lightning may increase toward higher elevations in the Southwest, but is common throughout much of the West (Reap 1986). The region is subject to annual dry weather, often with associated strong winds and significant droughts every few decades (Veblen et al. 2003). Fire has thus been a periodic disturbance, and most western trees are adapted to either survive fire or re-colonize after fire (Miller 2000). Fire provides important services in western forests, such as recycling nutrients, regulating the density and composition of young trees, creating and shaping wildlife and fish habitat, while also structuring the spatial pattern of landscapes and influencing water and sediment delivery in entire watersheds. Many plant and animal species are adapted to post-fire habitats, and populations of some of them (e.g., certain woodpeckers and other birds) decline after fire exclusion or post-fire logging (Hutto 1995). Importantly, different species benefit from different fire severities and times since fire, which suggests that on a regional scale, a variable fire regime is common under natural conditions and that managers should allow for a range of fire severities (Smucker et al. 2005).

Given fuel, lightning, and dry weather, wildfire is inevitable. Nevertheless, the characteristic fire regime, especially the extent and severity of the fire and how often it burns, varies over a surprisingly large spectrum. Fires may recur in western forests from once a decade or less in some dry ponderosa pine forests to 250 to 400 years or more in coastal forests (Hemstrom and Franklin 1982, Martin 1982, Covington et al. 1997, Agee 1998b). Even within a single forest type, fire frequency may vary up to two orders of magnitude across sites, for example from 6 to 600 years in coastal redwood (*Sequoia sempervirens*) forests (Noss 2000). More frequent fires typically are lower in intensity (energy output) and severity (impact on vegetation) than high-severity fires, which are usually infrequent, such as the 1988 Yellowstone fires. Major western forest types can be characterized by a historical dominance of high-severity fire, mixed-severity fire, or low-severity fire (Table 1), which are treated in the following sections.

These characteristic fire regimes result from a combination of forest type, fuels, topography, climate, and ignition sources (Schoennagel et al. 2004a). High fuel loads, high tree density, ladder fuels, and large dead wood are characteristic of many high-severity fire regimes typical of productive or high-elevation temperate forests (Agee 1993). However, these abundant fuels rarely dry out fully, because the climate is typically moist, except during unusual droughts, when fire intensity and spread can be exceptional. In contrast, dry, less productive forests may have annual droughts that lead to more fire. As a consequence of higher fire frequencies, fuel accumulations and fire intensities were characteristically lower in such forests than in coastal or subalpine forests.

Using Plant Associations to Identify Characteristic Fire Regimes

Characteristic forest fuel and fire regimes differ markedly among forest types and regions—as well as within major forest types, such as the ponderosa pine forests that are widespread across western North America (Covington and Moore 1994, Baker et al., in press). Recognition of these differences is an essential first principle in development of any science-based fire management policy (Franklin and Agee 2003). The diversity of forest types and fire regimes must be acknowledged and accepted as a natural imperative if fuel and fire management programs are to be scientifically credible and ecologically sound. Otherwise, unintended consequences may ensue.

Plant associations provide an existing scientific basis for recognizing forest communities that have predictable relationships to fuels and fire regimes, as well as to other attributes that are important to resource management, such as productivity. Plant associations are recognized through the systematic study and classification of forest communities based upon such characteristics as plant species composition (including both trees and shrubs and herbs found in the understory) and environmental features, including soils. Formal studies of plant associations in western North America were undertaken by academic scientists over 50 years ago (e.g., Daubenmire 1952). Research on plant associations expanded rapidly in the 1970s when federal land management agencies began employing large numbers of plant ecologists to develop comprehensive plant association classifications for their lands. Studies of this type were also undertaken by many states.

The forests of the western United States are now comprehensively covered by scientifically based plant association classifications. Publications, including field guides, identify and characterize the plant associations. Many include key attributes for recognizing each plant association in the field, such as by diagnostic plant species and forest structures. Plant associations have been mapped in some land management units, such as some national parks and forests. The majority of resource management personnel have been trained in the recognition and application of plant associations. There are literally hundreds of relevant published reports, but we only cite a few to indicate the diversity of geographic regions and land ownerships for which scientifically-based plant association classifications exist: forests of eastern Washington and northern Idaho (Daubenmire and Daubenmire 1968); ponderosa pine forests of northern Arizona (Hanks et al. 1983); forests of eastern Idaho and western Montana (Steele et al. 1983); Rocky Mountain plant communities (Alexander 1985); Colville Indian Reservation, Washington (Clausnitzer and Zamora 1987); Willamette National Forest, Oregon (Hemstrom et al. 1987); Black Hills National Forest, South Dakota and Wyoming (Hoffman and Alexander 1987); Custer National Forest, Montana and South Dakota (Hansen and Hoffman 1988); Mount Rainier

National Park, Washington (Franklin et al.1988); Olympic National Forest, Washington (Henderson et al. 1989); and Alaska plant communities (Viereck et al. 1992).

In developing policies and generalized forest practices it is generally useful to group together plant associations that have similar attributes; these are known as "plant association groups" or PAGs. Creation of PAGs is necessary because literally hundreds of plant associations have been identified in the western United States. While these differ in various attributes and management potential, it is not necessary to deal with each plant association individually. Rather, PAGs can be created, which can be used as a basis for policy development and management prescriptions related to fire that recognize the diversity present in the forests of the western United States.

Nevertheless, it is important to note that there is, in fact, a continuum of forest types and characteristic fire regimes. That is, fire regimes tend to vary continuously in space and time, rather than falling into a few discrete categories, although such categories may be useful for recognizing general patterns (such as the low-, mixed-, and high-severity fire regimes used in this report). Characteristic fire regimes will typically change gradually with such factors as elevation and location along a precipitation gradient. Often these changes will be reflected in shifting probabilities of specific types of fire behaviors. Similarly, characteristic fire regimes shift over time with climatic cycles that extend over decades, centuries, and millennia; it is often useful to recognize the possibility that past fire regimes almost certainly differed from the current regime and may change again—and, in fact, appear to be changing now with global warming (McKenzie et al. 2004, Pierce et al. 2004). Despite evidence of long-term changes in vegetation with climatic change, PAGs remain a useful tool for planning scales of decades.

<u>High-severity fire regimes</u>

Infrequent high-severity fire regimes are characteristic of subalpine forests found at higher elevations throughout the West and of the moist and relatively productive forests generally found in marine-influenced regions of the Pacific Northwest, including portions of northern Idaho and western Montana (Table 1). High-severity fires kill most or all trees in major portions of the burn, although fire pattern is often a mosaic that includes some areas of unburned forest and of low- to moderate-severity burn. This seems to be true in such diverse forests as the highly productive, coastal forests dominated by Douglas-fir (*Pseudotsuga menziesii*) (e.g., Agee 1993) and low productivity, high-elevation lodgepole pine (*Pinus contorta*) forests in the Rocky Mountains (e.g., Turner et al. 1994, Wallace 2004).

Stand-replacement fires occur predominantly as crown fires, which travel through the tree tops, although intense surface fires can kill trees by heating the roots and lower stem. Forests subject to high-severity fire regimes typically support high tree densities, so when these dense fuels dry out and an ignition source is present the resulting fires can spread rapidly and are essentially impossible to suppress. Many large, high-severity fires probably are associated with infrequent, severe drought events (Agee and Flewelling 1983, Kipfmuller and Swetnam 2000, Westerling et al. 2003), often in association with broad-scale climatic anomalies (Johnson and Wowchuk 1993, Gedalof et al. 2005, Schoennagel et al. 2005). However, many of the large, historic fires of the early 20th century (e.g., Yacolt and Tillamook Burns) were the result of short-term weather patterns involving relatively short periods of hot, dry east winds and human ignitions (e.g., smoldering fires associated with land-clearing activities and logging activities), so such short-term patterns may also have been important during pre-western-settlement times.

Forested landscapes subject to high-severity fire regimes are typically subject to large but infrequent fire events. However, these landscapes are also characterized by more numerous but smaller high-severity fires, which tend to occur under less extreme weather conditions. Examples are some subalpine and coastal forest landscapes (Renkin and Despain 1992, Agee 1993, Bessie and Johnson 1995).

Some coastal forests, such as Sitka spruce (*Picea sitchensis*)-western hemlock (*Tsuga mertensiana*) in the coastal ranges of Oregon and Washington and on the western Olympic Peninsula, and very high-elevation inland forests have rarely experienced fire. In these forests, wind replaces fire as the stand-replacement disturbance (Henderson et al. 1989, Franklin and Halpern 2000). Forests from which wildfires are essentially absent become even more common to the north in parts of coastal British Columbia (Lertzman et al. 2002) and southeast Alaska. Hence, not all western North American forests with high fuel loadings are characteristically subject to wildfire.

In contrast to climate, the effect of relative fuel abundance on high-severity fire regimes is ambiguous. Many forests characteristically experiencing high-severity fire are relatively productive; hence fuel loads are high. Under presettlement conditions this was true of both young and old forests, due to high levels of biological legacies (i.e., snags and logs) and relatively high tree densities (Spies et al. 1988). This is certainly the case in the Pacific Northwest and in portions of northern Idaho and western Montana.

Subalpine forests may not follow the model of uniformly high fuel loadings throughout succession as closely as more productive forests (Romme 1982). On the subalpine plateaus of Yellowstone National Park, comprised of lodgepole pine forests over relatively flat terrain, fuels related to stand age and density had no influence on fire severity during extreme weather conditions. Nevertheless, during less extreme climate conditions older stands appear more likely to burn (Renkin and Despain 1992, Turner et al. 1994). In the boreal forest, extreme climate also appeared to override the importance of variability in fuel loads related to stand age and composition (Bessie and Johnson 1995). In more heterogeneous subalpine landscapes of Colorado, however, forest species composition (aspen [*Populus tremuloides*] vs spruce [*Picea engelmannii*]-fir [*Abies lasiocarpa*]) has a significant influence on the spatial pattern of fire severity even under extreme weather conditions (Bigler et al. 2005.

Fire exclusion has had little to no effect on fuels in forests characterized by high-severity fire regimes – a fact that is especially relevant to fire policy. Many of these forests are remote and at high elevation, making access for fire-fighting difficult. Furthermore, high-severity fires are relatively infrequent (from one to many centuries; Hemstrom and Franklin 1982, Romme and Despain 1989, Agee 1993, Kipfmueller and Baker 2000, Veblen 2000), while the period of active fire exclusion in these remote forests is 50 years at most. Therefore, the length of the fire exclusion period is short relative to the expected fire-free interval. As a consequence, exclusion has not significantly lengthened fire intervals, which is in marked contrast to low-severity fire regimes (see below). Furthermore, biomass (fuel) is always high in productive forests (Franklin and Dyrness 1973), and there is no consistent relationship between time since the last fire and fuel abundance, even if fire frequency had been affected (Brown and Bevins 1986, Odion et al. 2004). Lastly, large high-severity fires occurring under extreme weather conditions are very difficult, if not impossible, to suppress once underway (Wakimoto 1989).

Fire exclusion may have had some effects on forests characterized by high-severity fire regimes, especially on a landscape scale. For example, it is possible that the size of high-severity fires has been reduced over the last few decades by fire exclusion and habitat fragmentation. It is also possible that fire suppression has eliminated the ignition sources for some high-severity fires that a smoldering fire might have provided. Nevertheless, neither fuel conditions nor fire-return intervals are outside the historical range of variation in forests characterized by high-severity fire. Furthermore, exclusion is not likely to be able to contain all intense fires in regions subject to stand-replacement fire regimes; hence, a continued regime of intense stand-replacement fire can be expected, particularly under extreme weather or drought conditions – circumstances that are expected to increase under some climate change scenarios, which predict generally longer fire seasons (McKenzie et al. 2004).

There is no evidence to suggest that any restoration of fuel structure or tree density is appropriate in forests subject to high-severity fire regimes because fire exclusion has had a minimal impact on the stand-level structure of such forests (Franklin and Agee 2003, Brown et al. 2004, Schoennagel et al. 2004a). Tree densities and fuel levels in these forests are either within the historical range of variability or, in the case of harvested stands, below characteristic levels as a result of logging and associated slash treatments. Indeed, reducing stand densities and fuels to levels that would significantly decrease the potential for stand-replacement fire would create forests incapable of fulfilling their ecological roles, including provision of wildlife habitat.

Consequently, neither prescribed fire nor mechanical thinning is warranted in wildland forests characterized by a high-severity fire regime, considering their characteristic fire behavior, stand structure, and biodiversity. Furthermore, it is doubtful whether such moderate fuel reductions would be effective in reducing fire severity under the extreme climatic conditions that typically trigger high-severity fires in these subalpine and productive coastal forests. On the other hand, fire suppression programs may be appropriate in some high-severity areas to protect less common old-growth and young post-fire stages of forest development, as well as in the wildland-urban interface (DellaSala et al. 2004).

