Effects of Timber Harvest following Wildfire in Western North America: Issues and Scientific Evidence

David L. Peterson, James K. Agee, Gregory H. Aplet, Dennis P. Dykstra, Russell T. Graham, John F. Lehmkuhl, David S. Pilliod, Donald F. Potts, Robert F. Powers, and John D. Stuart

D.L. Peterson.[1] USDA Forest Service, Pacific Northwest Research Station, 400 N. 34th Street, Suite 201, Seattle, WA 98103, USA.
J.K. Agee. College of Forest Resources, University of Washington, Box 352100, Seattle, WA 98195, USA.
G.H. Aplet. The Wilderness Society, 7475 Dakin Street, #410, Denver, CO 80221, USA.
D.P. Dykstra. USDA Forest Service, Pacific Northwest Research Station, 620 SW Main, Suite 400, Portland, OR 97205, USA.
R.T. Graham. USDA Forest Service, Rocky Mountain Research Station, 1221 S. Main, Moscow, ID 83843, USA.
J.F. Lehmkuhl. USDA Forest Service, Pacific Northwest Research Station, 1133 N. Western Ave., Wenatchee, WA, USA.
D.S. Pilliod. Department of Biological Sciences, California Polytechnic State University, San Luis Obispo, CA 93401, USA.
D.F. Potts. College of Forestry and Conservation, University of Montana, Missoula, MT 59812, USA
R.F. Powers. USDA Forest Service, Pacific Southwest Research Station, 3644 Avtech Parkway, Redding, CA 96002, USA.
J. D. Stuart. Forestry and Watershed Management, Humboldt State University, 1 Harpst Street, Arcata, CA 95521, USA

[1] Corresponding author (email: peterson@fs.fed.us). Note: Authors are listed alphabetically with the exception of D.L. Peterson.

Abstract. Post-wildfire timber harvest may lead to different outcomes depending on the biophysical setting of the forest, pattern of burn severity, and operational aspects of tree removal. Harvest generally is conducted on a small proportion of the total area burned by wildfire, with effects on some resources (especially soil and water) potentially being high at small spatial scales but low to undetectable at large scales. The scientific literature on postfire timber harvest in western North America lacks data from replicated, long-term studies, although it can be combined with the literature on individual effects of fire and timber harvest to develop inferences.

Timing of timber harvest following fire (same season as fire vs. subsequent years, winter vs. other seasons) can influence the magnitude of effects such as mortality of naturally regenerating trees, soil compaction and erosion, and commercial wood value. High fire hazard following fire can exist, and although removal of standing dead trees may help reduce this hazard, removal of surface fuels (including slash) is usually necessary to mitigate the effects of timber harvest. Historical data suggest that conditions that facilitate insect attack are in some cases more common following wildfire; removal of low-vigor trees is an option for mitigating susceptibility to insects.

Timber harvest with ground-based equipment and cable yarding can compact the soil and facilitate short-term erosion. Fire and timber harvest, individually and combined, reduce water uptake by vegetation, causing an immediate response in hydrological processes; stream flow and water supply can initially increase substantially, then typically decrease as vegetation regrows. Water quality in streams can decrease if increased stream flow is accompanied by erosion, with large debris movements being especially damaging.

Cavity-nesting birds and other cavity-nesting vertebrates may be affected by harvest of standing dead and live trees, with variable effects (mostly negative but some positive) depending on the habitat requirements of each species. Short-term effects of removing trees on aquatic systems are mostly negative, and timber harvest and transportation systems that disturb the soil surface or accelerate road-related erosion may be particularly harmful. Targets for retention of standing dead and live trees help maintain habitat for cavity-nesting animals, and targets for retention of large woody debris help maintain habitat for amphibians and invertebrates in upland and riparian forests.

Large wildfires provide opportunities for management experiments on the effects of postfire timber harvest. Long-term research and monitoring, using a quantitatively robust and consistent approach, would facilitate adaptive management of forest ecosystems subject to fire disturbance and reduce uncertainty in decision making.

Introduction

Timber harvest following large wildfires is typically conducted to capture the economic value of wood and sometimes may achieve other resource management objectives (e.g., reduced stand densities). This practice (often referred to as “salvage logging”) has been conducted for many decades in forests of western North America. The relative *social value* of economic benefits versus potential changes in resource conditions has been debated in scientific and policy forums for nearly two decades. However, the *scientific basis* for decision making about postfire management has not been effectively articulated.

In this paper, we synthesize scientific findings on the effects of timber harvest following large wildfires, with emphasis on forests in western North America. Our objective is to clarify the extent to which different issues are supported by scientific data, with a focus on reducing uncertainty in decision making about postfire management. We infer general principles where possible, while recognizing that biogeographic variability influences local ecological responses and management decisions.

The 25 or so studies that have been conducted on the effects of postfire timber harvest in western North America (Table 1) are disparate in geographic setting, study design, sampling, and analytical approach. In contrast, the effects of wildfires on forest ecosystems in western North America have been described in considerable detail in the scientific literature (e.g., Agee 1993; Schmoldt et al. 1999; Neary et al. 2005c). Similarly, the effects of timber harvest on forest ecosystems have been well described at different spatial scales from stands (Chen et al. 1995) to landscapes (Hunter 1999). We draw on this abundant literature on wildfire and timber harvest to consider scientific issues and evidence relevant to timber harvest on postfire forest landscapes in western North America. We synthesize information that is most relevant to inferences about postfire timber harvest, rather than conduct a comprehensive review of the literature.

Wildfires burn with a wide range of intensities and, depending on the prefire environment, can create a variety of postfire conditions in vegetation, fuels, and soils (Schmoldt et al. 1999). Severity of fire damage to trees is highly variable and can be characterized both vertically and horizontally (Peterson and Arbaugh 1986; Ryan and Reinhardt 1988; Turner et al. 1999). Wildfire effects range from minimal evidence of injury in crowns and boles to consumption of foliage, smaller stems, and bark, with severity varying widely across thousands of hectares in large fires. We confine our analysis to the effects of large wildfires and subsequent harvest in the postfire environment.

Geographical and Historical Context

This paper focuses on forests of western North America, extending from roughly the coastal ranges, Cascade Range, and Sierra Nevada eastward to the Rocky Mountains and associated ranges, and from southwestern Canada to the southwestern United States. These forests include woodlands such as pinyon-juniper (*Pinus edulis*/*Juniperus* spp.) in the American Southwest as well as forests characterized by large trees, including: (1) dry forests dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*), (2) moist forests dominated by grand fir (*Abies grandis*), western redcedar (*Thuja plicata*), and western hemlock (*Tsuga heterophylla*), and cold forests dominated by Engelmann spruce (*Picea engelmannii*), fir species (*Abies* spp.), and lodgepole pine (*Pinus contorta* var. *latifolia*, var. *murrayana*). The diversity of vegetation and biophysical settings in these forests is affected by interaction of disturbance agents with various intensities, severities, frequencies, and extents. Fire, grazing, insects, fungal pathogens, wind, and timber harvest, along with vegetation establishment, growth, and succession interact across a range of spatial scales to create fine-scale (e.g., windfall: <0.1 ha) to large-scale (e.g., wildfire: >500 ha) mosaics across landscapes (Alexander et al. 1990; Graham 1990; Lotan and Critchfield 1990; Shepperd and Battaglia 2002).

Forests as Reservoirs of Carbon and Nitrogen

Forests are large sinks and potential sources of carbon (C). Forests cover slightly more than 25% of the Earth's land surface (4 billion ha) but sequester nearly half of the terrestrial C, containing 640 Gt of C in biomass and soil to a depth of 30 cm (FAO 2005). Carbon in forest vegetation, litter, and soil combined to a depth of 1 m may exceed that contained in the atmosphere. On average, one-third of forest ecosystem C is above ground, with a smaller proportion in colder climates and a higher proportion in warmer climates (Van Cleve and Powers 1995).

Forests accumulate organic matter in the absence of major disturbance, and Western forests often exceed 500 Mg ha^{-1} in standing biomass (Table 2), of which roughly half is organic C. At maturity, 50 to 75% of above-ground C is in the boles of trees, followed by tree crowns and the forest floor (Table 3); understory vegetation in most forests contributes little to above-ground C. Nitrogen (N) follows a different pattern than C, because N concentrations are higher in foliage and litter than in wood. Quantities of N in forest floor detritus may exceed that in the crowns or boles (Table 3), and N concentrations in the forest floor of Western forests commonly are much higher than in the mineral soil (Cole and Gessel 1992). Forest floors are a reservoir of soil fertility. Where the forest floor has not been removed by fire, the quantity of forest floor N in temperate forests can equal that in all standing biomass (McColl and Powers 1984; Powers and Edmonds 1992). Rates of forest floor decomposition and N release depend on climate, with mean residence times of months to decades in temperate regions (Cole and Gessel 1992).

Soil C concentrations are generally high near the surface and decline with depth. Humic compounds help bind mineral soil particles into aggregates, improving their resistance to erosion. The high ratio of surface area to weight of soil organic matter maintains soil macroporosity and resistance to compaction, thereby facilitating soil aeration, infiltration of water, and retention of soil moisture. Soil organic matter from litter decay and root turnover is the main source of soil N, which is concentrated near the soil surface and declines with depth (Fig. 1). Thus, the upper few centimeters of surface soil are enriched in N and contribute to the nutritional health of forest vegetation.

Effects of Fire Exclusion

Forest tree species abundances, stand structures, and fuel structures in western North America currently differ from those that occurred prior to around 1900 due to extensive forest harvest, cattle grazing, fire exclusion, climatic variability, and altered land use transitioning from native peoples (First Nations) to Euro-Americans. Area burned has varied considerably over time as a function of climatic variation (e.g., Little Ice Age, Pacific Decadal Oscillation), fuel accumulation, and recent fire exclusion (Agee 1993). Recent fires in some dry forest regions have been larger and more severe due to elevated fuel accumulations caused by several decades of fire exclusion (Graham et al. 2004). Therefore, much of the larger woody debris accumulating on the forest floor may be a legacy of fire exclusion (Skinner 2002; Agee 2003b). Because fire is the dominant disturbance agent at large spatial scales in most Western forests, altered spatial and temporal patterns of fire occurrence can affect vegetation, soil, water, and wildlife resources (Schmoldt et al. 1999). Throughout this paper, comparisons are made between burned vs. unburned and harvested vs. unharvested situations. For the purposes of decision making, these comparisons can also be considered in the context of historical conditions, and anticipated effects of a warmer climate that reduce the value of historical conditions as a reference point for management.

In some dry forests, dense stands of ponderosa pine and understory species such as grand fir have developed following removal of larger pines and/or fire exclusion (Stein 1988; Taylor and Skinner 1998). Outbreaks of some insect species and fungal pathogens are now more common as a result of these compositional and density changes (Harvey et al. 2000). Dense forest canopies and horizontal and vertical fuel continuity tend to favor crown fires rather than the low-intensity surface fires that historically occurred in many of these forests. In some moist forests, fire exclusion has modified dominance from early- and mid-seral species to mid- to late-seral species; nevertheless, when they do occur, fires in moist forests continue to be large, crown-fire events.

In the following sections, we describe the conditions that prevail following large, severe fires in Western conifer forests. In recent years, many fires >10,000 ha have occurred in western North America, with an extreme example being the 200,000-ha Biscuit Fire of 2002 in southern Oregon (USA[1]). There is no such thing as a "typical" large fire, and as noted above most large fires contain a spatial mosaic of patches burned by fire that differ in intensity and severity in terms of overstory injury and mortality.

[1] All locations hereafter are in the United States unless otherwise noted.

Effects of Postfire Timber Harvest on Forest Ecosystems

Effects of postfire timber harvest on ecosystems may be influenced by many factors, including management-induced outcomes such as: (1) number of living and dead trees and their spatial pattern following harvest, (2) ground disturbance caused by equipment and road use, (3) post-harvest fuel treatment, and (4) in some cases grass seeding and placement of various structures and materials to mitigate fire effects and timber harvest. Number of trees removed is a management decision and varies from highly selective to removal of all trees.

Ground disturbance depends on the yarding system used for timber harvest (McIver and Starr 2001). Tractor logging generally causes the most soil disturbance (although newer tractor harvest systems with "cut to length" technology cause less erosion than traditional systems), followed by cable yarding systems that fully or partially suspend logs above the forest floor during extraction (Fig. 2), followed by helicopter logging, which lifts logs out of the forest with a minimum of soil disturbance (Rice et al. 1972; Dykstra 1976). Postfire timber harvest may or may not be accompanied by road construction, and postfire fuel treatment and mitigation may be minimal or extensive. The intensity of each of these actions can influence the effects of postfire timber harvest on ecosystems.

Spatial scale issues make it challenging to accurately infer effects of ecological disturbance and management activities on ecosystems (Peterson and Parker 1998). For example, Chou et al. (1994a) found that measuring the combined effects of fire and management activities on soils and water was difficult because (1) erosion and sedimentation was in some cases high at the scale of forest stands but undetectable at the watershed scale, and (2) it was difficult to distinguish fire effects vs. management effects. In addition, the proportion of a given landscape that is harvested following fire is usually relatively small, at least on public lands. For example, of the 21,000 ha burned in the School Fire (Washington) in 2005, timber was harvested from 2500 ha, or 12% of the total fire area (Information from J. Plesha, Washington Department of Natural Resources [WDNR], Olympia, Wash.; D. Kuehn, WDNR, Ellensburg, Wash.; Powell, Umatilla National Forest, Pendleton, Ore.). Of the 200,000 ha burned in the Biscuit Fire, timber was harvested from 1400 ha, or 0.7% of the total fire area (Information from J. Fertig, Siskiyou National Forest, Medford, Ore.).

In this paper, we use a conceptual model of forest growth and succession to describe temporal patterns of overstory development, understory development, standing dead trees (snags), and woody debris (≈ surface fuels) following postfire timber harvest (Fig. 3). This model describes general ecological trends under different postfire management scenarios, based on our understanding of the literature on the effects of fire and timber harvest on forest ecosystems (described below). It is intended to complement rather than replace explicit models of forest succession and development (e.g., Oliver and Larson 1996; Franklin et al. 2002), and parameters displayed in our conceptual model are likely to vary among different forests and disturbances.

Vegetation

Fire effects on vegetation

Initial responses of vegetation to wildfire within a given forest type depend on pre-fire vegetation, seed banks, and the ability of plants to colonize the post-burn environment. Rowe (1981) developed a life history classification (Table 4) to describe postfire responses of plant species (also see Kauffman 1990). After severe fires, *invaders*, *endurers*, and *evaders* are likely to be the dominant species. Species with no fire adaptations (*avoiders*) may be locally extirpated from the site for undetermined periods. Various successional sequences may occur depending on pre-fire vegetation, site conditions following fire, postfire management actions, and seed sources. For example, Stickney (1986) recognized four vegetation stages on the Sundance Fire (Idaho) of 1967 (Fig. 4).

Western conifers fit these classes well. Many of the historical low-severity fire regimes in dry forests were dominated by *resisters* like ponderosa pine or Douglas-fir because of their thick bark and high, open crowns. In mixed-severity fire regimes, *evader* species with serotinous cones (e.g., knobcone pine, *Pinus attenuata*) are common. In historical high-severity fire regimes with long fire-return intervals, *evaders* (e.g., lodgepole pine) and *avoiders* (e.g., subalpine fir) are present. Following severe fires, only the evader species may be able to codominate early successional vegetation. In historical low-severity fire regimes with only resister-type conifers, successful regeneration of conifers following severe fires can in some cases be difficult without replanting.

Opportunistic understory species that sprout or germinate from seed following heating, removal of surface organic material, or exposure to light tend to dominate the first few years following fire (Fig. 3). Germination of some tree species, such as pines, Douglas-fir, and western larch (*Larix occidentalis*) is facilitated by exposure of mineral soil. Postfire seedling densities may be high when a reliable seed source occurs in conjunction with favorable soil and weather conditions (e.g., Donato et al. 2006); seedling densities may be low under poor conditions, especially if competition with other plants is high, resulting in low germination and/or high seedling mortality over time. In practice, planting is often done following fire if the management objective is prompt regeneration with a desired species or mix of species (Sessions et al. 2004). With a sufficient fire-free interval, shade tolerant species (e.g., fir species) typically dominate the understory (Fig. 3).

Tree mortality and dead wood dynamics

Tree mortality is an immediate effect of large, severe fires, with most mortality occurring within two years of the fire (Peterson and Arbaugh 1986). Crown fire kills foliage, buds, and cambial tissue, often consuming leaves and small branches. Hot surface fires, with or without crown fire, can do the same thing through radiative heat transfer, and can also kill surface roots. When a tree is killed by fire, dead needles not consumed by fire fall to the forest floor within a year or two, followed by branches and eventually boles.

Snags provide many important habitat features (described later) as well as long-term delivery of carbon and nutrients to the forest floor when they fall. Persistence of snags varies greatly depending on species, size, and site conditions. Timber harvest typically modifies snag dynamics by lowering density and average size through removal of dead trees (size distribution varies from site to site) and by altering site conditions such as soil bulk density and wind flow. Following fires in the Blue Mountains of Oregon, snag biomass on harvested areas was generally <50% of that on unharvested sites (McIver and Ottmar 2006). Following fires in Idaho, the predicted half-life of ponderosa pine snags was 7-8 yr (harvested) and 9-10 yr (unharvested), and predicted half-life of Douglas-fir snags was 12-13 yr (harvested) and 15-16 yr (unharvested) (Russell et al. 2006). In this case, postfire timber harvest affected the persistence of snags, at least partially because the remaining snags were smaller.

Trees adjacent to but not directly killed by crown fire may be at increased risk of attack by insects. This second-order fire effect can significantly increase mortality in the years following fire. Low-vigor trees with higher levels of crown injury (usually measured by crown scorch) or that have been subjected to basal heating (forest floor consumption, basal charring) are more likely to be attacked. *Dendroctonus ponderosae* and *Ips* spp. are major threats to pines, *Scolytus* spp. attack true fir, and *D. pseudotsugae* attack Douglas-fir. Populations can build up over the early postfire years to levels at which trees undamaged by fire inside or outside the fire perimeter may be attacked. In severe fires, many trees will have been killed directly by the fire, but bark beetles may significantly increase mortality in portions of the fire where canopy fires did not occur and in unburned islands or areas surrounding the fire (Furniss and Carolin 1977; Edmonds et al. 2005).

Following large fires in Yellowstone National Park (Wyoming) and adjacent areas in 1988, Amman and Ryan (1991) found that on permanent sample plots in areas without canopy fire, nearly 50% of the lodgepole pines were killed by *I. pini*, 70% of the Douglas-fir were killed by *D. pseudotsugae*; 80% of the Engelmann spruce were killed by *D. rufipennis*, and all of the subalpine fir (*Abies lasiocarpa*) were killed directly by fire (70% were infested by wood borers).

Insect outbreaks vary considerably as a function of postfire tree vigor, soil moisture, and proximity to existing outbreaks (Edmonds et al. 2005). The probability of insect attack cannot be reliably quantified, because cause-effect relationships for postfire insect hazard have not been rigorously evaluated at large spatial scales or across a large number of fires. While insect outbreaks can increase mortality and accelerate deterioration of standing trees, high populations of insects provide a major food source for insectivorus bird species.

From a wood utilization perspective, trees killed or badly damaged by fire begin to deteriorate immediately, and the commercial value of fire-killed trees declines as the wood progressively decays and/or blemishes. Deteriorating agents include insects, stain fungi, decay fungi, and weather (Lowell et al. 1992). Insect and woodpecker activity provides a mechanism for introducing fungal agents into the sapwood (Farris et al. 2004). Once initiated, sapwood decay progresses quickly, commonly moving ahead of any insect damage. Insects and staining fungi generally reduce product grade (and value) but seldom render the wood unusable. Decay fungi significantly affect wood properties, especially strength, thus reducing the volume of wood that can be converted into structural products, and given enough time will eventually make the wood unusable. Weather deterioration commonly causes longitudinal splits (or "checks") in the wood by differential shrinkage. Checking occurs predominantly where the bark is missing or thin, or on the ends of logs after a tree has been felled and bucked. It is usually minor in large standing dead trees and often affects only the upper part of the tree where bark is thinner, although the checks may provide an entry point for decay fungi. Smaller trees or species with thin bark may be affected more extensively, reducing the grade of smaller logs (Lowell and Cahill 1996). Checking may be extensive at an advanced decay stage, even in large trees, when tree bark begins to slough.

Deterioration in a Douglas-fir killed by fire is illustrated in Fig. 5 (Kimmey and Furniss 1943). The progression is similar in other softwood species, although decay rate varies considerably; decay rate of Douglas-fir is slower than for ponderosa pine and grand fir and much faster than for subalpine fir, Engelmann spruce, western larch, and lodgepole pine (Hadfield and Magelssen 2006). In Fig. 5, sapwood is invaded by insects during the first year following fire and quickly becomes infected with staining fungi. Sapwood of species that are susceptible to staining, such as ponderosa pine, is often completely stained within a few months. Decay fungi also infect the sapwood within the first year, and the sapwood usually deteriorates beyond commercial use by the second or third year (Kimmey 1955). By the second year the heartwood becomes infected with decay fungi, and heart rot begins to spread. Wood borers may also move into the heartwood, although damage from these insects is seldom extensive and usually results in grade reductions rather than volume loss. The tree in Fig. 5 may lose 15 cm of heartwood and all of its sapwood by 6 to 10 years following fire. The bark then falls off, and the sapwood begins to break up and slough. The rate of volume loss is considerably smaller for large, old-growth logs (Fig. 6). Actual decay rate for any species depends on factors such as local precipitation and temperature.