Mixed-severity fire regimes

Fire is quite variable in intensity and severity in certain mid-elevation forests of moderate to high productivity across variable topography (Table 1). In these forests, both low- and high-severity fires may occur with the former probably more frequent than the latter. Furthermore, in the mixed-severity fire regimes individual fires may be variable in severity. For example, during the 2002 Hayman fire in a ponderosa pine-Douglas-fir landscape in Colorado, about 25,000 ha burned at high severity on a single day during exceptionally dry and windy conditions, but when the winds subsided and fuel moisture increased, the fire burned at low severity (Finney et al. 2003).

In addition to effects of shifts in weather during a fire, variable fire severity can result from spatial heterogeneity in topography, which affects microclimate, the relative abundance of fuels, and the legacies of past episodes of fire and other natural disturbances. Topographically complex western mountain landscapes at mid- to low-elevations may be especially prone to mixed-severity fire, because drier south-facing slopes with lower fuel loads may burn at low-severity while adjacent moister north-facing slopes that support higher tree densities may experience high-severity fire (Taylor and Skinner 2003, Spies et al. 2006) or, alternately, escape fire due to their moister conditions. Patches burned at high-severity in the past may tend to perpetuate themselves by characteristically regenerating to young, dense forests that provide places where a subsequent low-severity fire can again climb into the forest canopy, perpetuating a patchy landscape mosaic (Baker et al., in press). Patchy accumulations of fuels may lead to a finer-scale patchiness as fires run along the surface, torching individual trees and groups of trees (Taylor and Skinner 1998), but mixed-severity regimes have moderate to large patches of high-severity fire as well (Veblen and Lorenz 1991, Brown et al. 1999).

Due to the complexity of the mixed-severity fire regime, little is known about characteristic spatial and temporal variation in patch sizes, which are partially stochastic and partially predictable based on topography (Agee 1998b). Ecologists have only recently begun to characterize quantitatively this mixed-severity fire regime by combining evidence of stand-replacement fire, which is harder to reconstruct, with evidence of surface fires. In the past, some sites experiencing mixed-severity fire regimes may have been mischaracterized as low-severity fire regimes, because only surface-fire evidence was assembled. Recently, there has been an increase in the number of studies recording mixed-severity fire in moister ponderosa pine forests (Shinneman and Baker 1997, Brown et al. 1999, Kaufmann et al. 2000, Ehle and Baker 2003, Sherriff and Veblen in review) as well as in mixed evergreen-conifer forests (Odion et al. 2004). Lack of sufficient landscape-scale data in many places makes it challenging to determine whether or not a mix of fire severities is characteristic.

The inherent variability of mixed-severity fire regimes precludes easy detection and analysis of the effects of fire exclusion. Exclusion of fire since EuroAmerican settlement may have allowed tree densities to increase in some areas (e.g., Parsons and DeBenedetti 1979). Yet, post-fire tree density was characteristically high in patches killed by high-severity fire (Taylor and Skinner 1998, Baker et al., in press). We lack adequate data on stand structure and development following fires of different severity under the mixed-severity fire regime. Consequently, one cannot assume that high tree density or an abundance of shade-tolerant trees on mixed-severity sites is the result of fire exclusion. On a landscape scale, an effect of fire exclusion in this fire regime might be to create a more homogeneous, less patchy structure (Taylor and Skinner 1998, Odion et al. 2004). However, landscapes subject to this fire regime may go through periods of relative homogeneity, associated with infrequent episodes of higher-severity fire, which alternate with periods of high heterogeneity during intervals with smaller, less severe fires (Baker et al., in press).

Restoration of mixed-severity landscapes thought to be affected by fire exclusion thus is exceedingly complex and uncertain at the present time. Where sufficient fire-history information is available, a combination of thinning and prescribed fire might be used in restoration, yet only parts of these landscapes may warrant treatment, because spatial variability is a hallmark of these landscapes. Hence a landscape- rather than a stand-level approach to restoration is warranted, especially when considering that many species (e.g., birds) depend on diversity in fire severity and post-fire conditions across the landscape (Smucker et al. 2005). Prescribed fires or thinning that do not match the scale or pattern of the historical fire regime or target the right patches may not lead to restoration, but simply to further alteration (Baker 1993, Taylor 2000). Low-severity prescribed fires will not restore the mosaic of patches created by high-severity fire that characterized these forest landscapes, yet managers may be averse to burning when conditions might lead to some high-severity fire. These fire regimes are inherently complex and therefore it is difficult to define and implement appropriate restoration.

Low-severity fire regimes

Low-severity fire regimes characterized many pine and mixed-conifer forests that were found on warm, dry sites prior to European settlement (Table 1). Woodlands of juniper (*Juniperus* spp.), pine, and hardwoods often bounded these forests at lower elevations and forests with mixed- and high-severity fire regimes at higher elevations. Fires in these ponderosa pine and dry mixed-conifer forests historically burned fine fuels (e.g., grasses and litter on the forest floor) at regular intervals (Swetnam and Baisan 1996, Veblen et al. 2000, Heyerdahl et al. 2001, Stephens and Collins 2004). These surface fires rarely killed large fire-resistant trees but killed smaller trees of all species, thereby helping to maintain open-canopied stands.

Forests that characteristically experienced this fire regime include the majority of dry ponderosa pine (or, sometimes, Jeffrey pine [*Pinus jeffreyi*) and dry mixed-conifer forests that occur on the east slopes of the Cascade Range in Oregon and Washington and are widespread in the Sierra Nevada, the intermountain region, and the southwest. However, some dry western forests are exceptions and naturally experience mixed- or high-severity regimes, including some ponderosa pine forests around the West (e.g., Shinneman and Baker 1997) and many gray pine (*Pinus sabiniana*) forests in the Sierra Nevada of California. A reassessment in the Rocky Mountains suggests that a small fraction (i.e., < 20%) of ponderosa pine-Douglas-fir forests—particularly those on the driest sites at lowest elevations—fit the model of a typical low-severity fire regime (Sherriff and Veblen, in review; Baker et al., in press).

Human activities following European settlement—including fire exclusion, grazing, logging, and tree planting –dramatically modified the fuel structure in forests that were characteristically subject to low-severity fire. The high level of modification is the result of many factors including: accessible locations of many of these forests, their vulnerability to modification (often due to relatively low productivity), and the relative ease of extinguishing the surface fires. Furthermore, diverse activities have often interacted synergistically, as noted below.

Organized fire suppression programs have been underway since the early 1900s in many western forests and, as a consequence, the length of fire intervals has increased markedly on sites characterized by low-severity fire (Covington and Moore 1994, Veblen et al. 2000). On the other hand, the contribution of active fire suppression programs to reduced fire frequencies has probably been over-exaggerated in the popular literature.

Grazing by livestock has been a significant factor affecting forests for longer than fire suppression throughout the western United States. Grazing reduces the fine (grassy) ground fuels that carry surface fires and also indirectly facilitates establishment of dense tree reproduction by reducing or eliminating herbaceous competition (Pearson 1942, Rummell 1951, Madany and West 1983, Savage and Swetnam 1990, Baisan and Swetnam 1997, Belsky and Blumenthal 1997). Hence, fire suppression and grazing both contributed to an increase in stand densities in many western forests, notably the southwestern ponderosa pine forests in Arizona and New Mexico (Covington and Moore 1994, Friederici 2003). It is clear that forest restoration programs need to pay greater attention to the effects of livestock grazing on fire regimes and regeneration conditions.

Logging of large fire-resistant trees using either selective cutting or clear cutting methods has eliminated key ecological elements of forests that were historically maintained under low-severity fire regimes. This means that structural habitat elements (large trees, snags, and logs) that are essential to many ecological functions, such as fish and wildlife habitat, are no longer present on many sites. Logging also has promoted higher stand densities in many dry ponderosa pine forests by stimulating dense natural regeneration (Agee 1993, Smith and Arno 1999, Kaufmann et al. 2000) or by tree planting with the intent to increase wood production through dense, even-aged stands. Such logging and reforestation programs have dramatically altered the fuel structure of these forests, moving them from relatively open stands with low fuel loadings and limited ladder fuels to dense stands that can develop and carry crown fires (Skinner 1995). Tree plantations are particularly likely to burn at high severity. When high-severity fires occur in forest types usually characterized by low-severity fire, the suite of birds and other species that are post-fire-habitat specialists in other parts of the West may fail to colonize these areas (Saab and Powell 2005). Thus, unexpectedly low bird diversity may occur in some post-fire forests.

Restoration of dry ponderosa pine and dry mixed-conifer forests, where low-severity fires are characteristic, is often appropriate and desirable (McKelvey et al. 1996, DellaSala et al. 1998, Allen et al. 2002, Franklin and Agee 2003, Friederici 2003, Brown et al. 2004, Schoennagel et al. 2004a). Mechanical thinning and prescribed fire have been shown to be effective means of reducing stand densities to their level prior to the influences of fire exclusion, grazing, or logging (Covington et al. 1997, Moore et al. 1999). However, such management alone may be insufficient to restore these forests, if other key components of these ecosystems (e.g., native understory plant composition) remain compromised by historical and on-going land uses, such as grazing and logging.

Evidence from recent wildfires indicates that thinning of smaller trees with removal of slash (small trees, branches, and tree tops) or prescribed fire can effectively reduce fire severity in dry ponderosa pine forests (Schoennagel et al. 2004a, Agee and Skinner 2005). This suggests that, in contrast to the high-severity fire regime, even under extreme climate conditions fuel abundance and configuration can effectively influence fire behavior in forests that characteristically experienced low-severity fires (Agee 1997, Perry et al. 2004). Nevertheless, not all thinning prescriptions are effective in reducing fire severity (Brown et al. 2004, Agee and Skinner 2005, Stephens and Moghaddas 2005). Empirical research is sparse on the type, degree, frequency, scale, and pattern of thinning and prescribed fire for effective return of low-severity fire to these forests (Rhodes and Odion 2004). Furthermore, returning low-severity fire to these forests may hinge on restoring understory plant communities that can carry surface fires and compete with tree seedlings, thereby helping to maintain more open and species-rich stands. Forest understories are intrinsically important, because much of the forests' plant diversity is found there. Plant diversity in the understory declines in many forests as forest canopies close due to logging and grazing, accompanied by the loss of low-severity fires (e.g., Covington and Moore 1994).

RESTORATION OF FORESTS WITH ALTERED FIRE REGIMES AND STRUCTURES

Various aspects of forest restoration are considered in this section, which begins with a view of forest restoration as a continuum of possibilities.

Principles of Forest Restoration

Forest restoration may vary along a continuum from restoring structure, such as density of large trees, to restoring the processes (e.g., low-severity fire, grass competition) that create and maintain that structure (Figure 1). The continuum also represents a gradient from symptoms (e.g., uncharacteristic tree density) to causes (e.g., exclusion of fire). It is a well-established principle in land health, as in human health, that treating symptoms may be necessary, but to restore health requires identifying and treating causes.

Ecological restoration invariably begins on the left side of the continuum (Figure 1), but if restoration focuses only on symptoms, periodic and costly re-treatments will inevitably be necessary. Therefore, identifying and reforming the causes that led to the need for restoration is the essential first step in any ecological restoration project (Hobbs and Norton 1996, DellaSala et al. 2004). For example, consider a ponderosa pine forest that currently is uncharacteristically dense due to overgrazing by livestock; if this stand is thinned but the fire regime and native understory are not restored, dense tree regeneration is likely to develop, leading to the need for another thinning treatment in a potentially endless cycle. The density of young trees in ponderosa pine forests is held in check, in part, by competition from native bunchgrasses. Restoration of native understory shrubs, forbs, and particularly bunchgrasses would allow competition to aid in cementing the restored structure. If livestock grazing is maintained at levels that are too high to allow the restored understory to persist, periodic restoration of the understory may be needed. Restoration of the processes that regulate structure is more likely to lead to a low-maintenance system that is more sustainable, requiring little or no costly re-treatment.

The restoration continuum also represents a gradient from high human inputs on the left to more maintenance by nature on the right. This is useful to consider because our understanding of how to manage ecosystems remains imperfect. We do not know the details of how to manage fire in a way that will fully maintain ecosystem processes and services, nor is it common practice to focus on this goal. Prescribed fire, for example, often differs from unmanaged fire in pattern, severity, or effects. Prescribed fires are typically planned for safe periods when fuel moisture is relatively high, and often aim to simply reduce fuel loads—i.e., they are "cool" fires and often leave a more homogeneous pattern compared to the heterogeneous mosaic produced by most natural fires (e.g., Allen et al. 2002, Turner et al. 2003). Prescribed fires can actually exacerbate past effects of fire exclusion, rather than restore landscapes, if they are not done in the right place at the right time (Baker 1993). Restoring the natural fire regime to operate without human intervention, where possible, is most likely to lead to sustainable and effective ecological restoration (Noss et al. 2006).

Appropriate models for restoration will undoubtedly vary with current forest conditions, management objectives, and plant association group, among other factors. An essential early step is identifying the Desired Future Condition (DFC) or the target of the trajectory of change of a management program. DFCs are often based on conditions that are considered to be within the historical range of variability (HRV; Landres et al. 1999, Veblen et al. 2003). Precisely achieving some past condition is likely to be difficult, but conditions broadly representative of the HRV can often be approximated or partially restored. Restoration of processes (e.g., low-severity fire) may allow the re-structured forest to eventually equilibrate with contemporary natural conditions.

The identified restoration model, trajectory of change, or DFC may represent a significant departure from the historical range of variability in some cases or even represent a unique condition. Restoring a fire regime that resembles the pre-European settlement regime may not be possible in densely settled regions or in the wildland-urban interface (Marzluff and Bradley 2003, DellaSala et al. 2004, Radeloff et al. 2005). Understory species composition or structure may have changed as a result of invasive species. The DFC may include elements that were not characteristic historically, such as higher populations of old-growth trees or abundant forage production for wildlife in the understory. The model for restoration in these cases may need to be built from considerations of how fire fits into the contemporary landscape and how it can be used to meet specific goals, such as satisfying the needs of imperiled native species (Engstrom et al. 2005, Purcell and Stephens 2005). Even in these cases, a review of historical conditions is an important starting point for restoration planning, as such conditions are broadly representative of the evolutionary environment of the species native to a forest type (Covington 2003). Of course, any DFC or restoration model needs to be sustainable under the fire regime that is characteristic for the site.

Two common restoration situations recur in dry ponderosa pine and mixed-conifer forests in the West. The prevalent restoration situation occurs in landscapes dominated by post-logging, grazed, or post-fire forests on sites that were characteristically subject to low-severity fire regimes. These landscapes are currently dominated by dense, middle-aged (i.e., 50-150 years) forests, which often have been commercially thinned or otherwise treated. Prior to logging they were old-growth forests, but currently they have few or no remnant pre-EuroAmerican trees. A restoration focus based on HRV is facilitating redevelopment of old-growth structure in these landscapes (Covington et al. 1997, Allen et al. 2002). The risk of loss of individual stands to high-severity fire is not high, because this stand structure is common in the West.

The less common situation occurs where remnant old-growth trees persist on sites characterized by low-severity fire regimes but with a significant cohort of small post-suppression trees. Here both risks and value are higher than in the predominant situation. If HRV is the model, restoration may focus on protecting and perpetuating the pre-EuroAmerican trees. This is done by lowering the density of post-settlement trees, particularly around the remnant old-growth trees, thereby reducing competition and potential fuel ladders (Spies et al. 2006). *Old-growth* *trees and the habitat they provide require centuries to replace, and it is thus reasonable to give such trees and other scarce habitats special attention as a part of restoration efforts, even if it requires departures from the HRV to perpetuate them.*

Indeed, level of threat to particular natural values—such as critical wildlife habitat, watershed and aquatic values, and existing populations of veteran old trees—is a key issue in setting priorities for restoration treatments and in determining whether HRV or other restoration goals may be most appropriate. Departure from historical fire frequencies may not be the best way to index existing threats to natural values, even though this method of prioritization is specified in the Healthy Forests Restoration Act of 2003. For example, in some parts of the western United States sites belonging to the ponderosa pine PAGs may be out of synchrony with their natural fire cycle. However, other sites on the same landscapes belonging to the dry mixed-conifer PAGs, may actually have larger fuel accumulations and be more at risk of uncharacteristic fires, even though they may have missed fewer burn cycles. This is because the dry mixed-conifer forest sites have much higher productivity and include shade-tolerant tree species, such as grand fir (*Abies grandis*) or white fir (*Abies concolor*), that provide extraordinarily effective ladder fuels (i.e., branches and other fuels that can carry fire up into the forest canopy).

On public lands, restoration programs must consider impacts on ecological values associated with terrestrial and aquatic ecosystems, not just effects on forest fuels and potential fire behavior (Friederici 2003). Ecological restoration involves much more than fuels reduction and other narrow treatments, and should be carefully designed to minimize impacts to sensitive biotic and abiotic components of the ecosystem (e.g., herbaceous plants, soils). Urban fringe areas may be an exception to this rule in that concerns about high fuel loads and potentially extreme fire behavior may overwhelm consideration of negative ecological impacts of treatments (Covington 2000, Marzluff and Bradley 2003, Radeloff et al. 2005, Prather et al. in press).

Case Studies of Restoration

Forest types and regions differ widely in the degree to which they have been altered from their characteristic structure and composition and, therefore, in the appropriate steps (treatments) that are needed to achieve the restored (desired) condition. In this section we provide two examples to illustrate the various restoration treatments that may be needed.

Klamath Reservation Forest Restoration

The strategy for restoration of the Klamath Reservation Forest was developed for the Klamath Tribes as part of their effort to regain tribal lands purchased by the federal government in 1954 and incorporated into the national forest system as portions of the Winema and Fremont National Forests. The plan was guided by a tribal vision of restoring the forests and landscapes to a more characteristic and sustainable condition (Johnson et al. 2003). However, the US Forest Service and other stakeholders are interested in the strategy as a general approach to restoration of forests in eastern Oregon, regardless of land ownership.

Open ponderosa pine forests dominated the forest landscape early in the 20th century based on a detailed inventory and mapping completed in 1921 and confirmed by USDA Forest Service type map produced in 1936. The structure of these forests typically was a complex fine-scale patch mosaic produced by chronic, low severity fires, which is dominated by large, fire-resistant old-growth pine trees (Figure 2) (Franklin and Van Pelt 2004). Extensive selective logging of the ponderosa pine forests began in 1916 and continued under the supervision of the Bureau of Indian Affairs until the reservation was terminated and lands were sold in 1954. Subsequently the lands under consideration were purchased by the United States government and incorporated into portions of the Winema and Fremont National Forests. Extensive timber harvesting continued under Forest Service administration, shifting from single-tree selection based upon bark-beetle risk ratings to even-aged management regimes in all forest types (ponderosa pine, mixed conifer, and lodgepole pine) through the 1980s. Logging coupled with tree planting and fire suppression have resulted in existing forests which, on average, are substantially denser, structurally simpler, and richer in shade-tolerant species than was characteristic of the pre-20th-century landscape. Currently, however, most stands still contain residual old-growth trees.

Under the Klamath Tribes' Economic Self-sufficiency Plan, lands were to be managed under a restoration strategy with the following objectives: 1) restoration of forest stand structure to levels similar to the HRV; 2) reduction of overall fuel levels and continuity to decrease the potential for uncharacteristic stand-replacement fires; 3) restoration of more natural fire regimes; 4) increased carrying capacity for deer, elk, and other favored wildlife and fish species; 5) enhanced spiritual and cultural values; and 6) production of sustained monetary and subsistence income.

Plant association groups and current structural condition classes provide the basic stratification for silvicultural treatments. Primary PAGs recognized in the Klamath forest are Ponderosa Pine, Dry Mixed Conifer, Moist Mixed Conifer, and Lodgepole Pine; in some cases these have been further subdivided. Forests within each PAG or habitat type are classified into one of four current structural conditions: complex, simplified, recovering clearcut, and open. In the case of Ponderosa Pine and Dry Mixed Conifer PAGs, complex forests are those which include a large-diameter tree component, a spatially complex pattern of stand structural units (Figure 2), and a well-developed understory of shrubs and herbs. Simplified forests have a uniformly dense stand structure of small to medium-sized trees with few or no large- diameter trees and uncharacteristically high fuel loadings, reflecting the combined consequences of logging and fire suppression. Priority areas identified for treatment are forests with a residual old-growth pine component that is at risk of loss to either uncharacteristically intense fires (due to fuel accumulations) or bark beetle attack (due to stress from competing vegetation).

Silvicultural goals for forests on sites belonging to the Ponderosa Pine and Dry Mixed-Conifer PAGs include: 1) retention of all existing old-growth pine trees and, ultimately, restoration and maintenance of a characteristic old-growth population structure; 2) restoration of stand densities and fuel loadings to levels that will significantly reduce the potential for stand-replacement fire and excessive loss of old-growth trees to competition; and 3) management of young and mature components of the stand to provide for old-growth tree replacements. In this restoration strategy, all dead and dying old-growth trees are retained along with all live old-growth trees in order to provide snags and logs that are characteristic of old-growth stands on these sites. This approach is consistent with the old-growth direction provided in the Healthy Forest Restoration Act: "fully maintain, or contribute toward restoration of, the structure and composition of old-growth stands according to the pre-fire suppression old-growth conditions characteristic of the forest type. . .

and retaining the large trees contributing to old-growth structure" (Healthy Forests Restoration Act of 2003).

The Klamath restoration plan proposes to consider and treat entire large (e.g., 5,000-10,000 ha) landscape units; not all portions of each landscape will actually be treated, but each will be planned as a complete entity. Northern spotted owls (*Strix caurina occidentalis*) are present in some portions of the Klamath forest proposed for restoration, and the plan incorporates a landscape-level component to provide for maintenance of populations of the owls and their primary prey species. Dense mixed-conifer forests (with crown closure >60%) with a significant component of white or grand fir and/or Douglas-fir are needed as nesting, roosting, and foraging habitat for northern spotted owls where they occur in dry landscapes on the eastside of the Cascade Range. Such forests are at high risk of loss to uncharacteristic stand-replacement wildfire; for example, on the Deschutes National Forest 18 of 24 northern spotted owl home ranges (all in Late Successional Reserves) have been lost to recent wildfires. Although such losses have not yet occurred in owl habitat in the Klamath restoration landscape, the risk of habitat loss is very high.

The restoration strategy will attempt to conserve owl habitat by embedding islands of denser forest within a landscape that has otherwise been treated to reduce fuel loadings. Embedding owl habitat in a landscape that has been treated to significantly reduce the potential for large scale, stand-replacement wildfire should dramatically increase the potential for owl habitat persistence in these fire-prone landscapes. A similar strategy has been proposed to reconcile conflicts between conservation of Mexican spotted owls (*S. o. lucida*) and restoration of ponderosa pine landscapes in the Southwest (Beier and Maschinski 2003, Noss et al. 2006, Prather et al. in press).

<u>Rocky Mountain Ponderosa Pine-Douglas-fir Forests</u>

Recent research suggests that the fire regime was of mixed-severity in much of the forests composed of ponderosa pine and Douglas-fir in the Rocky Mountains (Baker et al., in press). This fire regime is characterized by large spatial and temporal variation in tree density, ladder fuels, large wood, tree age, and other aspects of forest and fuel structure (Kaufmann et al. 2000, Veblen et al. 2000, Ehle and Baker 2003, Sherriff 2004).Young post-fire patches characteristically have high tree density; reconstructions and estimates document that tree density near A.D. 1900 was often as high as 1,500-3,000 trees/ha and occasionally reached 25,000 trees/ha (Kaufmann et al. 2000, Ehle and Baker 2003, Sherriff 2004, Baker et al. in press). Nearby patches of older forest, however, may have had much lower tree density (< 750 trees/ha), although they were still denser than comparable forests dominated by low-severity fires (Baker et al., in press). Fuels were similarly variable, with low fuels in some areas at some times and high loads after windstorms, insect outbreaks, or other disturbances (Brown and See 1981). Fuels in Rocky Mountain ponderosa pine-Douglas-fir forests are variable, but often low today. For example, loadings for large dead wood range from 3-23 Mg/ha (Baker et al., in press), below levels that lead to high fire hazard (Brown et al. 2003).

Large areas of these Rocky Mountain forests were logged or burned in the late-19th and early 20th century and the trees are now about a century old, while mature or old-growth forests are rare (Veblen and Lorenz 1991). Tree density may have been increased by livestock grazing or logging, but high tree density is characteristic of century-old forests in these landscapes. Thus, at the stand level, tree density in these forests may be within or close to characteristic levels for their stand age and not require much thinning, but other structural elements (e.g., understory shrubs, grasses, forbs) and processes may warrant restoration.