Fire-killed trees exhibit a higher rate of breakage when felled or extracted as compared to green trees (Lowell et al. 1992). This may be due to brittleness of wood from loss of internal moisture, lack of foliage and undergrowth to cushion the fall (because it has been burned away by the fire), and weakened sapwood resulting from insect or fungal activity. Breakage results in direct volume loss and may result in logs being cut shorter than desired, thereby reducing value.

Effects of postfire harvest on vegetation

The effects of postfire management on vegetation recovery are complex, because multiple successional pathways (*sensu* Frelich and Reich 1995) exist within myriad forest species and structures that can be burned by severe wildfires. If management activities such as timber harvest and mitigation for the effects of fire and timber harvest occur, then additional pathways are possible for both vegetation and fuels. Data from some forest ecosystems are available that allow for generalizations about vegetation development, biomass accretion, and fuel succession following postfire timber harvest (e.g., Fig. 3).

Regardless of timber harvest systems, harvesting trees can reduce natural conifer regeneration, at least in the short term, if it occurs *after* significant tree establishment has occurred (McIver and Starr 2001), although soil-disturbing activities can in some cases enhance establishment of species that require mineral soil (Sessions et al. 2004). Decreased tree regeneration following postfire timber harvest has been documented in at least two cases (Roy 1956; Donato et al. 2006), although the functional effect may depend on residual regeneration, subsequent establishment and mortality of seedlings (affected by factors such as understory competition and seed availability), and timing of timber harvest (same season as fire vs. subsequent years; winter vs. other seasons) (Newton et al. 2006).

In an analysis of successional pathways and understory diversity in Douglas-fir/hardwood forests in California, Stuart et al. (1993) found lower forb and shrub cover on sites that were burned and harvested than on sites that were burned and not harvested, although this was only two years following treatments. In contrast, hardwood cover was higher and shrub cover lower on burned, harvested sites than on burned, unharvested sites 12 years following treatments. Douglas-fir regeneration (which included planting) was inhibited in both cases, but by different competing understory species. In the same forest type, Hanson and Stuart (2005) found that harvest of burned forest facilitates an increase in native, invader species at the edge of unburned, adjacent sites, with the influence of the edge extending 15 to 30 m; forest interior species were not affected by timber harvest.

Post-harvest slash treatment may be expected to affect post-wildfire/post-harvest plant succession as suggested in Fig. 3, because it is a ground-disturbing activity that can kill colonizing vegetation. The effect is to (1) encourage more (native and non-native) invader species, (2) reduce emergent evader species before they can develop another seed crop, (3) probably have a neutral effect on endurer species that can resprout, and (4) reduce emergent conifer regeneration that can potentially have a long-term effect if a reliable seed source is not nearby (Agee 1993). It is possible these effects can be mitigated if tree planting is part of the post-wildfire management plan.

Grass seeding as part of Burned Area Emergency Rehabilitation (BAER, a U.S. federal program that strives to mitigate for effects of severe fire on erosion and other landscape elements following fire) or post-harvest mitigation may be justified to address short-term erosion, but may have negative impacts on native flora and conceivably could have negative effects on long-term productivity. Native shrubs that rely on a seed bank for postfire recovery can be nearly extirpated if seeded grass cover exceeds 40% (Schultz et al. 1955). Heavy seeding (up to 157 kg ha^{-1}) following wildfire in the Sierra Nevada (California) produced first-year cover of 95% but reduced native species richness by nearly 50% (Keeley 2004), and natural regeneration of ponderosa pine was reduced by an order of magnitude.

Seed-bank species have a difficult time competing with aggressively seeded non-native species, and effects on locally extirpated native species can persist even if non-natives decrease after the first year. In eastern Washington, natural regeneration of conifer seedlings declined by 50%, and cover of seed-bank species such as snowbrush and native sprouting species decreased sharply in areas where emergency grass seeding (50 kg ha^{-1}) was done (Schoennagel and Waller 1999). Because N is volatized by fire, and is often a limiting element for plant growth, inhibition of nitrogen-fixing taxa such as *Ceanothus* spp., *Lupinus* spp., *Lathyrus* spp., and *Vicia* spp. following fire may have long-term consequences for site productivity (Newland and DeLuca 2000).

Fuels

Fire effects on fuel dynamics

Fuels and potential fire behaviour in forests burned by high-severity fire change over time – with or without timber harvest – as suggested by our conceptual model (Fig. 3), and a wealth of empirical data (e.g., Graham et al. 2004). Postfire fuelbeds typically have a shrub or herbaceous component, fine woody fuels that fall from the crown. Surface fuel dynamics following wildfires are a function of (1) prefire live and dead biomass, (2) time, because standing fuels decay and become surface fuels, and (3) events following fire such as timber harvest, windstorms, and delayed mortality due to insects (Agee 1993; Schmoldt et al. 1999). Data on all potential fuels, including herbs, shrubs, and dead woody fuels, contribute to fire potential. Fuel mass, fuelbed depth, and moisture content of available fuels all contribute to definitions of the fuel complex. Larger fuels (>7.6 cm [3 in] diameter) are generally not considered in fire behaviour prediction systems (e.g., Rothermel 1972; Andrews 1986) but do affect spotting of firebrands, long-term smouldering, and long-term productivity.

Larger fuels (>7.6 cm diameter) follow a predictable pattern after crown fire (Agee 2002a) (Figs. 3, 6). Following initial mortality, snag biomass decreases over time (Agee and Huff 1987; Spies et al. 1988), and smaller snags usually fall first (Lehmkuhl et al. 2003). Log biomass correspondingly increases for decades, especially in dry and cold environments where decomposition is slow, but may decline as the original snags that fell and created those logs begin to decay; log biomass from self-thinning in the new forest consists of smaller, more easily decayed material (Harmon et al. 1986). As noted above, harvest of a portion of the snag component reduces the snags left to fall, and therefore the mass of downed logs, relative to unharvested sites over time (McIver and Ottmar 2006). Rates of downed wood accumulation vary greatly depending on site productivity, general climate (dry and/or cold – slower decomposition; wet – faster decomposition), and local microclimate.

Understory response by forbs and shrubs is a less appreciated factor in postfire fuel dynamics. If the fire creates a favorable environment for annual grasses, or if annual grasses are seeded as an emergency rehabilitation response or introduced in straw mulch, a substantial fine fuel can be present within a year of the fire, and the potential for subsequent fire exists (Zedler et al. 1983). Most forested sites respond with growth of shrubs and other perennial species, and flammability depends on surface area-to-volume ratios and moisture content. If response is by shrubs and perennial grasses, moisture content generally remains high enough during the fire season that potential fire behaviour may be reduced (Agee et al. 2002; Raymond and Peterson 2005) (Fig. 8). Even within a single fire, the pattern of understory dominance may be mixed (Stickney 1986). If herbaceous-dominated areas contain persistent, fine dead fuel such as bracken fern (*Pteridium aquilinum*), straw mulch, or grasses contained in straw mulch, these fuels may have a strong influence on fire behaviour.

The forest developing after a stand-replacing fire will over time also constitute a fire hazard because trees can act as part of the understory fuelbed. As crowns emerge from the shrub layer, the low canopy base height will create torching potential (cf. Scott and Reinhardt 2003). If the stand is dense (e.g., 10-cm dbh trees at a density of >1,200 ha^{-1}), canopy bulk density may be high enough (>0.12 kg m^{-3}) to carry independent crown fire under severe fire weather. Canopy base height will eventually increase, reducing torching potential.

Fuel dynamics can also be affected by site productivity. Following wildfire, an old forest with limited biomass due to low soil fertility generally has lower fuel mass than a similar-aged forest on a site with high soil fertility. Fuel dynamics for a chronosequence of wildfires without postfire harvest was studied in the Olympic Mountains (Washington) Agee and Huff (1987) (Table 5). Fine fuel mass at this productive site was higher than short-term fine fuel masses immediately following fire on drier sites (Table 5). In southwestern Oregon, sites burned with high-severity fire had lower fine fuel loads than unburned sites, but on the Olympic site, fuel mass in the first year postfire was twice that of unburned forest. Highest fine fuel loads occurred in the first year postfire due to branches being snapped off the canopy by a windstorm the first winter following fire. Without that storm, fine fuels would probably have peaked much later with a broader, flatter peak over time.

Some historical evidence supports the idea that recent fires block the spread of subsequent fires (i.e., reducing reburn potential) in low- and mixed-severity fire regimes. In ponderosa pine forest, large historical low-severity fires appear to be followed by fires of smaller extent (Everett et al. 2000), implying a fuel-limited effect on fire spread. Wright and Agee (2004) documented a similar phenomenon in mixed-conifer forest where historical (pre-1900) fires acted as barriers to subsequent fires. In Douglas-fir forest with a mixed-severity fire regime, Taylor and Skinner (2003) showed that most fires burned different sites than the preceding fire, although they cautioned that lack of fire-scar evidence does not necessarily indicate that fires did not burn there. In another mixed-severity fire regime, van Wagtendonk (1985) showed for red fir (*Abies magnifica*) forest in Yosemite National Park (California) that many natural fires stopped at the boundary of previously burned areas, creating a mosaic of forest with different times since the most recent fire.

In the Entiat River watershed in the Cascade Range (Washington), dry forests burned by severe wildfires in recent decades may be less flammable than unburned forest. A series of large fires has burned much of the dry forest in the lower watershed since 1970 (Fig. 9), yet the pattern of fire spread does not support reburn potential in previously burned areas. It appears that previously unburned areas have the highest potential for burning, and that fires have been stopped adjacent to or within old burn boundaries.

Fuel conditions following high-severity wildfires were recently quantified along a chronosequence in the Cascade Range (Washington): (1) age 1 yr, Fischer fire (Wenatchee River watershed, adjacent to the Entiat watershed), (2) age 11 yr, Tyee fire, (3) age 17 yr, Dinkleman fire, and (4) age 35 yr, Entiat fire) (data from J. Agee, University of Washington, Seattle). Fuel loadings, based on surface fuel models (*sensu* Scott and Burgan 2005) were analyzed on dry and mesic sites within each burn, and on areas with and without timber harvest. Compared to unburned forest, fire behaviour estimates under local worst-case fire weather (Fig. 10) reinforce the visual impression in Fig. 9. Harvest treatment on dry sites maintained low rate of spread and flame length for at least 35 years, in contrast to other treatments. The old-growth forest had high torching potential (*sensu* Scott and Reinhardt 2003) with higher potential rate of spread and flame length than recently burned locations. Burned sites had a cover of perennial grass and shrubs, with lower potential rates of spread and intensity until new tree cover emerges as forest canopy (20+ yr), at which time torching potential again increased due to low canopy base height of the young forest. Postfire age appeared to be more important to fire danger than whether or not timber harvest occurred.