In mixed-severity fire regimes, some high-severity fire is characteristic and should not trigger efforts to create forest structures that would exclusively support low-severity fire. Fuels do not need uniform reduction but management targeted at particular needs in specific places. Fuel-reduction treatments, for example, may be warranted in the wildland-urban interface to protect human life and property, but not within the wildlands (DellaSala et al. 2004). Past land uses are the source of today's forest structure and fuel situation and different use patterns require different restoration actions. Large dead fuels, for example, may be in deficit in areas subject to past logging, while overgrazed forests may have elevated tree densities in some cases. Old patches of dense forest may not require any restoration.

Mixed-severity fire regimes require landscape-level planning and analysis to determine appropriate restoration goals. The impacts of logging and burning are most evident at the landscape level where there is often a deficiency of mature and old-growth forest patches and a surplus of century-old forests. Restoration at the landscape scale should seek to enhance the development of old-growth structure in some, but not all patches, using light tree thinning accompanied by prescribed burning and restoration of other structures. Of course, an essential step, if old-growth is to be truly restored, is to retain all trees that pre-date EuroAmerican settlement, as is recommended in the Southwest (Friederici 2003).

THE ROLE OF RESERVES IN FIRE-PRONE FOREST LANDSCAPES

A reserve is an area where the conservation of biodiversity, ecological integrity, or similar values take precedence over other uses (Scott et al. 1993). The reserve strategy becomes complicated when applied to dynamic landscapes shaped by fire. As for forest management in general, the question of how reserves in fire-prone landscapes should be managed cannot be addressed by application of a "one-size-fits-all" philosophy. Instead, their management must be guided by the vegetation structure and composition of the area in question, the characteristic fire regime, the historic or natural range of variability in both vegetation and fire regime, and specific objectives for the reserved area. In this section we discuss why reserves are important to forest conservation strategies and the special considerations needed for their management.

Why Reserves?

Reserves placed in a well-managed landscape matrix are usually needed to meet conservation goals (Lindenmayer and Franklin 2002). Reserves have values that range from scientific to aesthetic (Noss and Cooperrider 1994, Noss et al. 1997, 1999, Groves 2003, Lindenmayer and Franklin 2002). One well-accepted function of reserves is to represent all forest and other ecosystem types, thereby providing a "coarse filter" of protection to species associated with these ecosystems (Noss 1987). Another critical function of reserves is to offer security to species, biological communities, and habitat elements that are sensitive to impacts from human activities, ranging from hunting to habitat loss, fragmentation, and degradation. Declining species are often at high risk of extinction due to stochastic environmental processes as well (Beissinger and McCullough 2002).

Species closely associated with late-successional forests often may be dependent on reserves, because forests outside reserves are logged too regularly and contain trees too small to meet their needs. The spotted owl and red-cockaded woodpecker (*Picoides borealis*) are well-known examples of such species in the U.S. (Simberloff 1998). Species typical of post-fire habitats with abundant standing dead trees, such as many woodpeckers, may also depend on reserves because areas outside reserves are typically salvage-logged (Hutto 1995, Lindenmayer et al. 2004). The disturbance regime to which the regional biota adapted over their evolutionary histories provided habitat refugia on a variety of spatial scales, from standing dead and downed logs to large patches of old or dense forest in a mosaic of more recently disturbed patches (Lindenmayer and Franklin 2002).

Ecologists have long recognized the importance of large reserves as places where scientists could study species and ecosystems under relatively natural conditions (Croker 1991). This reference value of reserves for management and restoration becomes more significant as more of the Earth's surface is disturbed by humanity. As pointed out by Leopold (1941), wild areas provide a "base datum of normality" for a "science of land health." Reference areas have proved invaluable in setting standards for indices of biotic integrity for streams in various stages of degradation (Karr 1991). Most modern proponents of ecosystem management recognize the value of reserves as comparison areas for management experiments (Christensen et al. 1996, Arcese and Sinclair 1997, Lindenmayer and Franklin 2002). Reserves, especially when large relative to the scale of natural disturbance events (Pickett and Thompson 1978) may perpetuate ecological processes, including fire, within HRV. Reserves are never perfect comparison or control areas for large-scale management experiments because they are usually unreplicated, differ in a variety of ways from treated areas, and have many confounding influences. Nevertheless, they are still crucial for providing insights on the impacts of management. Because logging and other human activities now impact entire landscapes, at least some comparison areas should be this size (Noss 1991, Franklin et al. 2000).

Roadless areas provide many of the functions of designated reserves. For instance, they serve as refugia for terrestrial and aquatic species that are sensitive to management practices, help reduce invasions of non-native species, and provide reference sites over a broader range of conditions than designated reserves. Roadless areas better represent the natural landscapes of the U.S. than do formally protected areas (USFS 2000, DeVelice and Martin 2001, Strittholt and DellaSala 2001, Loucks et al. 2003, Crist et al. 2005). The harmful impacts of roads on sensitive ecosystems and species are well documented (Trombulak and Frissell 2000). Among other problems, roads in forest landscapes promote the invasion of non-native plants (e.g., Tyser and Worley 1992, Gelbard and Belnap 2003), which, in turn, often change fire regimes and make restoration difficult. Roads act as unnatural fire breaks and, conversely, are major sources of human-caused fire ignitions (DellaSala and Frost 2001). Likewise, roads are major sources of human-caused erosion, landslides, and siltation of streams. Roads also provide access to humans, increasing the potential for recreational uses to affect biodiversity adversely, for instance, through hunting and use of off-road vehicles. A precautionary approach to forest management is to protect existing roadless areas, avoid building new roads, and close and revegetate existing forest roads in sensitive areas wherever possible.

Restoration in Reserves

The general concept and definition of reserves does not imply absence of active management or prohibition of all human uses, but does suggest minimizing the impact of management and use. However, active management, such as prescribed burning, and uses such as ecological research and monitoring are often essential to the persistence of populations of native plants and animals, habitat features, and key ecological processes within reserves.

In cases where fully restoring a natural fire regime is infeasible, active management may be necessary to restore and maintain biodiversity and the conditions for which reserves were set aside. The objectives in such cases are to bring the ecosystem back within the HRV and maintain it there with a minimum of additional active management. The National Park Service follows essentially this strategy, which is consistent with its policy to maintain "…natural environments evolving through natural processes minimally influenced by human actions," but in cases where natural conditions have been disrupted, ecosystems in parks "…may be manipulated where necessary to restore natural conditions" (National Park Service 1988, as cited in Stephenson 1999). Toward this end, the National Park Service has conducted reconstructions of presettlement vegetation and performed prescribed burns and other restoration activities in such parks as Sequoia-Kings Canyon, Yosemite, and Redwood National Park (e.g., Parsons et al. 1986, Stephenson 1999, Thornburgh et al. 2000).

Although restoration treatments inevitably carry risks, in some cases the risks to native biodiversity of *not* managing reserves may be greater than risks associated with management. Nevertheless, active management of reserves is controversial. Restoration, especially mechanical thinning, within some reserves (such as designated Wilderness) may be socially or legally unacceptable. Many protected areas are at high elevations (Scott et al. 2001) and contain forest types that burn infrequently and have not been significantly impacted by fire protection programs (Brown et al. 2004, Schoennagel et al. 2004a). In these cases, active restoration of forest structure generally is not needed, so controversy can be avoided. On the other hand, many protected areas have been impacted by past management practices, including fire suppression; in these cases some initial active restoration may be needed.

When restoration treatments are undertaken within reserves they should be of the minimum intensity needed to restore natural conditions and protect old growth trees and imperiled species habitat. Management options within reserves are properly constrained and shaped by the mandates of the enabling legislation and by public opinion. For example, wilderness areas prohibit the use of motorized equipment. Certainly, no new roads should be built within reserves or roadless areas to accommodate restoration. After they are restored, reserves that are sufficiently large may be able to incorporate an unmanaged natural disturbance regime (Pickett and Thompson 1978, Noss and Cooperrider 1994). Because variability in landscape conditions is desirable, and because management inevitably involves mistakes, some sizable proportion of protected areas should be exempt from active management. Prescribed burning and thinning within buffer zones adjacent to reserve boundaries may be advisable in some cases to reduce the probability of undesirable stand-replacing fires occurring inside. An example of where restoration within a reserve benefits a rare species is the Bear Valley National Wildlife Refuge in the eastern Cascades of Oregon. Here, the U.S. Fish and Wildlife Service successfully applied low-density thinning and prescribed fire to restore one of the most important communal roosting areas for bald eagles (*Haliaeetus leucocephalis*) in the lower 48 states (DellaSala et al. 1998).

ECOLOGICAL ISSUES RELATED TO FIRE MANAGEMENT POLICIES

Fire management policies provide direct responses to wildfire, including such basic issues as whether wildfires will be suppressed or allowed to burn. Ecological science and ecological concerns are only one element in formulating these policies, albeit these are fundamental considerations. The level of attention that ecological considerations receive varies with the nature of the landscape and the management objectives. At this point, few national forests have well developed fire management plans that incorporate principles of restoration ecology (D. DellaSala, personal communication). There is thus an opportunity to develop fire management policies that are consistent with the long-term sustainability of ecological values.

Current fire management policies are not consistent with maintenance of many characteristic forest values, including biological diversity. For example, private timberlands, which typically are managed for high levels of timber production, may require intensive fire exclusion programs in regions that would otherwise be characterized by low fuel densities. Similarly, forests located within the wildland-urban interface may require intensive fuels-reduction treatments and adoption of fire-exclusion programs that are inconsistent with maintaining many of the organisms and ecological processes characteristic of the region (Marzluff and Bradley 2003, Radeloff et al. 2005, Noss et al. 2006). Consequently, it is inappropriate to develop a single, universal policy with regards to fire exclusion; rather, it should vary with social and ecological circumstances, including the characteristic fire regime. While current wildfire exclusion practices are often inappropriate, exclusion may be ecologically appropriate in cases such as the following:

- where rare or unique ecological values (including species habitat and populations) could be lost. Examples include fire-sensitive species, such as the narrow endemic Brewer spruce (*Picea breweriana*), and some relic stands of old growth;
- where uncharacteristic fuel accumulations have created the potential for fire that is outside the HRV, such as high-severity stand-replacement fire on a site that was characteristically subject to low-severity fire, but where restoration efforts have not yet been carried out; and
- where high-severity, stand-replacement fires were characteristic but where such fires are not currently viewed as ecologically desirable (e.g., old-growth forests in Pacific Northwest).

From an ecological standpoint, there is no better policy for fire as a process than allowing fire to serve its natural role. Unmanaged fire is infeasible in proximity to human settlements and infrastructure, or in some cases where economic resource values are high. Away from these areas, however, and especially in wilderness areas, roadless areas, national parks, and other large wildlands, there is opportunity to allow increasing wildfire and to benefit the species that require a diversity of natural fire regimes (e.g., Hutto 1995, Smucker et al. 2005). Wildfire is part of wildness, which is one of the management goals of these areas, but fire use is also desirable in more managed landscapes. After forest structure is restored, unmanaged wildfire may be the cheapest and most ecologically desirable way to maintain fire-dependent forests. Prescribed fires, in contrast, typically must be set during narrow windows of suitable fire weather, which often do not occur during the summer when pre-EuroAmerican fires typically burned, and these fires are costly to undertake. Moreover, cool-season prescribed fires cannot be expected to mimic the work accomplished by the hotter, natural fires of summer. Where prescribed fires must be used, a more ecologically-based approach would be to attempt to better mimic the seasonality, intensity, and pattern of fires typical of the HRV.

Wildlands dedicated to sustaining high levels of ecological value are predominantly public lands. However, private landowners may also find it beneficial to maintain characteristic fire regimes, as is the case with longleaf pine (*Pinus palustris*) forests in the southeastern USA, which have been managed to sustain native biodiversity, including game species, going back to the early 20^{th} century (Engstrom et al. 2005). The importance of involving private landowners in restoration is clearly stated in the HFRA, where priority is given to hazardous fuel reduction projects developed through community wildfire protection plans.

TREATMENT OF AREAS FOLLOWING INTENSE WILDFIRE

Ecological Significance of Post-Fire Landscapes with Their Legacies

Forest landscapes that have been affected by a major natural disturbance—such as a severe wildfire or windstorm event—are commonly viewed as devastated, biologically impoverished areas or even as moonscapes, such as in the case of the Mount St. Helens blast zone (Figure 3). In most cases such a perspective is far removed from ecological reality. Naturally disturbed, early successional forest landscapes typically are sites of high biological diversity and critical ecological processes (Lindenmayer and Franklin 2002, Franklin and Agee 2003). There are several key reasons for this, including an abundance of biological legacies, such as living organisms and dead tree structures, the migration and establishment of additional organisms adapted to the disturbed, early successional environment, and the release from tree dominance.

Post-burn landscapes typically have substantial capacity for natural recovery including rapid re-establishment of vegetative cover and, ultimately, closed forest. This is the result of both surviving organisms and the arrival of individuals and propagules from outside the disturbed area. The immigrants typically include an array of early-successional or pioneer species previously excluded from the site by the closed forest. In fact, overall species diversity—at least of higher plants and vertebrates—often achieves its highest levels following a natural stand-replacement disturbance and before re-development of closed canopy forest.

Important factors contributing to the high diversity of the pre-forest-closure successional stage(s) are: the combination of pioneer and surviving forest species; the presence of diverse plant life forms (herbs, shrubs, and trees) and structures (e.g., snags and logs) for habitat and food; availability of resources (e.g., light and moisture); and the diverse microclimates available in the absence of site-dominance by large trees. Of course, there are many other species, including closed forest specialists that cannot survive and utilize these early successional sites.

A number of key ecological processes are associated with the biologically and structurally diverse early successional conditions that exist prior to re-development of a closed forest canopy, many of them related to the species diversity that is present. One important process is nitrogen fixation—the conversion of elemental nitrogen to chemical forms (ammonium and nitrate) that can be utilized by higher plants. Significant nitrogen fixation is associated with many pioneer species, including species of alder (*Alnus* spp.), legumes (e.g., lupines [*Lupinus* spp.], peas, and vetches), and *Ceanothus* spp., which host azotobacteria in their root systems that are capable of nitrogen fixation. Depending upon site conditions and the density of host species, nitrogen fixation during this stage in succession can reach hundreds of pounds per acre, replacing nitrogen that has been volatilized by the disturbing fire and enriching the productivity of site. Many of the angiosperms characteristic of early successional habitats may also be important in sustaining fungal species that are essential ectomycorrhizal associates of conifers. The richness of organisms in ecosystems recovering from stand-replacement disturbances have been documented in such diverse habitats as the Mount St. Helens blast zone (Dale, Swanson, and Crisafulli 2005) and the 1988 Yellowstone fires (Turner et al. 1997, Wallace 2004)**.**

Biological legacies

Post-fire landscapes have high levels of biological legacies, which are the most important feature of the post-disturbance environment in many forest types (Figure 4) (Foster et al. 1998, Franklin et al. 2000, Lindenmayer and Franklin 2002). Biological legacies are biotic elements (organisms and structures) that persist from the pre-disturbance forest ecosystem. They include all types of living organisms and organic matter, much of the latter occurring in the form of snags, logs, and other coarse woody debris. Living legacies come in many forms from spores to mature trees and are particularly important as source pools for repopulating the disturbed site. By persisting on the site species avoid the difficult processes of dispersal from long distances and successful recolonization. Nevertheless, in some forest types (e.g., pinyon [*Pinus* spp.]-juniper woodland) the immediate post-fire landscapes can be quite barren and deficient in legacies, with a lag of a few (e.g., 3) years before surviving plants re-sprout abundantly. In these cases, significant erosion may naturally occur.

Organic matter is an important class of biological legacy, since even severe wildfires kill trees but actually leave most of the organic matter behind, generally consuming no more than 10 to 15% and often much less. Much of this organic matter survives in the form of roots, snags, logs, and other coarse woody debris that is essential habitat for wildlife and in influencing physical processes and conditions, such as erosion and stream channels (e.g., Maser et al. 1988 and Harmon et al. 2004). The essential roles of large snags and logs as habitat elements for the majority of forest animals, both vertebrate and invertebrate, are now widely known and well documented. (e.g., Thomas 1979, Brown 1985, Maser et al. 1988, Hutto 1995, Johnson and O'Neil 2001, Harmon et al. 2004, Smucker et al. 2005). Animals utilize such structures for a variety of purposes, including nesting, protective cover, and feeding. Woody debris is particularly important for aquatic ecosystems. In streams, it not only forms cover and substrate for fish and invertebrates, but is important in the processes which shape stream channels and create habitat diversity (Naiman et al. 2005). Viewed from the perspective of species dependent on woody debris and associated habitats it can rarely, if ever, be argued that there is an excess of such structure. Under natural conditions, these structural legacies persist and play essential ecological roles for very long periods of time (e.g., Maser et al., 1988, Spies et al. 1988, Harmon et al. 2004).

Biological legacies play critical roles in the rapid re-establishment of an ecosystem that is diverse and functional. Initially, legacies serve as lifeboats for elements of biological diversity, both directly and indirectly (Franklin et al. 1997, Franklin et al. 2000). Organisms that survive a fire in some form, such as root systems or seed banks, have in-situ repopulating mechanisms. Structures such as trees, snags, and logs also provide the habitat and sources of energy and nutrients that allow other organism to survive, from populations of epiphytes on surviving trees to animals and fungi that survive within snags and logs. Furthermore, these same structural legacies also modify the microclimates of the disturbed areas, allowing other organisms to persist.

Biological legacies play a significant role in structurally enriching the regenerated forest later in succession. For example, without legacies of large living trees, snags, and logs, the regenerated forest would consist of a single cohort of even-aged and relatively even-sized trees for many decades or even centuries. Legacies contribute to significant structural diversity throughout the first 100 to 200 years of even-aged stands, providing habitat for a much broader array of species than would otherwise be the case.

The long-term roles of snag, log, and coarse wood legacies in development of late-successional forest habitat is as important as their short- and mid-term contributions (Lindenmayer and Franklin 2002). The massive input of wood structures is characteristic and critical to stand development processes and, ultimately, the provision of habitat for late-successional species following stand-replacement fires. These wood structures may persist and play functional roles for several centuries, particularly in the case of decay-resistant species, such as Douglas-fir, western larch (*Larix occidentalis*), western redcedar (*Thuja plicata*), and species of juniper. Large pines also may be present as snags for several decades (or many decades in the case of western white pine on some sites) and as logs on the forest floor for additional periods. In streams, large decay-resistant logs may provide bank stability and habitat structure for centuries at a single locale, while others may gradually work their way downstream in response to freshets and other stream processes. At each stopping place, such logs become an integral part of the stream environment.

The pulse of snags, logs, and coarse wood generated by a stand-replacement-fire is the recovering forest ecosystem's sole source of coarse wood until the new stand begins to generate snags and logs of comparable size and heartwood content (Figure 5), which generally takes 150 to 200 years in some forest types (Maser et al. 1988, Franklin et al. 2002, Harmon et al. 2004). Furthermore, the larger and more decay-resistant the wood (a property of the species), the greater its long-term ecological significance. Port Orford cedar (*Chamaecyparis lawsoniana*) is an example of a species widely sought for its rot-resistant timber, but rot-resistance also plays a key role in its ecosystem.

The long-term persistence and multiple roles played by the large pulse of snags, logs, and other woody debris rarely has been given adequate attention in evaluating the ecological effects of post-fire logging, particularly on sites characterized by mixed- and high-severity fire regimes. The persistence and importance of these structures make it appropriate to create *higher* base snag and coarse wood retention goals than those found in existing mature and old forests. Such forests have a continuing source for input of such material, whereas the burned stand will have no further inputs following the fire until the stand has lived for at least a century.

As noted below, there may be circumstances under which other objectives, including risks associated with subsequent fires, mitigate against retaining all coarse wood structures. There are also no natural models to provide guidance with regard to retention of snags, logs, and other coarse wood following uncharacteristic stand-replacement fires on sites that were historically subject to low-severity fire regimes. Nevertheless, biological legacies, including live trees and large snags, are ecologically important on all sites, regardless of the characteristic fire regime.

Natural re-establishment of forest

Natural re-establishment of closed forest following stand-replacement fire occurs at widely varying rates, from essentially instantaneous establishment of a new cohort of trees to their gradual establishment over decades or even centuries. Important variables include the availability of tree seed, competing vegetation, and severity of site conditions.

Availability of seed in time and space often limits coniferous tree regeneration. Absence of seed following a fire may be due to an absence of surviving seed sources or, even if seed sources are present, a delay of one to several years before abundant seed are produced. Some western coniferous tree species—most notably lodgepole pine—have serotinous cones that survive and open following fire, providing a canopy "seed bank." Most western conifers are not serotinous, however, and also lack a seed bank in the forest floor.* Coniferous tree species typically produce abundant seed crops only at intervals of several years; hence, several years may pass following a burn before significant quantities of tree seeds are available. Further, even when seed is produced, dispersal of the seed may be limiting since most tree seeds fall only a short distance from parent trees. Hence, retaining surviving "legacy" trees is useful to provide potential seed sources as well as for the favorable microclimatic conditions that they provide for seedlings. In Yellowstone National Park, some 75% of sites experiencing stand-replacement fire were within 200 m of an unburned edge; hence, sources of tree seeds for recolonization of burned sites were readily available (Turner et al. 1994). It should be noted that lags in regeneration are normal in some cases and allow other species to dominate temporarily. Hence, lags in regeneration do not justify the planting of trees.

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* Occasionally a canopy "seed bank" may occur with species that do not have serotinous cones as a result of unusual circumstances. For example, in early October of 1993 the Walker Creek wildfire severely burned nearly 10,000 acres of old-growth Douglas-fir and western hemlock forest in the western Oregon Cascades. A bumper seed crop of Douglas-fir and associated conifers was ripe but still within the green cones at the time of the fire. The crown fire killed the trees by consuming or scorching the crown but much of the seed crop survived in the unopened cones. Large numbers of seed were subsequently released from these cones and resulted in rapid establishment of large numbers of coniferous seedlings throughout the burn (Larson and Franklin 2005). Similar conditions may have been associated with extensive wildfires that occurred in the

Cascade Range in the fall of 1902, and may also explain some of the results of Donato et al. (2006) in southwestern Oregon, where natural regeneration of conifers after high-severity fire was generally abundant in unsalvaged stands, in contrast to salvaged areas.

Competing vegetation can reduce the natural regeneration of desired forests following a fire. Many hardwood species, such as madrones (*Arbutus menziesii*), maples (*Acer* spp.) and oaks (*Quercus* spp.), and the odd conifer (most notably redwood) can reproduce vegetatively by sprouting. Such species are, therefore, not constrained by availability of seed and are often prominent in the early recovery of burned sites. Cover of grasses, herbs, and shrubs may develop rapidly and compete with tree seedlings, particularly if there is there is a delay in arrival of tree seed.

Severe site conditions can slow natural regeneration of coniferous trees following a stand-replacement burn—for instance, conditions on particular sites or in certain years that are hot and droughty or subject to frequent intense frosts or short cold growing seasons. Quick reburns that recur within 10-20 years on sites naturally experiencing relatively infrequent fire can reduce conifer regeneration (Schoennagel et al. 2003); for example, reburns may potentially eliminate many or most of the mature legacy seed trees that survived an initial stand-replacement fire event and stimulate the development of competing vegetation that is adapted to fire. However, seed sources might be limiting for other reasons, for example the young age of trees that originated following the earlier fire. Although the re-burn potential of different kinds of stands is not well documented, it appears that reburns occur more commonly in some settings than others. Burned snags and large logs, however, have little influence on initiation and spread of fire. What generally matters most for reburn potential is the amount of fine fuels, rather than the coarse fuels that are targeted in post-fire logging projects. Indeed, when large trees are removed by post-fire logging, the least flammable portion, the boles, are taken and the most flammable portion, the slash, is typically left behind. On the other hand, fire intensity likely would increase with greater densities of logs and snags in low-severity systems, and may be a problem if levels are uncharacteristically high.

Multiple burns in a short timeframe at the same site almost certainly create unfavorable conditions, such that sites could require up to a century or more to become fully restocked. For example, pioneer tree species (Douglas-fir, noble fir [*Abies procera*], and western white pine [*Pinus monticola*]) were still establishing over 100 years after the last wildfire in the Cowlitz River drainage of Mount Rainier National Park (Hemstrom and Franklin 1982). Whether the result of single or multiple burns, extended early successional conditions (i.e., limited areas of closed forest canopy) were probably common at times on sites subject to either mixed- or high-severity fire regimes under natural conditions.

Currently, natural early successional forest habitat—naturally disturbed areas with a full array of legacies (i.e., unsalvaged) and experiencing natural recovery processes (i.e., not seeded or planted)—is among the scarcest habitat condition in some regions, such as the Pacific Northwest. Indeed, it is often rarer than the old-growth forests that have justifiably attracted much conservation attention. As noted earlier, natural early successional habitat is typically very diverse in species, structures, and functions. Mount St. Helens and the Yellowstone fires of 1988 provide rare examples of large naturally disturbed tracts that have largely been allowed to recover on their own. Research at Mount St. Helens supports the concept that such large, slowly reforesting disturbed areas of this type may be important as regional hotspots of biodiversity for many taxa (Dale et al. 2005). Birds, amphibians, and mammalian predators are examples of groups that exhibit both high diversity and high populations in the Mount St. Helens landscape.