Some dry forests have the potential to reburn at relatively short (<20 yr) intervals. For example, an analysis of fire hazard conducted after the Cerro Grande Fire (New Mexico) of 2000 concluded that fire hazard would begin to increase only six years following the fire (Greenlee and Greenlee 2002). This was attributed to dead trees falling and contributing to accumulation of surface fuels, particularly in areas that experienced high fire intensities and high tree mortality. A recent analysis of wildfires in the Umatilla National Forest (Washington), an area dominated by dry ponderosa pine and mixed-conifer forests, documented 18 locations that reburned within 20 years of a previous fire (Data from D. Powell, Umatilla National Forest, Pendleton, Oregon). Twelve of these locations reburned within 10 years of the previous fire. Although local patterns of fire severity may have varied, these data indicate that some dry forests have sufficient fuels to experience multiple fires at relatively short intervals. In addition, large wildfires (>4000 ha) were found to occur at intervals of 15-20 yr across any given location in pine-dominated forests of northeastern California (Norman and Taylor 2003). Finally, the Biscuit Fire of 2002 (Oregon) completely burned through the Silver Fire of 1987, which suggests that fuels had accumulated sufficiently in 15 years to reburn, in this case in a mixed-severity fire regime.

The Tillamook fires of 1933-1951 in the Coast Range of Oregon are another example of reburn potential but in a much wetter environment. The first fire in 1933, started by timber harvest operations in old-growth western hemlock–Douglas-fir forest, spread to older slash and burned >80,000 ha. In 1939, 76,000 ha burned with 37% of this area a reburn of the 1933 fire. In1945, 73,000 ha burned with 50% of the area a reburn of the 1933 and 1939 fires. In 1951, 13,000 ha burned with 100% of this a reburn of areas originally burned in 1933 and 1939 (Oregon Department of Forestry 1983). Unfortunately fuel conditions in the burned forest areas were not well documented. Repeated harvest operations were conducted, and one-third of the timber killed in the 1933 through 1945 fires was harvested up to the time of the 1951 fire. Reburn areas were a mix of harvested and unharvested lands. Similar reburning occurred after the 1902 Yacolt burn in southwest Washington (Gray and Franklin 1997). These coastal forests, with bracken fern as a fuel, have significant reburn potential.

Reburn potential depends on many factors including local biophysical environment, fuel conditions, and ignition circumstances, and it is difficult to generalize about this topic even for dry forests. Historical data on burn frequency and extent are helpful in determining the intervals at which a subsequent fire might be expected following a wildfire, and simulation modeling (e.g., Reinhardt and Crookston 2003) can be used to estimate fuel accumulation, fire hazard, and potential fire behavior over time.

Effects of postfire harvest on fuels

Postfire timber harvest can overlay at least two patterns of variability on the unharvested condition: (1) harvest without regard to removal of fine fuels (<7.6 cm diameter), and (2) harvest with some form of fine-fuel treatment (Figs. 11). Using actual fuel loads for three stand treatments in the Blue Mountains (Oregon), McIver and Ottmar (2006) projected fine fuel loading over 30 years (Fig. 12). The two harvest options utilized whole tree yarding, but fine fuels created in the harvest operation increased fine fuel loading above that in the unharvested stand, and this increase persisted over the 30-year period, although levels in all three treatment were similar after 20 years. In the Olympic Mountains (Washington), Agee and Huff (1987) found that surface fire behaviour potential was highest in early seral stands (age 1-20 yr postfire) in burned forest that was not harvested.

Because postfire timber harvests occur at different locations and different times and apply a variety of standards, effects on fuels and subsequent fire behaviour may be specific to local conditions. Trees can be whole-tree harvested, different size classes of trees can be removed, and remaining fine fuels can be treated or not. Each of these treatments can affect the amounts and timing of fuels reaching the ground that can contribute to surface fire behaviour in the burned forest. If generalizations are made, it is important to qualify them with the fuel dynamics specific to a particular site.

In the case of the Tillamook fires (discussed previously), slash from the harvest operations likely helped to carry some of the subsequent fires, but two other documented sources of fire spread were also responsible for reburns. First, snags left from previous wildfires acted as "candles," allowing fire to spread from snag to snag independent of surface fuels (Oregon Department of Forestry 1983). Second, there was a vigorous understory response from bracken fern, which is tall (often >2 m), semi-woody, and deciduous. The fluffy fuelbed from the previous year's growth of fern allowed surface fires to move quickly across previously burned landscapes (Isaac 1940). Woody fuels added from either harvested or unharvested trees were not necessary for fires to spread.

The Tillamook cycle of reburns was broken after the 1951 fire. More than 300 km of snag-free fuelbreaks were constructed, with widths ranging from 300 to 1200 m. Some snags were harvested, and others were dropped and left on site, but tree removal was not the only action that ended the cycle of fire. Better road access, fire lookouts, and fire prevention (all Tillamook fires were human caused) also contributed to changing the fire pattern. Although postfire timber harvest may have helped end the cycle of reburns, the effects of harvest operations were not well quantified.

In dry forests of interior western North America, large woody debris was limited historically by frequent fire that consumed logs, so these types of forest maintained much lower levels of large woody debris than in mixed-severity or high-severity fire regimes

(Skinner 2002; Agee 2003b). After a century of fire exclusion, the occurrence of stand-replacement wildfire leads to unprecedented levels of large woody debris in these forests. For example, historical log biomass was around 5 Mg ha^{-1} for dry ponderosa pine forest (Agee 2002a). Following a stand-replacement wildfire in a contemporary ponderosa pine forest, large woody debris of 10 to 100 Mg ha^{-1} may be present (Agee 2002a; McIver and Ottmar 2006; Passovoy and Fulé 2006), with even higher woody debris in Douglas-fir forest with a mixed-severity fire regime (Wright 1998).

In these forests, restoration of open forest conditions (and surface fire regimes) following crown fire may require fuel reduction. One management option is to use prescribed fire to maintain low fuel conditions. However, logs and residual large woody debris are often dry enough that a prescribed fire will ignite and burn much of the large woody debris (Agee 2002a). Heat from the smouldering remains of logs can penetrate the soil and kill roots of understory plants and trees. There will be fewer logs and less mass of logs on sites where harvest occurred following fire (with similar initial conditions), as projected in Fig. 3.

The potential for abundant large woody debris to produce lingering smoke may also limit use of prescribed fire. Fire prescriptions designed to conserve large woody debris have such a narrow window that they are rarely practical to implement (Agee 2003a). Another concern posed by large woody debris is additional resistance to control of wildfires (USDA Forest Service 1976; Brown et al. 2003). Loading as low as 30 Mg ha^{-1} of 7-25 cm diameter material may result in an "extreme" rating if fine fuels are >30 Mg ha^{-1}; 90 Mg ha^{-1} are required for an extreme rating if fine fuel loads are <11 Mg ha^{-1}. If many logs >25 cm diameter are also on site, less of the 7-25 cm material is required to reach these ratings.

Fireline construction and firefighter safety may be an issue where large amounts of large woody debris exist. For example, in the Tyee Fire (Washington) of 1994 which started from lightning in a location burned in 1970 (but not harvested), most of the snags had fallen and were scattered randomly, making access to safety zones difficult for firefighters.

Soils and Hydrology

Fire effects on soils and hydrology

Intense wildfire typically volatilizes soil C in the forest floor and surface layers of mineral soil (Dyrness et al. 1989). Organic matter ignites at 260 to 425°C (DeBano 1979), and C is lost as gaseous oxides and particulate matter. Such ground temperatures are common in forest fires (Neary et al. 1999). Heat duration and depth of penetration depend on fuel characteristics (loading, size, arrangement, moisture content), weather conditions (temperature, wind speed, humidity) and fire behaviour (rate of spread, flame length, intensity, duration) (Neary et al. 1999). Soil moisture also affects heat transfer; soil temperatures at 5 cm depth can reach 200°C and last for more than 1 hr if soils are dry, while temperature and duration of heating for moist soils are much lower (Busse et al. 2005).

Heating effects of fire on soil depend on temperature and duration of heating and can range from increased plant nutrient availability (Haase and Sackett 1998) to reduced nutrient capital, biota, and physical condition (Neary et al. 1999). In general, 60°C is lethal to biotic activity (Busse et al. 2005). Organic N begins thermal oxidation at 200°C, and most has been volatilized by 500°C (DeBano 1979). In general, N volatilization is a linear function of the amount of material consumed by fire (McColl and Powers 1984). Temperatures below N volatilization levels can produce ammonium from thermal ammonification, protein denaturation, and death and turnover of soil microbes. This leads to a temporary flush of soil fertility (including cations) that typically lasts for <5 years or so. In a chronosequence study of burned and unburned stands in Montana, soil N mineralization potentials were lower in underburned stands more than a decade after treatment (DeLuca and Zouhar 2000).

Fire-induced soil water repellency (hydrophobicity), a byproduct of temperatures between 176 and 288°C (DeBano 1981), occurs as hydrophobic substances produced during the volatilization of organic matter migrate downward along a steep temperature gradient and condense about colder soil particles. In part, such substances may result from thermal loss of oxygen-containing functional groups during the pyrolysis of humic substances in this specific temperature range (Almendros et al. 1990), and such substances seem to be destroyed at higher temperatures (Doerr et al. 2004). Hydrophobicity impedes water infiltration into the soil profile, and unwettable layers create a functionally shallow soil. As water accumulates to fill pore spaces, pore pressure increases and soil shear strength decreases until the soil mass moves through the force of gravity (DeBano 2000). Hydrophobicity is transient. A study in the Front Range of the Colorado Rocky Mountains following wildfire found that water repellency weakened progressively over a year and diminished as soils became wetter (McDonald and Huffman 2004). Unburned sites showed non-fire related water repellency when soil moisture was <10% and lightly burned soils were repellent to soil moisture of 14%. However, repellency persisted on severely burned sites even at soil moisture contents of >26%. Activities that disturb the soil surface, such as timber harvest, can in some cases disrupt hydrophobic layers and improve infiltration (Poff 1989).

Fire affects hydrology through the removal of aboveground canopy, removal of litter, and, in some cases development of a hydrophobic layer. Direct measurements of the effects of fire on hydrology are rare, while the effects of vegetation removal are well studied and can be used cautiously to infer fire effects (Neary et al. 2005c). Bosch and Hewlett (1982) reviewed 94 studies on the effects of vegetation changes on water yield. Stednick (1996) updated this review, and studies continue to be reported (e.g., Troendle et al. 2001). Studies have involved rain-dominated and snow-dominated hydrologic systems and deciduous and coniferous forests, and although results of these studies vary considerably, some general inferences can be made.