Ecological Impacts of Post-Fire Logging

General reviews of the physical and biological impacts of post-fire logging on terrestrial and aquatic systems have appeared recently in the peer-reviewed literature (Beschta et al. 2004, Karr et al. 2004, Lindenmayer et al. 2004, Lindenmayer and Noss 2006).These reviews, in addition to discussing biological and physical impacts of post-fire logging, acknowledge a fundamental problem of post-fire logging: projects must be implemented relatively soon after a fire while the commercial quality of the wood remains high; hence, such projects are often rushed rather than planned and reviewed carefully over an extended period (Le Goff et al. 2005). The impacts of post-fire logging in western forests can be best illustrated by examining its effects after high-severity and low-severity fire, respectively.

High severity fire. High levels of tree mortality are typical after high-severity or stand-replacement wildfires. Many forest types and sites in the western USA are naturally characterized by such fires, as noted earlier. The tree species in such forests are adapted to intense fire, which typically occurs at relatively long return intervals, although in some cases there is the potential for reburns early in the recovery cycle (see below). Such fires typically exhibit variable intensity on a landscape scale. That is, not all areas within the boundary of a large wildfire are subject to high or complete tree mortality; there are typically many unburned and lightly burned patches that form a mosaic with severely burned patches (Figure 6).

Post-fire logging is almost always inappropriate from an ecological standpoint (Lindenmayer et al. 2004, Lindenmayer and Noss 2006), perhaps especially after high-severity fire. Trees that survive the fire for even a short period of time are critical as seed sources and as habitat that will sustain many elements of biodiversity both above- and below-ground. Removal of structural legacies—living and dead—is inconsistent with scientific understanding of natural disturbance regimes and short- and long-term recovery processes on sites characterized by high-severity fire regimes. Removal of any material is a potential detriment, but removal of mature living trees and the largest and most decay-resistant snags and logs produces the greatest negative impact on recovery processes, slowing the recovery of ecosystem function and characteristic biodiversity. This is without reference to negative effects of the logging process itself, such as impacts of roads on soils and streams.

Concerns are sometimes raised about the retention of snags and other coarse wood leading to a higher likelihood of reburns or safety hazards. Reburn potential varies among forest types. In Rocky Mountain subalpine forests and Canadian boreal forests, little evidence exists for a high potential for quick reburns, perhaps in part because shrubs are a minor component of early-succession (Schoennagel et al. 2003, Krawchuck et al. in review). In Yellowstone National Park, some post-fire stands burned at intervals of less than 10 years, but these young stands were not necessarily favored. In fact, fire frequency modeling indicates that, on average, older stands are more likely to burn (Schoennagel et al. 2003). Although unburned or partially burned snags with dead fine fuels attached may attract fire, burned snags and large woody debris do not appear to provide a suitable substrate for fire initiation. Burned snags and large logs can increase fire severity and the duration of a burn (Brown et al. 2003), and may sometimes be a source of spot fires, but they are not a source of reburns. Thus, the role of coarse wood as fuel for re-burns is not a serious issue ecologically. Although intense re-burns early in the recovery process from a stand-replacement fire are characteristic of some sites (as observed in such regions as western Oregon and Washington, northern Idaho, and Wyoming), from an ecological perspective it is not valid to remove fuels from such sites in order to reduce the potential for reburns. Spies and Thompson (2006) show that areas logged and planted following a 1987 fire burned during a 2002 fire with significantly higher severity than comparable areas that were burned but not logged and planted.

In some cases snags may be significant hazards to humans involved in fighting subsequent fires. There are ways that these concerns can be dealt with while minimizing negative ecological impacts. For example, it may be possible to create snag-free zones in critical areas in the landscape, such as along ridgelines. Such practices were widespread during the first half of the 20th century in areas subject to repeated stand-replacement fires, such as the Yacolt and Tillamook Burns.

Low severity fire. Sites characterized by low-severity fire regimes would, by definition, rarely have experienced wildfires with high levels of tree mortality under characteristic fuel loadings. There are no natural models to use as guides to post-stand-replacement fire events on sites characteristically subject to low-severity fire, because this situation was probably rare under historic conditions. On the other hand, there is little hard information on the historic occurrence of stand-replacement fires in low-severity fire sites (e.g., Baker and Ehle 2001, 2003). Given these circumstances, approaches to post-fire logging are best based on first principles: assume harm unless it can be proven otherwise.

Some of the negative impacts of post-fire logging following a stand-replacement fire on a site characteristically subject to low-severity fire would be similar to those on sites where such fires are characteristic. Living trees, snags, logs, and other coarse wood are important to recovery processes (e.g., they are sources of energy and nutrients) and as wildlife habitat, throughout the first century (or more) of recovery. There is also the potential for damage to surviving non-arboreal vegetation (e.g., sprouting shrubs) and to soils as a result of the salvage process. Post-fire logging is also likely to increase the short-term loadings of fine and medium fuels, which may contribute to fire ignition and spread. In southwestern Oregon post-fire logging significantly reduced tree regeneration and increased the short-term risk of reburn (Donato et al. 2006).

Positive ecological effects of post-fire logging are possible in forests historically characterized by low-severity fires when such forests have experienced stand-replacement fires (for example, due to years of fire suppression) that left uncharacteristically high fuel accumulations. Such high fuel accumulations may render subsequent prescribed fires too hot, potentially adding further damage to the site. However, prescribed burning may not be immediately necessary in a forest that experienced an uncharacteristic high-severity fire. In the first decades, the main effects of such a fire would be to prevent tree recovery. In later decades, after trees are large enough to survive surface fire, a prescribed fire might be appropriate, but there is little ecological justification for such a fire until a natural stand-thinning stage has been reached, which in some regions (e.g., Rocky Mountains) is 80+ years after the severe fire. At that point, the main fire threat is not due to decomposed wood from the previous severe fire, but from abundant smaller dead wood from recent self-thinning.

A further consideration is that restoration to historical condition of ponderosa pine forests (and perhaps some other types) that have experienced severe fires may be slow or divergent in some cases. An historical analysis of the trajectory of recovery of southwestern ponderosa pine stands after crown fires showed convergence on one of two conditions, neither of which is considered characteristic: 1) extremely dense stands of ponderosa pine, or 2) non-forested grass or shrub communities (Savage and Mast 2005). Hence, forests subject to unusual severe fires may "recover" to alternative community states unlike the most prevalent historic condition.

Ecological Impacts of Revegetation and Reforestation Practices

After sites have been affected by fire, forest managers commonly initiate grass-seeding, planting of conifers, and other revegetation practices, on the assumption they accelerate forest regeneration. However, the effects of these practices are generally not well studied. In many cases, their effects are not consistent with maintenance of native biodiversity and other ecological values.

Post-fire revegetation: grass seeding

A common practice on federal lands is aerial seeding of areas burned by moderate- to high-severity fire in order to reduce post-fire erosion. The Burned Area Emergency Response (BAER) program is responsible for prescribing and implementing post-fire rehabilitation treatments with the goal of minimizing threats to life or property and degradation of natural and cultural resources due to erosion following wildfire. Although aerial seeding of non-native grasses is a common post-fire rehabilitation activity, its effectiveness in reducing erosion is varied, while impacts on native plant establishment appear to be mostly negative.

The effectiveness of seeding in reducing erosion appears mixed. Seeding generally does not mitigate erosion during the first winter following fire, when seeded grasses are not yet well established (Krammes 1960, Boyle 1982; Wright et al. 1982). Amaranthus (1989), for example, showed no significant difference in erosion between seeded and unseeded sites by the December following a summer fire, although during the first growing season seeded plots had twice the plant cover as unseeded plots. Roby (1989) found no difference in plant cover or erosion 2 years after fire due to seeding, and others report no difference during any of 3 years after fire (Leege and Godbolt 1985). Two Forest Service post-fire rehabilitation reports reviewed by Robichaud et al. (2000) show less sediment movement in seeded conifer stands the first year after fire; one report indicated less erosion the second year after fire due to seeding, while two other reports showed no difference. Some observers have indicated less erosion during the second year following fire due to the protective layer of dead grass that develops (Robichaud et al. 2000). Grass seeding does not appear to mitigate erosion during intense storms (Krammes and Hill 1963, USDA 1990).

Much of the confidence in seeding as an effective erosion control is based on the assumption that higher plant cover confers a reduction in surface erosion, but few empirical studies have tested this relationship under varied site conditions and seeding mixes. Few erosion studies exist primarily because of the cost and difficulty of measuring erosion during multiple years following fire. Extrapolation from existing studies is difficult because of the great variation in slope, soil, and precipitation following particular seeding treatments, which also vary in type and amount of seed. Furthermore, few studies provide replicated study designs and statistical analyses; many are observational rather than experimental (Beyers 2004). Lastly, there is a lack of understanding of how long the threat of high erosion typically persists; most studies only examine 1-3 years post-fire.

The ecological effects of seeding, in contrast to the varied results from erosion studies, are more apparent. Aerial seeding of non-native grasses generally reduces native plant cover or richness in conifer forests, primarily after the first year following fire (Anderson and Brooks 1975, Crane et al. 1983, Conard et al. 1991, Geier-Hayes 1995, Schoennagel and Waller 1999, Beyers 2004, Barclay et al. 2004, Keeley 2004, Kruse et al. 2004). Annuals appear more affected by seeding than perennial forbs, which typically resprout after fire. Fires provide infrequent opportunities for early successional colonizers such as annuals to maintain viable populations across the landscape, hence the impact of seeding could be significant for some functional groups.

Few longer-term seeding studies (> 5 years) exist, so persistent impacts of exotic grass seeding on the native understory community remain unknown. However, a number of studies report a significant reduction in conifer seedling establishment due to seeding (Griffin 1982, Conard et al. 1991, Schoennagel and Waller 1998, Keeley 2004). Low conifer recruitment presents a significant long-term consequence of seeding on forest recovery, even when the presence of seeded grasses is relatively ephemeral.

An additional ecological ramification of seeding is the potential for seeded sites to reburn relatively soon after seeding. Grasses provide a dense thatch of fine fuels that cure early in the season compared to native forbs that typically dominate mid- to high-elevation conifer sites. This shift in the understory fuel complex due to seeding may present a significant subsequent fire hazard (Beyers 2004). The effect of quick reburns on understory communities is still relatively untested, although resprouting shrubs appear to be most negatively affected (Zedler et al. 1983, Schoennagel et al. 2004b).

Seeding has been relatively successful in rangelands of the intermountain West in reducing the post-fire invasion and spread of exotics such as cheatgrass (*Bromus tectorum*) (see Beyers 2004 for review). In order to effectively reduce cheatgrass, and likely other exotics, sites must be seeded immediately after fire and subsequently protected from grazing (Beyers 2004). The effect of post-fire seeding on exotic species is less clear in conifer stands; however, lower cover of non-seeded exotics has been observed in seeded areas compared to unseeded sites (Schoennagel and Waller 1998, Keeley 2004).

The high cost, limited availability, and high demand for seed during large fire years make post-fire seeding with native species a significant challenge, although federal agencies are encouraged to use natives when feasible. Most seeding mixes include an annual grass for quick establishment, a perennial grass for long-term persistence, and a legume to enhance nitrogen fixation at the site. Sterile cereals and cereal-grass hybrids that persist only one season are increasingly used, although research on the ecological impacts of such mixes is currently lacking (Beyers 2004).

In areas where protection of human structures and key watersheds is paramount, seeding may be warranted. The ecological impacts of aerial seeding of non-native grasses in remote areas, however, could be significant across large areas (Keeley et al. 2006). Such seeding could lead to reduction in cover and richness of early successional native plant species, inhibition of post-fire conifer establishment, and a threat of reburns at uncharacteristically short intervals. Clearly, major post-fire erosion can pose threats to both human and ecological systems. In areas where severe wildfires are characteristic, however, a precedent for post-fire erosion exists and should not be considered a novel ecological threat from which the system cannot recover. In areas where severe wildfire is uncharacteristic, reducing the threat of significant post-fire erosion may be a higher priority. Resprouting perennials are the dominant post-fire colonizers, so examination of the pre-fire community (based on local knowledge or adjacent unburned areas) should be part of any evaluation of the potential for the site to revegetate naturally. In areas where resprouting perennials, especially grasses, were abundant before the fire, seeding non-natives grasses is not recommended. Indeed, tradeoffs exist between potential erosion reduction and the ecological consequences of seeding, which are difficult to evaluate.