Removal of living forest overstory increases water yield, although increased water yield is not detectable unless 20 to 40% of watershed or forest basal area is removed. As the percentage of forest removed increases beyond the detection threshold, water yield increases proportionally. Magnitude of increased water yield is related to total annual precipitation, and larger water yield increases are observed in wet years than in dry years. Absolute increases in water yield reported in the literature range from negligible to >700 mm yr^{-1}. In western North America, water yield increases in coastal forests are greater than those in dry, interior forests.

Water yield typically increases significantly in the first year following fire or timber harvest, then decreases with time as vegetation reoccupies a watershed. This “hydrologic recovery” is directly related to site productivity, which in turn is determined by temperature regime, moisture regime, and soil fertility. Few studies have been active for long enough to actually document recovery, but estimates have been made from trajectories of declining water yields. Recovery may take 60 to 80 years in high elevation (ca. >1,000 m), interior watersheds of western North America. It may take as little as 25 years in coastal watersheds, although analysis at Alsea Experimental Watershed (Oregon) revealed that mechanisms of stream flow generation and routing had not recovered 28 years following clearcut timber harvest though water yield had recovered (Stednick 1996). Effects generally persist longer when the contiguous spatial extent of disturbance is greater; for example, a 10-ha clearcut may take longer to recover than 10 1-ha clearcuts.

DeBano et al. (1998) and Neary et al. (2005b,c) provided comprehensive reviews on the effects of fire on water and watersheds. The literature quantifying the effects of prescribed fire on streamflow is limited, with even less literature on the effects of wildfire. In general, when vegetation is killed and forest litter and duff are consumed by wildfire, then soil and duff water storage, interception, and evapotranspiration may be reduced. Changes in annual water yields would be about the same as those caused by forest harvest, although perhaps greater because fire would kill understory vegetation not killed by harvest. Normal patterns of snow accumulation and melt may be changed. Infiltration may be reduced for a number of reasons, which in turn may increase overland flow and alter subsurface flow pathways. Changes in these processes result in peak flow increases, altered runoff (response) timing, changes in base flow, and probable changes in water quality (Table 6).

Following fires in the Bitterroot Mountains (Montana) in 2000, the U.S. Geological Survey instrumented a number of burned watersheds with stream gauges and tipping-bucket rain gauges, including the 24.5-km^2 Laird Creek watershed of which 30% burned with high severity (Nickless et al. 2002). Following a series of thunderstorms that passed over Laird Creek on 20-21 July 2001, some rain gauges in the vicinity received little rain (Parrett et al. 2004) which is typical of small storm cells. However, two of the rain gauges in Laird Creek received rain with 10-25 yr recurrence intervals for 5-, 10- and 15-min durations. The consequent runoff events (Fig. 13) produced nearly identical flood peaks on Laird Creek.

The recurrence interval for the peaks was 200-500 yr based on unburned conditions (Parrett et al. 2004), and the floods triggered significant debris flows (Fig. 14).

Effects of postfire harvest on soils and hydrology

Burn severity in soils varies depending on fuels, long- and short-term weather, burning conditions, and suppression efforts; burn severity in soils is often independent of burn severity in trees (Neary et al. 2005c). Fire-induced changes in the forest floor can range from relatively intact forest floor conditions where surface organic horizons (litter, humus, rotten wood) are relatively thick following fire, to no surface organic matter present (Graham et al. 2004; Jain et al. 2004; Peterson et al. 2005).

Literature on specific effects of postfire management on soils centers on erosion impacts of road construction and sensitivity of the soil when the surface has been bared by fire (Beschta et al. 2004; McIver and Starr 2001; McIver and McNeil 2006). Skidding logs across bare ground clearly increases soil disturbance over other methods (Klock 1975) and compacts soil (Beschta et al. 2004), particularly when soil moisture is near the plastic limit. Soil compaction can be detrimental to plant growth (Froehlich and McNabb 1984). However, the degree to which soil compaction is detrimental is largely a function of soil texture; soil quality is degraded on clayey soils by increasing strength and reducing aeration porosity, and is enhanced on porous sandy soils by improving soil water availability (Gomez et. al 2002; Powers et. al 2005). The immediate concern about soil compaction from postfire timber harvest is that the size of surface soil pores can be reduced, and infiltration can be impeded, potentially causing increased surface erosion. However, the presence of even a thin litter layer can reduce soil erosion substantially. In a field study on a clayey soil in

California, erosion rates increased significantly when the forest floor was experimentally removed, but rates were not increased when the forest floor was present (Powers 2002). When soils are bared by wildfire, slash produced during timber harvest may create soil cover that can reduce soil erosion by up to 95% (Shakesby et. al. 1996).

Large losses of C occur during wildfire, but the greatest effect on soil fertility is caused by the loss of the forest floor, the greatest single reservoir of above-ground N and other nutrients. Although removal of standing vegetation has little impact on soil N availability, that availability declines substantially when the forest floor is lost (Powers et al. 2005), and even moderate surface fires can reduce potential soil N availability (DeLuca and Zouhar 2000).

As noted above, different timber harvest systems can have different effects on soil compaction and fertility. Tractor logging and ground-based equipment on relatively level areas (ca. <30% slope) cause the most soil compaction, although some effects can be mitigated by avoiding wet soils and by operating over slash rather than areas with thin forest floors. Cable yarding systems also cause soil compaction, which is typically localized and potentially causes more erosion where logs are dragged upslope. Skyline yarding, a form of cable yarding that suspends logs above the ground, avoids most physical abrasion of the forest floor and mineral soil (Fig. 2). Helicopter logging greatly reduces soil impacts by minimizing movement of logs along the ground.

Forest roads present special challenges in the postfire environment. As noted earlier, stream flows can be expected to increase for the first several years after a forest fire, because vegetative cover has been reduced, and because severe fires may char the surface layer to form a hydrophobic surface that accelerates runoff (DeBano et al. 1998). As a result, culverts that are sized correctly for drainages with intact vegetation may prove inadequate to handle increased runoff. In addition, streams are likely to carry more woody debris and sediment than usual (Klock 1975; McIver and McNeil 2006), increasing the likelihood that culverts may become plugged during storms. This can lead to culvert failure, greatly increasing the quantity of sediment delivered to streams. Even when culverts do not fail, plugging can result in temporary diversion of stream channels and increase sedimentation. Where streams become diverted across roads, increased erosion of road surfaces, cut banks, and fill slopes is almost certain (Furniss et al. 1998; Keller and Sherar 2003).

Postfire timber harvest may change the timing of runoff, with outcomes that are either detrimental or beneficial depending on management objectives. Peak discharges may increase (usually detrimental in terms of flooding) or decrease due to desynchronized snow melt or storm flow (Brooks et al. 2003). Similarly, low flows may increase (usually beneficial for fish) or decrease. Road networks in managed watersheds effectively become extensions of the channel network, intercepting precipitation, not allowing infiltration of snow melt or precipitation, and intercepting subsurface flow paths (Wemple et al. 1996; Jones et al. 2000). Most "small watershed" studies in the literature cover land areas <1000 ha. Not all studies have assessed peak flow changes, and while many report increases of 50 to 100% following removal of overstory vegetation, some report decreases or negligible change (Bosch and Hewlett 1982; Stednick 1996). It would be more useful if increases in peak discharge could be expressed in terms of recurrence interval, but most studies are not of sufficient duration to develop reliable pre- and post-disturbance flood frequency information.

Forest harvest following wildfire may alter water quality, with sediment and stream temperature being the biggest concerns. Sediment can be produced by harvesting and site preparation and delivered to streams (McIver 2004), but the vast majority of it may be expected from new road construction, road maintenance, and failure of culverts and roads. Stream temperatures are altered primarily by loss of riparian canopy or shade (Reeves et al. 2006), but may also be influenced by conduction and advection of water warmed while flowing through areas where canopy has been removed (Johnson and Jones 2000).

Forest roads are the largest source of erosion and sedimentation in forestry (Rice et al. 1972; Sidle 1980), and may cause as much as 90% of total erosion resulting from forest management activities (Megahan 1980). Soil erosion associated with roads is particularly severe during the first year or two after construction, before the cut banks and fill slopes have revegetated and stabilized. This may be especially important when roads are built to provide access for postfire timber harvest, because landscape capacity to trap sediments above streams can be greatly reduced following wildfire. Best-practice guidelines for engineering of forest roads focus on drainage structures and methods for controlling water on the road surface and in cut banks, roadside ditches, and fill slopes (e.g., Dykstra and Heinrich 1996; Keller and Sherar 2003).

Special considerations associated with forest roads following postfire timber harvest are summarized in Robichaud et al. (2000). Because stream flow and overland water flow can be expected to increase for several years following fire, stream-crossing and cross-drain culverts (to drain water from roadside ditches) can be upgraded in order to handle increased water quantity. This can be done even if postfire timber harvest is not implemented to prevent increased erosion and sedimentation from forest roads. Trash racks built above culverts can trap woody debris and help prevent culvert clogging. Risers installed on culvert in-feeds can allow sediments to settle out so that scouring is reduced. Armoring culverts or protecting them with geo-textiles can prevent washouts. Bales of straw placed below culverts can trap sediment so that it settles in place rather than moving downstream (Simon et al. 1994). In some cases, it may be advisable to remove stream-crossing culverts until the watershed revegetates and runoff returns to pre-fire levels.

Where fires have removed vegetation from cut banks and fill slopes, vegetation regrowth can reduce stream sedimentation. Silt fences, riprap, or gabions (wire baskets filled with rocks) can be used below cut banks or fill slopes to trap sediments. Revegetation of slopes by aerial seeding (preferably with native species) is commonly used to establish ground cover, although its effectiveness varies (Robichaud et al. 2000). Stabilization of slopes with weed-free straw or other materials (e.g., “wood straw,” Anderson and Dooley (2006) is also used to impede sediment movement until slopes are revegetated. Slash can be distributed over slopes, or chipped and spread to provide temporary ground cover.

In some cases, stream channels may be so heavily modified by wildfire and increased streamflow that significant scouring and channel instability may result. In such cases, channel treatments may be necessary to stabilize the streambed, although Robichaud et al. (2000) recommend that such treatments be used only if the downstream threat is high. Common treatments include stabilizing stream banks with large woody material, and installing check dams of straw bales, logs, or gabions. Check dams reduce stream velocities and allow coarse sediments to settle out. Straw bales may function for three months or less, whereas log dams and gabions survive longer but may aggravate erosion if they fail.

Regular maintenance and monitoring over time are essential, because mitigation efforts may be only partially effective or may themselves cause additional problems (Robichaud et al. 2000). Unsurfaced roads are vulnerable to severe erosion when wet, but even surfaced roads are more susceptible to traffic damage (e.g., postfire hauling of logs) following fire because of increased runoff and thus increased likelihood that water will flow on the road surface. Limiting access to recently burned areas can reduce erosion potential.