Although seeding is typically conducted under emergency circumstances, we encourage BAER teams to create unseeded controls within seeded areas, which permit replicated studies to statistically test the effectiveness and ecological impacts of seeding under varied site and weather conditions. We also encourage use of native plants for post-fire seeding and rehabilitating bulldozed fire lines created during fire suppression activities, which can amount to hundreds of miles in relatively remote areas following large fires.

Post-fire revegetation: tree planting

Some of the most serious problems with post-fire forest recovery result from tree-planting after fire. The desire to rapidly restore a fully stocked forest is the predictable response of land managers to a stand-replacement fire. The fundamental assumption is that "recovery" or "rehabilitation" involves the prompt re-establishment of forest cover. This response probably reflects traditional perspectives and land management objectives more than any ecological rationale.

Wildfires in forests have been historically viewed as events that destroyed valuable standing forest and created large expanses of deforested landscape. Re-establishment of forest was viewed as critical to ecological recovery on such sites and was a focus of traditional forestry. Circumstances and knowledge have changed significantly since the early 20th century. Through recent research, knowledge of disturbances, biological legacies, and the importance of structurally complex early successional habitat (i.e., burned areas prior to development of closed forest canopies) has expanded dramatically. The justification for rapid re-establishment of closed forest conditions no longer exists. Each area needs to be examined in the context of land management objectives and existing post-burn conditions, including the potential for natural regeneration. Goals for tree regeneration need to consider the desirability for variable stand densities, rather than stands that are “fully stocked” and uniform in density. Hardwood trees and shrubs, many of which sprout following fire, are important components of the recovering ecosystem and are used by many species of birds and mammals. Ecological objectives are likely to be better served by natural regenerative processes than establishment of plantations, however much that idea goes against forestry tradition and popular perception.

There are ecological models to guide reforestation of burns in those relatively rare cases where reforestation is needed to achieve ecological goals. Planting or seeding generally should be varied in density; large areas of dense natural reproduction are not characteristic, although they may occur, particularly where a persisting seed source (e.g., from serotinous pine cones) is available. As noted earlier, tree reproduction on many burns is patchy and often of low overall density. Old Douglas-firs in western Oregon appear to have originated and developed in stands with densities of around 50 trees per acre, for example.

Reforestation should utilize tree species and genetic material that are of local origin and, hence, known to be suited to the site. Many examples exist of attempts to reforest areas using species or genotypes that are not suited to the site (e.g., Isaac 1949). Climate change may justify some deviation from this principle, however, such as to provide for introduction of species and genotypes that are more tolerant of warmer, drier (droughtier) conditions than were historically characteristic of the site.

As a final comment on reforestation, perhaps the least appropriate activity that can be undertaken is creation of a dense uniform plantation following an uncharacteristic stand-replacement fire on a site that is characterized by low-severity fire. Such a fire is likely a consequence of an uncharacteristically dense stand on the site as a result of some human activity, such as fire suppression. Traditional plantings on such sites, such as with 600 to 800 trees per acre in some uniform grid, simply re-create the conditions for the next, uncharacteristic stand replacement fire.

EFFECTS OF FIRE AND FIRE MANAGEMENT ON AQUATIC ECOSYSTEMS

Most of this report has focused on forests as terrestrial ecosystems. However streams, lakes, marshes, and other aquatic habitats are literally at the bottom of every terrestrial ecosystem, so the effects of fire are often concentrated as the residues of fire flow, roll, or are carried downhill into the water. Yet aquatic ecosystems, especially those of streams, are also extremely resilient and have great capacity to recover rapidly from the effects of fire. Even given this resilience, recovery from severe events (and post-event human actions, such as post-fire logging) can take decades. Rapid recovery of aquatic ecosystems and their associated riparian systems following fire and other large disturbances is important because much biodiversity is concentrated in and around wet places, often in the form of fish, migratory birds, and other organisms with high value to humans (Naiman et al. 2005).

While the literature on fire effects on aquatic systems is sparse, recent reviews have made much of it readily accessible (Gresswell 1999, Dunham et al. 2003, Minshall 2003, Pilliod et al. 2003, Rieman et al. 2003, Spencer et al. 2003, Karr et al. 2004). These reviews suggest that in forests with a natural fire regime there is a fairly predictable response to fire (see figure in Gresswell 1999), as follows:

1. During a fire, mortality of aquatic organisms is rarely observed because of the natural buffering effect of the water and the tendency of fires to skip or only lightly burn the wetter riparian areas.

2. In the first year following a fire, there is likely to be an increase in sediment, organic debris (litter), and nutrients flowing into streams, which diminishes as vegetation reclaims the burned areas. The increase in solar radiation with the diminished canopy combined with the increase in nutrients can result in increased algae production. This in turn can lead to increased growth and survival of grazing invertebrates and the fish (and perhaps birds) that feed on them.

3. In the next year or two following the fire, nutrient and sediment inputs gradually drop down to background levels, although the reduced canopy results in an increase in primary production in the streams and a decrease in litter (leaves etc.) from the trees. Usually in this period there is an increase in the abundance of aquatic insects, resulting in an increase in fish abundance and growth rates. The fish and invertebrates may also benefit from the increase in coarse woody debris in streams as trees killed by the fire fall into the water. Coarse woody debris not only provides cover for fish and substrate for invertebrates but makes stream channels more complex with its interactions with flowing water.

4. Typically after 5-10 years, the aquatic system has more or less returned to its original condition.

Obviously there is considerable variation on this theme in natural systems, related to the intensity of the fire, the size of the stream (e.g., small, low-order streams are more likely to be severely impacted than large, high-order streams), the nature of the forest, the aquatic species present, the degree of prior fire suppression, and other factors. The general picture, however, is one of rapid return to pre-fire conditions. In forests subjected to severe fire and post-fire logging, however, streams and other aquatic ecosystems will take longer to return to historic conditions or may switch to a different (usually less desirable) state altogether. Post-fire logging may aggravate existing road conditions or create more roads that are vulnerable to erosion. Minshall (2003) found that post-fire logging of 25-40% of the standing volume of timber after fire can significantly increase sediment impacts to watersheds. The negative effects of post-fire logging can continue for decades after the logging because many streams depend on a continuous source of large logs to create structure needed directly by fish and indirectly to generate complex stream channels (Naiman et a. 2005). Removal of the large trees and logs from an area essentially stops the normal succession of events that maintains complex stream habitat until large trees have grown back again. For this reason, Reeves et al (2006) recommend that logging restrictions in riparian zones be maintained even if the area has been seriously burned.

Where management suppresses fire, it is likely that aquatic ecosystems are less productive than they were historically, as the result of heavy shade from both the canopy and fallen trees, especially in low-order streams, and from the impact of roads and other human activity. Under these conditions, inevitable high-severity fires are likely to cause an abrupt shift in conditions, exposing the stream to sunlight, increasing erosion (and sedimentation), and dumping large amounts of burned material into the water. Barber et al. (2003) point out that intense fires can even release toxic concentrations of chemicals such as cyanide that are stored in vegetation, resulting in fish mortalities. Chemicals used to suppress fires may also have toxic effects (Backer et al. 2004).

Following a severe fire, often the biggest impacts on aquatic ecosystems are increased sedimentation caused by run-off from new roads, at levels far above natural levels. High sediment loads from eroding roads may continue for years, greatly increasing the time for recovery. Roadless areas thus serve as important refugia for aquatic as well as terrestrial organisms (Strittholt and DellaSala 2001, Noss et al. 2006). Dunham et al. (2003) note that change by severe fires may increase the likelihood of successful invasions of streams by alien fishes. In general, the impacts of any fire on aquatic ecosystems are likely to be most severe on systems that are already fragmented or altered by human activity (e.g., by dams, roads, logging, mining, development), because a fire is likely to aggravate or expand existing unfavorable conditions.

CONCLUSIONS

Despite the complexity of fire ecology in western forests and uncertainty over the effects of particular management actions, the scientific basis for rational decision-making has improved dramatically in recent years. Ecological science should be incorporated systematically in the development of wildfire and fuel polices and management practices prior to, during, and following wildfires. Recent advances in ecological theory and empirical research underscore the importance of variability in ecosystems and the processes, such as fire and other disturbances, which shape them. Logging, livestock grazing, and fire exclusion have led to forests that are outside their historical range of variability, particularly in dry ponderosa pine and dry mixed conifer forests, but the effects of these land-use changes are not necessarily easy to identify or demonstrate using scientific evidence (Bridge et al. 2005). In too many cases, it has been assumed that forests in general or all forests dominated by particular tree species have suffered from these land-use changes in the same way. Our review suggests that land-use changes have led to significant deviations from historical variability in some forests but not in others; variation occurs not only among forest types, but also within forest types.

Interest is increasing among land managers in lowering the risk of severe fire through thinning and other fuels-reduction treatments, particularly in ponderosa pine and dry mixed conifer types. However, not all of these forests have been altered significantly in structure (e.g., tree density), relative to historical variability, and not all require thinning or other treatments. Broad generalities about the stand structure and fire regime (severity, frequency, seasonality, etc.) of these forests have been refuted in specific cases (e.g., Shinneman and Baker 1997), and cannot be considered valid across the range of these ecosystems. In many cases, the data needed to determine whether high-severity fires historically occurred in these forests have not been collected (Baker and Ehle 2003). Where the evidence has been collected and examined, there are cases where high-severity fire was and was not a part of historical variability. Sometimes, both of these may occur within the same region, even on adjoining or nearby slopes (Ehle and Baker

2003, Baker et al., in press, Sherriff et al., in review). Therefore, substantive landscape-scale evidence must be collected and analyzed before it can be determined that a particular forest setting experienced a particular fire regime historically. Appropriate evidence includes 1) tree-ring dating of stand-origins across a landscape, accompanied by a network of cross-dated fire scars; 2) examination of historical records, including forest reserve reports, early newspaper articles, early stand exams, early maps, historical accounts and local histories, and other sources of qualitative or semi-quantitative historical evidence, and, if possible, 3) charcoal and pollen analyses. Fire history evidence is needed across whole landscapes, and it should be collected across a statistically unbiased network of sites.

High fuel loads, dense forests, and ladder fuels are not necessarily ecological problems, because many forests had these characteristics as part of their historical range of variability. Land uses and fire exclusion can certainly increase fuel loads and fire risk, but they do not do so universally; instead they may alter fuels in divergent or complex ways that lead to a need for decreases in particular fuels and increases in other fuels, if restoration to historical variability is the goal. For example, fire exclusion can increase tree regeneration and ladder fuels in some cases (Covington and Moore 1994) and decrease tree regeneration and ladder fuels in other cases (Ehle and Baker 2003). Logging may decrease large, dead fuels by removing trunks, but increase fine, dead fuels by leaving slash. Thus, a common restoration need in logged forests is to increase snags and large, dead wood (potentially increasing the risk of higher intensity fire), while decreasing fine, dead fuels left behind by logging (potentially decreasing the risk of ignition and rate of spread). The particular history of land uses, and the specific effects of these uses on individual fuel components must be considered in order to formulate ecologically based restoration prescriptions. Restoration of fuels and fire risk to a historical or natural range of variability, rather than blanket fuels reduction, is most compatible with restoring and maintaining biodiversity.

Because ecological variability is great, there are few universal principles for integrating insights from ecology and conservation biology into fire management and restoration policies. Nevertheless, one principle that seems to hold is that as forests are managed or restored, they "should not only support the fire regime of interest, but also viable populations of native species in functional habitat networks across space and through time" (Hessburg et al. 2005: 136). A universally desirable conservation goal is a forest ecosystem with its fire regime, fuels, tree population structure, and other living organisms restored to within historical variability. Where this cannot be fully achieved, the greatest conservation value may be obtained where deviation from HRV is minimized in general and particular attention paid to maintenance of the whole range of native terrestrial and aquatic biota.

Over much of the West, ponderosa pine and dry mixed-conifer forests are most altered by logging, livestock grazing, and human-set fires. Large expanses of forest are dense because they are young, due to logging or stand-replacement fires in the late 1800s and early 1900s, and overgrazing by livestock, which facilitated tree regeneration by reducing competition with grasses. If restoration is to be successful, these land uses themselves must be reformed, so that an endless cycle of treatment, degradation by unsustainable land uses, and a new need for treatment does not continue. Forest understories, in particular native bunchgrasses, forbs, and shrubs, play a significant role in competing with young trees and in preventing or slowing tree regeneration.

In this regard, restoration of degraded understories is as essential to successful ecosystem restoration as is restoration of tree population structure. Moreover, forest understories often contain the richest biodiversity (especially of plants) in the forest and provide food for a variety of other organisms.

HFRA, Fire, and Ecology

To the extent that scientists are involved in developing policies and practices, they tend to be specialists in fire and fuel management, not ecologists, conservation biologists, or other broadly trained scientists. It is not surprising, then, that current forest policy such as the Healthy Forests Restoration Act (HFRA) of 2003 does not adequately incorporate ecological science in its implementation and promotes a narrow definition of restoration that focuses almost exclusively on fuels (DellaSala et al. 2004, Schoennagel et al. 2004a).

The objective of HFRA is to restore forests that have experienced moderately to significantly altered fire regimes over the last century. How departure from historical fire regimes is assessed for this purpose is a highly structured and detailed process (www.landfire.gov, www.frcc.gov), requiring quantitative spatial and temporal data on fuels and fire regimes for numerous forest types that are often unavailable or vary widely in accuracy. Expert opinion and unvalidated models often are used in place of systematic, quantitative, peer-reviewed studies. Often, such studies are not available, but sometimes they exist but are not used. Hence, although prioritization of sites for restoration activities under HFRA appears to be a rigorous and highly structured process, the specificity required by this process far exceeds available data, and in effect, renders it of questionable scientific merit.

Additionally, the concept of forest restoration under HFRA is very narrow, focusing almost exclusively on structural restoration of fuels and tree density important to fire risk mitigation. Ecological restoration, however, requires the maintenance of ecological processes, native species composition, and forest structure at both stand and landscape scales, as we have discussed in this paper. Broader conception and implementation of restoration objectives beyond fuel and fire mitigation are necessary to satisfy more comprehensive, scientifically based approaches to ecological restoration of western forests.

Recommendations

A primary conclusion of this report is that forest and fire policy should recognize and accommodate wide differences that exist in the characteristic role of fire among forest types and sites. Such policies also need to explicitly recognize that fires and their aftermaths affect many organisms besides just trees and that the value of these organisms to humans and ecosystems may greatly exceed the value of the harvestable timber. Hence, incorporation of scientifically defensible ecological information and judgment in policy and management decisions is critical. It follows that:

- Flexibility and discretion on the part of forest managers are necessary in designing and applying treatments, but can lead to undesirable consequences unless accompanied by substantial experience and knowledge of forest ecology. Management actions must be considered within broad and acceptable policy objectives and with the opportunity for meaningful public review. As part of developing and holding this consensus, such approaches as third-party reviews should be considered (see following)

- Credible third-party scientific reviews are critical when major controversies arise as to the scientific credibility of proposed activities, such as the use of fuel condition classes for stratifying and prioritizing activities. The National Academy of Sciences is an appropriate venue for such reviews, but we are encouraged that the more specialized scientific societies (e.g., the Society for Conservation Biology, which sponsored this report) are already addressing this need.

- Issues of economic or political vs. ecological goals should be considered explicitly. We have addressed ecological rather than economic issues in this report. The choice to implement treatments and serve economic or political goals by sacrificing ecological values (e.g., paying for restoration treatments by logging large, old trees) is a policy decision, which science can only inform. However, the comprehensive and long-term consequences of such decisions on ecological values must be given due consideration. It is important to recognize that there are risks associated with restorative treatment of stands and landscapes. These include: 1) uncertainties/risks associated with basing treatments on inadequate knowledge (in which case a precautionary approach should apply); and 2) risks associated with not taking restorative actions, including the potential for loss of significant ecological values. An example is the loss of northern or Mexican spotted owl habitat to uncharacteristically large stand-replacement fires on the eastern slopes of Cascade Range or in the Southwest, respectively.

- Priorities for restoration need to be determined on the basis of ecological considerations and urgency. High-priority cases may include areas where significant ecological values are at risk of uncharacteristic stand-replacement fire. We note that these are not necessarily the sites that are most out of synchrony with natural fire cycle; for instance, mixed-conifer sites often develop heavy fuel loadings and ladder fuels much more quickly than old-growth ponderosa pine sites, even though the mixed-conifer sites have not missed as many burning cycles, due to higher productivity and presence of species (e.g., grand or white fir) that provide superb ladder fuels. Areas where biological values are likely to be paramount include essential habitat for rare, threatened, or endangered species and key watersheds for protection of aquatic species.

Decisions regarding management of fire-prone forests should be made with the understanding that forests face threats other than uncharacteristic fire or inappropriate management actions. Threats that have the potential for producing uncontrollable or irreversible changes in forests include rapid climate change and invasions or population increases of non-native pest species (weeds, pathogens, etc.). Where invasive species are present, they should be controlled before restoration treatments that open the forest canopy, disturb the soil, or include fire are undertaken, because all three of these activities have been shown to favor invasive species. It is not sound restoration policy to trade desired forest structure and fire regimes for invasive species dominance. Livestock are also non-native species, and forest restoration programs must consider the effects of livestock grazing on fire regimes and regeneration conditions.

A major complication to science-based forest and fire management is the rapid increase of the wildland-urban interface (WUI), a result of urban sprawl, which already includes 9% of the U.S. land area and is expanding in the western United States as people build houses in and near fire-prone forests (Dombeck et al. 2004, Radeloff et al. 2005). WUI areas may require fuel reduction and fire management policies that are inconsistent with maintaining the biodiversity of those sites, even though carefully tailored treatments can maintain some aspects of biodiversity. An expanding WUI fragments landscapes, introduces invasive species, and may be a source of uncharacteristic fires; all of these lead to ecological impacts that extend well outside the WUI itself. To minimize the adverse impacts of the WUI requires, not just appropriate fire management policies, but also growth-management policies directed at minimizing urban sprawl into wildlands (Marzluff and Bradey 2003, Radeloff et al. 2005, Theobald et al. 2006).

Overall, a valuable and common-sense conservation goal is to seek to achieve restored forests that are low maintenance and do not require repeated treatment. In this regard, wildland "fire use" is the cheapest and most ecologically appropriate fire management policy for most forests. We envision a future where fire is seen by land managers and the public as a friend of healthy forests but where each forest and each patch of the forest mosaic is recognized for its individuality and managed according to its needs. The precautionary principle should therefore always be the first principle of forest management: "do no harm."

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Figure 1. A restoration continuum.

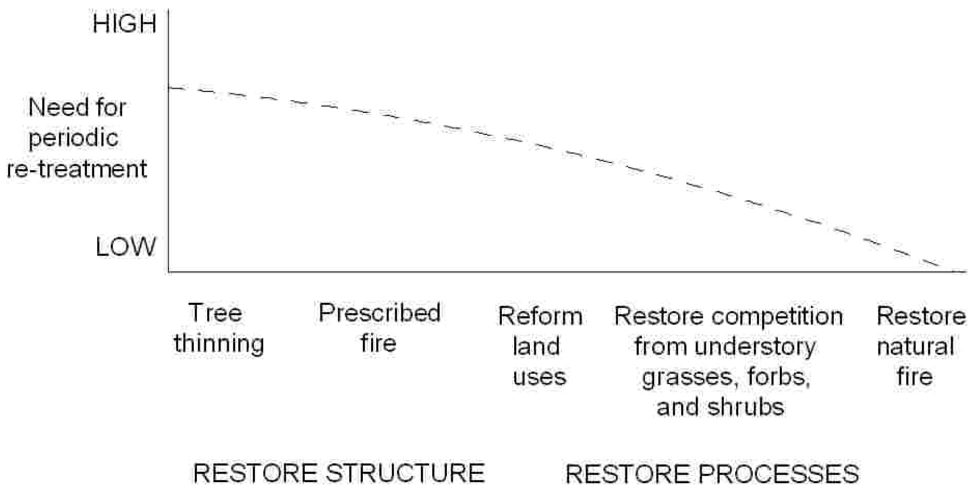

Figure 2. Profile of a ponderosa pine stand (courtesy Robert Van Pelt).

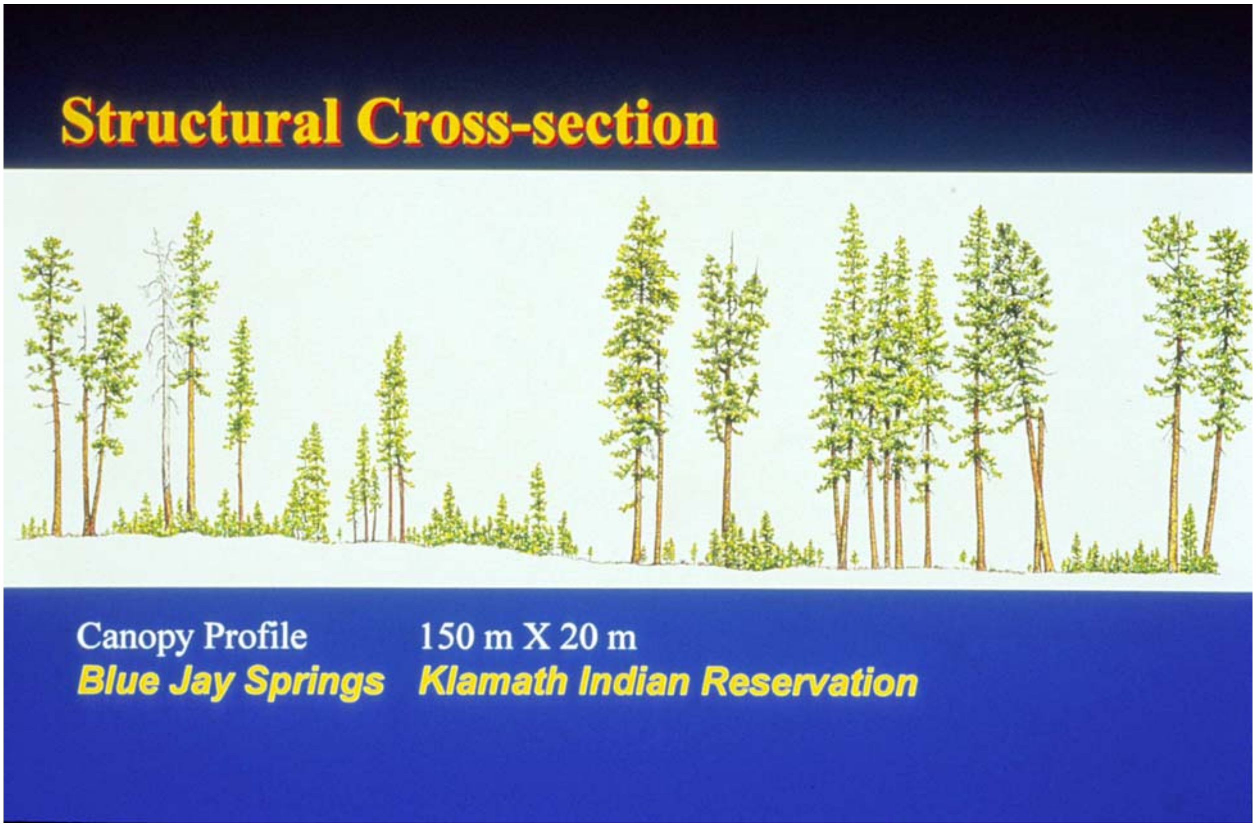

Figure 3: Mount St. Helens blast zone.

Figure 4. Structural biological legacies (snags and down logs).

Figure 5. Douglas-fir regeneration amongst snags (i.e., the stand described in Larson et al. 2005).

Figure 6. Mosaic fire pattern in Madison Canyon, Yellowstone National Park; July 24, 1989 (Photo: Jim Peaco).

Table 1. Fire regimes of major western forests (Kilgore 1981, Agee 1993, 1998b; Arno 2000) and some examples of plant association groups in each type.

Dominant fire type(s)	*General forest type*	*Common plant association groups*
High-severity	Coastal temperate forests	Sitka spruce, Western hemlock, Western red cedar, Douglas-fir
	Coastal subalpine forests	Mountain hemlock, Pacific silver fir
	Pinyon pine-juniper woodlands	Colorado pinyon, Singleleaf pinyon, Utah juniper, Western juniper
	Interior Northwest montane forests	White pine, Western red cedar, Western hemlock
	Interior subalpine forests	Engelmann spruce-subalpine fir, Lodgepole pine, Bristlecone pine, Limber pine, Whitebark pine, Quaking aspen
Mixed-severity	Coastal oak woodlands	

	Rocky Mt. ponderosa pine-Douglas-fir forests	Ponderosa pine, Douglas-fir, Western larch
	Interior mesic mixed conifer forests	Douglas-fir, White fir, Aspen
	Klamath-Siskiyou mixed-evergreen forests	
	Sierra Nevada red fir forests	Shasta red fir
	Sierra Nevada giant sequoia forests	Giant sequoia
Low-severity	Dry ponderosa pine forests	Ponderosa pine, Jeffrey pine
	Dry mixed conifer forests	Ponderosa pine, Douglas-fir, dry Grand fir