Road-related problems in terrestrial and aquatic systems (Forman and Alexander 1998; Trombulak and Frissell 2000; Gucinski et al. 2001; Beschta et al. 2004) are affected by road densities, topography, and climate. Effects of roads on yield, seasonality, and quality of water flow, and the lack of studies separating effects of roads from timber harvest and/or fire were discussed earlier. The key questions related to postfire road issues are: (1) How can existing roads and road systems be adequately mitigated in the postfire environment (which may include timber harvest)?, and (2) Can new road construction be adequately mitigated?

While roads can create chronic aggravations, acute impacts seem to be stochastic in nature. For example, exceptional rain-on-snow events in winter 1995-1996 on the Clearwater National Forest (Idaho) resulted in massive floods and landslides. Over 900 mass failures were evaluated (McClelland et al. 1999), with 58% road related, 29% natural, and 12% associated with timber harvest. Only 2% of the mass failures were in areas burned by wildfire within the previous 10 years, although the total proportion of the Clearwater National Forest burned during the previous decade was small. The study concluded that mass failure risk was determined by parent material, elevation, aspect, slope steepness, and landform. Most of the roads that failed were 40-50 years old and had been built to accommodate timber harvest systems outmoded by new harvest technologies. Most of the road failures were related to failures of culverts that could not handle the volume of water and debris (Gucinski et al. 2001).

Well-designed engineering structures and new harvesting technologies generally help to mitigate the effects of severe fires and timber harvest on water quality, although Beschta et al. (2004) suggested that we do not know whether cumulative degradation of aquatic habitats can be reduced and therefore advocated prohibition of new road construction in postfire environments. They acknowledged that road practices have improved over the years, but until “legacy roads” like those on the Clearwater National Forest are upgraded, roads have the potential to degrade aquatic systems for decades to come.

Postfire BAER treatments for existing roads are commonly implemented following large wildfires (Robichaud et al. 2000), although the effectiveness of these efforts has not been fully evaluated. Culvert “upgrades” are a high priority. In some cases, models like TR-55 (USDA Natural Resources Conservation Service 1986) are used to estimate culvert design capacity, but the urgency of the situation often results in “overkill,” and replacement culverts exceed expected capacity. Installation of culverts that can handle increased water flow following severe fire would preclude some problems, although planning and designing for a 200+ year event like that reported for the Laird Creek watershed may be unrealistic.

Terrestrial Wildlife

Fire effects on terrestrial wildlife

Terrestrial forest vertebrates have evolved with fire regimes in their ecosystems, thus most forest species have adaptations to escape mortality from heat and smoke by moving underground, or to seek refuge in unburned patches of forest (Lindenmayer and Noss 2006). Depending on species tolerance to postfire conditions, recolonization of a burned forest may occur within days or years. If the postfire environment provides low quality habitat for an animal, it may remain in unburned forest patches or move elsewhere until habitat conditions become favorable. Hence, recovery of populations to prefire levels depends on characteristics of the animal as well as characteristics of the postfire environment.

Despite the importance of fire as a natural disturbance for forest vertebrates, understanding of the effects of fire on species distribution and abundance is limited. Most studies involve retrospective analyses comparing burned to nearby unburned forests, instead of pre-fire versus postfire. Few studies are experimental, and we have little understanding of causes or mechanisms of observed patterns. For example, one may observe a decrease in canopy-nesting birds following fire, but without a rigorous test of what might cause the decrease in nest abundance.

Most studies generally infer causes of observed postfire patterns from life history traits and habitat relationships (Bunnell 1995; Smith 2000; Kotliar et al. 2002; Saab and Powell 2005), such that changes in wildlife abundance are generally mediated by fire effects on vegetation composition and structure. In the previous example, tree mortality and reduced canopy following fire would be a logical explanation of reduced abundance of canopy-nesting birds. It has been generally assumed that stand-replacement fires affect vegetation and animal habitats more than less severe fires (Smith 2000). However, data for most species are inadequate to quantify the effects of differences in burn severity on population abundance or persistence (Lyon and Smith 2000; Kotliar et al. 2002; Saab and Powell 2005).

In general, arboreal species associated with closed forest canopies decline following crown fires, and species associated with open forest conditions and snags increase (Hutto 1995; Lyon and Smith 2000; Kotliar et al. 2002; Saab et al. 2005). Terrestrial species dependent on shrub and herb understories for food and cover generally benefit from increased diversity of understories following fires, although species associated with woody debris may decrease in the short term until new down wood is recruited (Lyon and Smith 2000). Fires that have areas with both low-severity and high-severity effects on the overstory may maximize the patchiness and diversity of green and dead wood habitat components (Kotliar et al. 2002; Lehmkuhl 2005; Smucker et al. 2005).

Effects across large landscapes likewise may vary with extent and severity of fires, and with species habitat requirements, life histories, and scales of movement. For example, large-scale stand-replacement fires may negatively affect spotted owl (*Strix occidentalis*) populations in dry forests (Gaines et al. 1997; Bond et al. 2002), but habitat patchiness created by small stand-replacement fires (Franklin et al. 2000) or fires with both high and low severity may be beneficial. Managing for forest conditions characteristic of varied fire severities may support the most bird species, because different species respond positively to different fire severities (Smucker et al. 2005).

Effects of postfire harvest on terrestrial wildlife

Effects of postfire timber harvest on dead or green trees in low-severity or moderate-severity burns are added to the initial fire effects on wildlife habitat and poulations, and may range from positive to negative depending on animal species. Removal of green trees as part of postfire timber harvest can negatively affect arboreal species that nest or forage in tree canopies (Hutto 1995, 2006; Huff and Smith 2000; Kotliar et al. 2002) and species that indirectly benefit from ecological processes associated with green trees. Green-tree harvest affects white-headed woodpeckers (*Picoides albolarvatus*) that favor open forests created by low-severity and moderate-severity fires and nest in snags, but forage heavily on conifer seeds in live trees (Garrett et al. 1996). Potential indirect effects on northern flying squirrels

(*Glaucomys sabrinus*), a secondary cavity-nesting species in interior dry forests, result from a positive correlation between recruitment and survival rates and diversity and biomass of fruiting bodies (truffles) of ectomycorrhizal fungi that are symbiotic with green trees (Lehmkuhl et al. 2006). Likewise, squirrel survival rates are correlated with arboreal forage lichen (*Bryoria* spp., *Alectoria* spp.) biomass (Lehmkuhl et al. 2006), which is highest in large live trees (Lehmkuhl 2004). A critical unknown for most animal species is the threshold at which postfire removal of green trees, in addition to fire-caused tree mortality, has no effect on populations.

Removal of dead or dying trees can have positive, neutral, or beneficial effects on terrestrial vertebrates in forest ecosystems (Table 7). Species like Lewis' woodpecker (*Melanerpes lewis*) and American kestrel (*Falco sparverius*) that are associated with open-canopy stands following fire (Tobalske 1997) may realize immediate benefits of limited postfire timber harvest in stands with high snag density, as opposed to waiting 10 years until enough snags have fallen to create the necessary open conditions (Haggard and Gaines 2001; Saab et al. 2002). In contrast, black-backed woodpeckers (*Picoides arcticus*) and three-toed woodpeckers (*P. tridactylus*) are associated with dense stands of snags, and postfire harvest likely would be detrimental to their occupation of the site (Saab and Dudley 1998; Saab et al. 2002). Providing a mix of open and dense snag stands, either through retention of natural patchiness or by timber harvest, may increase the abundance and diversity of cavity-nesting species over the short term (ca. 5 yr) (Saab and Dudley 1998; Haggard and Gaines 2001).

Effects of postfire timber harvest on snag-dependent animal species are mediated by several factors. Snag harvest that targets tree species with little cavity-excavation value might have minimal effects on nesting potential. Tree species such as subalpine fir and lodgepole pine create hard snags that mostly fall entire (Everett et al. 1999) and are little used for cavity excavation (Lehmkuhl et al. 2003). Snags <25 cm dbh are generally too small for nests of cavity-excavating birds (Rose et al. 2001). If snags are very dense, then removal of snags down to some tolerance threshold for occupation by target species may have little effect (Mellen et al. 2002). Moreover, the density of primary cavity-nesting birds may be limited by territorial behaviour, so more snags do not necessarily lead to denser bird populations (Raphael and White 1984; Bunnell 1995), although that relationship may not hold for subsequent use by secondary cavity-nesting birds and mammals (Raphael and White 1984). In some cases, removal of small, high-density non-target tree species may remove insect food sources during the first 5 yr after fire when bark beetles and wood-boring beetles attract insectivorous birds (Lehmkuhl et al. 2003).

Postfire timber harvest can alter the abundance of large woody debris (Fig. 3), which provides important habitat for many wildlife species (Maser et al. 1979; Bull et al. 1997; Bull 2002; Bunnell et al. 2002). Large woody debris provides (1) increased abundance of insects (Koenigs et al. 2002) used by foraging birds, bears, and other insectivores, (2) denning and foraging areas for forest carnivores, (3) cover and fungal fruiting bodies for food for small mammals, (4) basking sites and cover for reptiles, and (5) moist microhabitats for salamanders. Some fires can consume substantial amounts of woody material on the forest floor, but available wood steadily increases with time since fire, as limbs and boles break and fall, as indicated in Fig. 3. This accumulation of wood is an important aspect of postfire recovery for wildlife.

Postfire timber harvest initially increases the amount of small-diameter wood (slash), but large woody debris declines over decades as fewer snags are available to replace wood lost to decomposition (Fig. 3). Lack of large woody debris may alter species occurrence or abundance of some wildlife species in regenerating forests following postfire timber harvest, depending on stand conditions and number of snags removed. However, few data exist on the relationship of down wood after fire to animal populations, making it difficult to determine if there are thresholds of down wood abundance beyond which more wood adds little to wildlife diversity or abundance.

Forest-dwelling vertebrates have evolved with and respond to changes in amounts, duration, and decomposition state of large woody debris in forests regenerating after natural disturbances such as fire. In ponderosa pine forests of central Oregon, golden-mantled ground squirrel (*Spermophilus lateralis*) density and survival were lower in stands with low levels of down wood (16 m^3 ha^{-1}) compared with high levels of down wood (118 m^3 ha^{-1}) (Smith and Maguire 2004). This pattern was not observed for yellow-pine chipmunk (*Tamias amoenus*) or deer mice (*Peromyscus maniculatus*). A synthesis for 12 forested biogeoclimatic zones in British Columbia (Canada) suggests that use of wood by vertebrates tends to increase with increasing precipitation and fire return interval (Bunnell et al. 2002).

Three time scales facilitate planning for postfire management of snags and down wood: pre-fire conditions, a short-term (<5 yr) postfire period, and a long- term (>5 yr) postfire period (Lehmkuhl et al. 2003). The prefire conditions of the stand influence the immediate postfire conditions for wildlife. Good postfire snag habitat occurs where (1) management actions promote tree species and large size classes favored for cavity excavation (Hutto 1995, 2006; Kotliar et al. 2002; Lehmkuhl et al. 2003), (2) patchiness in stand densities provides postfire habitats for different species (Hutto 1995; Haggard and Gaines 2001; Kotliar et al. 2002; Saab et al. 2002), and (3) defective trees that provide immediate postfire cavity excavation opportunities are created and retained (Hutto 1995; Saab and Dudley 1998; Lehmkuhl et al. 2003). Attaining desired size, density and distributions of snags and down wood (McIver and Ottmar 2006) is an important consideration for short-term postfire management (Thomas et al.1979; Bull et al. 1997). Managing for longevity of snag habitats helps to minimize the "snag gap" from the time when snag abundance falls below a habitat-use threshold to the creation of new snags in the burned landscape with regenerating trees (Saab and Dudley 1998; Everett et al. 1999; McIver and Ottmar 2006).

It is critical to interpret the effects of postfire timber harvest on animal species in the context of spatial scale. Deleterious stand-scale effects may indeed occur, but effects on animal populations and their viability in the larger landscape may be minimal. For example, would removing a large number of snags from 1,000 ha of a 25,000 ha stand-replacement fire, rendering that 1,000 ha unsuitable for snag-dependent species, represent a significant impact on the population viability of affected species? What would be the effect of harvesting 1,000 ha in a dispersed fashion, say in 50-ha blocks, as opposed to a single block? A 4% loss of habitat would likely have minimal impact on population viability across the burned area or the larger landscape context of the burn, unless the affected area had special habitat value. However, the habitat threshold for a particular species at large spatial scales cannot be easily quantified without modeling, nor quantified as a single number without specifying conditions for population viability and risk of local extirpation. Assuming that species are well adapted to persist across landscapes within that range, a parsimonious guideline for estimating "how much is enough" is to manage for a specified tolerance level of snag or down wood availability (Mellen et al. 2002) within a known range of variability (Landres et al. 1999; Hessburg et al. 1999; Agee 2003b).

Early efforts typified by Thomas et al. (1979) described snag dynamics, wildlife use of snags, diameter classes, and decay classes, and provided formulas for estimating snag density levels by Potential Population Values of cavity-using animals. Although this information on types of snags used by wildlife remains useful, the formulas for estimating the density of retained snags were underestimates (Rose et al. 2001; Hutto 2006).

These issues are addressed in the DecAID decision support system for snags and down wood (http://wwwnotes.fs.fed.us:81/pnw/DecAID/ DecAID.nsf) (Mellen et al. 2002). DecAID is an Internet-based summary, synthesis, and integration of the best available science on snag and down wood ecology and wildlife use in Washington and Oregon. It is a meta-analysis of the literature and research databases on wildlife use and forest inventory, rather than a model or a population viability assessment. DecAID uses statistical tolerance levels to advise on snag and down wood size and density requirements and their availability for vertebrates in green and burned stands (Fig. 15). Wildlife-use data specific to forest type and structural class, and density or cover data for snags and down wood are available in the Current Vegetation Survey and Natural Resource Inventory databases of the USDA Forest Service (USFS) and USDI Bureau of Land Management, respectively, and from the USFS Forest Inventory and Analysis database for private lands (Fig. 16). Examples instruct users how the database can be used for tree harvest projects to estimate size, density, and other attributes of snags and down wood, and their allocation across the landscape. The program does not give a single answer, but provides scientific information to meet management objectives, and links to the literature on which the decision support system is based.

Effects of Postfire Harvest on Riparian Systems and Aquatic Ecology

In many respects, riparian systems integrate the effects of postfire timber harvest on vegetation, fuels, soils, water, and wildlife discussed above. Few data exist on the effects of postfire timber harvest on aquatic ecosystems, so effects can be inferred from the literature on riparian fire regimes (Dwire and Kauffman 2003; Everett et al. 2003), fire effects (e.g., Gresswell 1999), and timber harvest effects individually. A recent summary of the effects of fire and timber harvest on riparian and aquatic systems (Reeves et al. 2006) concludes that fire in riparian areas (including smaller streams without fish) creates conditions that may not require intervention to sustain long-term productivity, and that protection of residual structures is critical.

Information on the effects of fire on aquatic vertebrates is sparse and has primarily focused on salmonid fishes (Gresswell 1999; Dunham et al. 2003; Rieman et al. 2003) and to a lesser extent amphibians (Pilliod et al. 2003). Short-term (<3 yr) effects on aquatic vertebrates is generally negative. Severe fires that burn through riparian areas can cause high mortality or emigration of fish and other organisms caused by heating and changes in water chemistry (Minshall et al. 1997; Rieman and Clayton 1997). Other potential changes include loss of in-channel wood, loss of vegetation, reduced infiltration capacity of soils, increased erosion, changes in timing and amount of runoff (discussed above), elevated stream temperatures, and altered channel morphology (Wondzell and King 2003).

Long-term (>10 yr) effects on aquatic systems and biota depend on the context and scale of fire occurrence (Reeves et al. 2006). Erosion following fires can provide wood and coarse sediment that maintain productive habitats (Reeves et al. 1995) and can create heterogeneity in channel structure and complexity, and nutrient transfer can temporarily increase aquatic productivity (Minshall 2003). Benthic macroinvertebrates populations recover within a few years following fire (Minshall 2003), and fish communities in Idaho were observed to recover within 10 years of severe fires (Rieman and Clayton 1997).

Substantial information exists on the effects of timber harvest on fish and amphibians (Murphy et al. 1981; Murphy and Hall 1981; Hawkins et al. 1982, 1983; Corn and Bury 1989; Murphy and Koski 1989; deMaynadier and Hunter 1995), and most of this literature documents adverse responses, especially in the first few years following timber harvest. Extrapolating this knowledge to postfire timber harvest, removal of snags from postfire landscapes may influence macroinvertebrates (Minshall 2003), fish, and amphibian populations, because large wood recruitment into streams following fire affects channel morphology (e.g., steps, pool habitats), sediment transport, and nutrient cycling (Gregory and Bisson 1996; May and Gresswell 2003).

Postfire timber harvest activities such as road building and log skidding can increase surface soil erosion, resulting in increased sedimentation of stream substrates (Ketcheson and Megahan 1996; Lee et al. 1997; Rieman and Clayton 1997; Church and Eaton 2001). High sediment loads can bury fish and amphibian eggs and eliminate protective interstitial cavities used by juvenile fish, larval amphibians, and benthic invertebrates (Beaty 1994; Minshall et al. 1997; Church and Eaton 2001; Gillespie 2002). Mass wasting and debris flows, often associated with structural failure of roads or removal of trees near drainage headwalls and shoulder slopes, are the most damaging to stream habitat. If postfire management activities depart from aquatic system recovery processes in terms of (1) physical and biological landscape elements (e.g., wood in streams) and (2) frequency and severity of natural disturbance, those activities can delay a return to conditions comparable to those prior to disturbance (Beschta et al. 2004). Conversely, management activities that complement ecosystem recovery processes may help minimize long-term damage to aquatic systems (Reeves et al. 2006).

Adverse effects of postfire timber harvest and roads on riparian and aquatic systems may be reduced by maintaining forested buffers in the riparian zone. Riparian buffers reduce sediment delivery to streams, maintain cooler stream temperatures through shading, and minimize changes in aquatic habitat for fish and macroinvertebrates (Davies and Nelson 1994). Vegetation in riparian buffers mitigates the effects of timber harvest on wildlife by providing habitat for small mammals (Cockle and Richardson 2003) and birds (Pearson and Manuwal 2001). The appropriate size of buffers varies depending on landscape characteristics and management objectives.

Conclusions

The scientific literature on postfire timber harvest, postfire effects, and post-harvest effects indicates that a diversity of possible outcomes – ranging from negligible to highly significant – can be expected in response to timber harvest following large, severe wildfires. One of the challenges for incorporating this literature in planning for postfire management is that most existing data have been collected at small spatial scales (<10 ha) (McIver and Starr (2001), while most of the applications are at large spatial scales (>1000 ha). Aggregation of fine-scale components is necessary for both computational efficiency and to minimize error propagation (McKenzie et al. 1996). This aggregation can occur through modeling and expert judgment in order to make well-informed decisions about postfire management.

Several principles about postfire timber harvest emerge from our review:

- Timing of timber harvest following fire (same season as fire vs. subsequent years, winter vs. other seasons) can affect the magnitude of ecological and economic effects. Specifically, (1) harvest can cause mortality of naturally regenerating trees if harvest occurs *after* trees have established, (2) harvest operations conducted over snow typically minimize soil compaction and erosion, and (3) harvest soon after fire provides the highest value of wood for commercial use.
- Potential for high fire hazard following fire can exist, depending on local vegetation and climate, if no management action is taken to remove fine surface fuels and ladder fuels. Harvest can exacerbate short-term fire hazard unless it is accompanied by fuel treatments that remove fine surface fuels created during the harvest operation.
- Potential for insect attack following fire can exist, depending on local tree species and climate, and this potential can in some cases be reduced if damaged, low vigor trees are removed. The extent of insect outbreaks following fire varies greatly depending on a variety of local site factors.
- Fire and timber harvest, individually and combined, generally have transient effects on physical, biological, and nutritional properties of soil. Both fire and harvest remove wood and C that would be potentially incorporated in the soil. Fire can create temporary hydrophobicity near the soil surface and result in a short-term pulse of nutrients. Harvest with ground-based equipment and cable yarding can compact the soil and cause short-term erosion.
- Fire and timber harvest, individually and combined, reduce water uptake by vegetation, causing an immediate response in hydrological processes. Stream flow and water supply can increase sharply, then decrease as vegetation regrows. Water

quality in streams can also decrease if increased stream flow is accompanied by erosion, with large debris movements being especially damaging. Water quality improves over time as vegetation regrows.

- Many of the short-term effects of removing trees on wildlife habitat are negative but can also be positive for some taxa. Cavity-nesting birds and other cavity-nesting vertebrates may be affected by harvesting standing dead and live trees, with variable impacts depending on the habitat requirements of each species; some species are sensitive to snag density, and others are not. Effects of tree removal on wildlife habitat typically decrease gradually over time, and are less negative (or positive) when evaluated across large spatial scales.
- Short-term effects of removing trees on aquatic systems are mostly negative, and timber harvest and transportation systems that disturb the soil surface or accelerate road-related erosion may be particularly harmful. However, if riparian buffers are used to maintain areas of undisturbed vegetation and soil, deleterious effects can be reduced. Effects on aquatic systems typically decrease over time as vegetation regrows.

Scientific Principles, Uncertainty, and Risk

Postfire management of forest ecosystems, which may include harvesting and planting trees (Sessions et al. 2004), can be incorporated into the broader task of sustainable resource management, rather than being addressed as an "emergency" or aberration in long-term planning (Agee 2002b). Forest ecosystems in western North America will continue to experience large fires, and larger expanses of forests will likely burn in a greenhouse climate

(McKenzie et al. 2004) despite fuel treatments and other mitigation efforts. Therefore, management of postfire environments may be facilitated if planning documents that articulate postfire management options are in place in anticipation of a large fire, thus allowing for timely implementation of appropriate actions. Post-wildfire treatments can be be evaluated spatially and temporally, with associated risks and uncertainty considered, including the effects of alternative silvicultural systems within and among stands and watersheds over time. In addition, increasing evidence suggests that fuel treatments in dry forests can reduce severity of wildfire effects except under very severe weather conditions (e.g., Pollet and Omi 2002; Peterson et al. 2005). Comparing risks associated with no action and with alternative management actions can help clarify postfire decision making.

Effective management of natural resources may be enhanced by using an interdisciplinary approach to address alternative actions prior to and following wildlfires. It is especially important that fire management and silviculture are integrated with respect to scientific concepts and on-the-ground applications (Graham et al. 1999; Peterson et al. 2005), because management of forest structural characteristics affects long-term patterns of fuels and fire hazard. Similarly, because management of forest structure affects wildlife habitat and biological diversity, coordination with wildlife biologists and vegetation biologists is helpful. Fortunately there are signs that this integration is starting to occur in both science (Reinhardt and Crookston 2003) and management (Johnson et al. 2006).

The complexities of forest ecosystems, variability in postfire vegetation response, and a changing climate provide a challenging situation for resource managers weighing alternative management actions. Even with good scientific information, uncertainty often remains about the outcome of management options. Managers are faced with identifying the "best available science," which in some cases contains conflicting information or is based on different analytical methods. Nevertheless, credible scientific information can be identified with a structured process that includes comparing alternatives, documenting how alternatives are selected and scientifically supported, consulting outside reviewers, and consulting potential stakeholders (Peterson et al. 2006).

At present, no single decision support system exists that can be used to select alternatives for postfire management. Most existing tools are designed for a single resource –vegetation, fuels, soils, water, or wildlife – without much integration. Therefore, integration is the responsibility of resource management staff for burned areas, perhaps in cooperation with local scientists. It remains to be seen whether a useful decision support system for postfire management that includes all resources will be developed. Until then, adaptive management will facilitate the evaluation of long-term effects of different alternatives.

Harvesting timber following fire has usually been an economic undertaking and rarely a restorative activity in the sense of ecological restoration (Society for Ecological Restoration 2004). Logs and slash can be used to mitigate for potential erosion, and removal of standing dead and damaged trees can reduce fuel loads and mitigate for potential insect hazard. However, these activities may or may not facilitate a trajectory towards desired resource conditions in the long term. Postfire harvest would be more effective in a restoration sense if management pathways for attaining desired combinations of species, forest structure, and ecological functions are specified, so that vegetation management is designed to be compatible and minimally conflicting with those pathways. Clearly stated management objectives – which may include physical, biological, and economic components – are an important guide for timber harvest or any other activity following a large wildfire.

A Long-Term Management Experiment

Lack of data from consistent, replicated studies on the effects of postfire timber harvest, especially data that cut across different resource disciplines, is a critical scientific gap. Retrospective research on the effects of postfire timber harvest across a number of sites could be helpful as a first step. However, long-term research and monitoring would inform adaptive management of burned landscapes and reduce uncertainty in decision making. Each large wildfire presents an opportunity to implement local studies and to expand the knowledge base on postfire management actions.

We propose long-term, consistent, well replicated management experiments at multiple locations to reduce scientific uncertainty in decision making about postfire timber harvest. A network of sites would ensure: (1) consistent application of scientific principles, (2) robust statistical design and analysis, (3) central management of data to ensure data quality and security, and (4) rapid dissemination of results. Scientific teams working closely with resource managers would ensure effective research-management collaboration.

Potential treatments would include: time since fire (e.g., <1 yr, 3 yr), post-harvest stand density (high [no harvest], moderate, low), and fuel treatment (none vs. effective removal of fine fuels). Factors beyond those measured in these treatments would probably be considered as covariates in order to keep the size of the experiments reasonable. Important issues addressed by response variables would include: vegetation composition and productivity; silvicultural treatments and outcomes; spatial and temporal patterns of fuels and fire hazard; effects on local hydrology; effects on soils, biogeochemical cycling, and carbon; and habitat for birds and small mammals.

Such experiments would ideally continue for at least 15 years to adequately capture temporal trends in postfire and post-harvest responses. Sampling intensity and frequency would initially be high, but could decrease over time depending on trends in the data. Data would be analyzed annually to examine trends and provide input to adaptive management of landscapes subject to large fires. It is anticipated that results obtained after the first few years would be highly informative for decision making and policy, as greater certainty is attained about responses to postfire treatments.

Acknowledgements

We thank Edward DePuit, Daniel Donato, Patricia Loesche, James McIver, David W. Peterson, David Powell, Kevin Ryan, Carl Skinner, John Townsley, and Cynthia West for helpful comments on an earlier draft of the manuscript. Kelly O'Brian assisted with graphics. Ellen Eberhardt assisted with editing.

References

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

Agee, J.K. 2002a. Fire as a coarse filter for snags and logs. *In* Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests. *Tech. coord. by* W.F. Laudenslayer, P.J. Shea, B.E. Valentine, C.P. Weatherspoon, and T.E. Lisle. USDA For. Serv. Gen. Tech. Rep. PSW-GTR-181. pp. 359-368.

Agee, J.K. 2002b. The fallacy of passive management. Conserv. Biol. Prac. **3**: 18-25

Agee, J.K. 2003*a*. Burning issues in fire: will we let the coarse filter operate? *In* Fire Conference 2000: First National Congress on Fire Ecology, Prevention, and

Management. *Edited by* K.E.M. Galley, R.C. Klinger, and N.G. Sugihara. Misc. Pub. 13. Tall Timbers Research Station. Tallahassee, Fla. pp. 7-13.

Agee, J.K. 2003*b*. Historical range of variability in eastern Cascades forests, Washington, USA. Landsc. Ecol. **18**: 725-740.

Agee, J.K., and Huff, M.H. 1987. Fuel succession in a western hemlock/Douglas-fir forest. Can. J. For. Res. **17**: 697-704.

Agee, J.K., Wright, C.B., Williamson, N., and Huff, M.H. 2002. Foliar moisture content of Pacific Northwest vegetation and its relation to wildland fire behavior. For. Ecol. Manage. **167**: 57-62.

Alexander, R.R., Shearer, R.C., and Shepperd, W.D. 1990. Silvics of subalpine fir. *In* Silvics of North America, Volume 1, Conifers. *Tech. coord. by* R.M. Burns, and B.H. Honkala. USDA For. Serv. Agric. Handb. 654. pp. 60-70.

Almendos, G., Gonzalez-Vila, F.J., and Martin, F. 1990. Fire-induced transformation of soil organic matter from an oak forest – an experimental approach to the effects of fire on humic substances. Soil Sci. **149**: 158-168.

Amman, G.D., and Ryan, K.C. 1991. Insect infestation of fire-injured trees in the Greater Yellowstone area. USDA For. Serv. Res. Note INT-398.

Anderson, R.C., and Dooley, J. 2006. Healing in a fire's aftermath – engineered WoodStraw™: effective new erosion-control product. Resource **13**: 9-10.

Andrews, P.L. 1986. BEHAVE: fire behavior prediction and fuel modeling system-BURN subsystem, part I. USDA For. Serv. Gen. Tech. Rep. INT-194.

Beaty, K.G. 1994. Sediment transport in a small stream following two successive forest fires. Can. J. Fish. Aquat. Sci. **51**: 2723-2733.

Beschta, R.L., Rhodes, J.J., Kauffman, J.B., Gresswell, R.E., Minshall, G.W., Karr, J.R., Perry, D.A., Hauer, F.R., and Frissell, C.A. 2004. Postfire management on forested public lands of the western United States. Conserv. Biol. **18**: 957-967.

Blake, J.G. 1982. Influence of fire and logging on nonbreeding bird communities of ponderosa pine forests. J. Wildl. Manage. **46**: 404-415.

Bond, M.L., Gutierrez, R.J., Franklin, A.B., LaHaye, W.S., May, C.A., and Seamans, M.E. 2002. Short-term effects of wildfires on spotted owl survival, site fidelity, mate fidelity, and reproductive success. Wildl. Soc. Bull. **30**: 1022-1028.

Bosch, J.M., and Hewlett, J.D. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J. Hydrol. **55**: 3-23.

Brooks, K.N., Ffolliott, P.F., Gregersen, H.M., and DeBano, L.F. 2003. Hydrology and the management of watersheds. 3d ed. Iowa State Univ. Press, Ames, Ia.

Brown, J.K., Reinhardt, E.D., and Kramer, K.A. 2003. Coarse woody debris: managing benefits and fire hazard in the recovering forest. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-105.

Bull, E.L. 2002. The value of coarse woody debris to vertebrates in the Pacific Northwest. *In* Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests. *Tech. coord. by* W.F. Laudenslayer, P.J. Shea, B.E. Valentine, C.P. Weatherspoon, and T.E. Lisle. USDA For. Serv. Gen. Tech. Rep. PSW-GTR-181. pp. 171-178.

Bull, E.L., Parks, C.G., and Torgersen, T. 1997. Trees and logs important to wildlife in the interior Columbia River basin. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-391.

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

Bunnell, F.L., Houde, I., Johnston, B., and Wind, W. 2002. How dead trees sustain live organisms in western forests. *In* Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests. *Tech. coord. by* W.F. Laudenslayer, P.J. Shea, B.E. Valentine, C.P. Weatherspoon, and T.E. Lisle. USDA For. Serv. Gen. Tech. Rep. PSW-GTR-181. pp. 291-318.

Busse, M.D., Hubbert, K.R., Fiddler, G.O., Shestak, C.J., and Powers, R.F. 2005. Lethal soil temperatures during burning of masticated forest residues. Int. J. Wildland Fire **14**: 267-276.

Caton, E.L. 1996. Effects of fire and salvage logging on the cavity-nesting bird community in northwestern Montana. Ph.D. dissertation, University of Montana, Missoula.

Chen, J., Franklin, J.F., and Spies, T.A. 1995. Growing season microclimatic gradients from clearcut edges into old-growth Douglas-fir forests from clearcut edges. Ecol. Applic. **5**: 74-86.

Chou, Y.H., Conard, S.G., and Wohlgemuth, P.M. 1994a. Analysis of postfire salvage logging, watershed characteristics, and sedimentation in the Stanislaus National Forest. *In* Proceedings ESRI Users Conference. Environmental Systems Research Institute, Redlands, Calif. pp. 492-499.

Chou, Y.H., Conard, S.G., and Wohlgemuth, P.M. 1994b. Postfire salvage logging variables and basin characteristics related to sedimentation, Stanislaus National Forest, California. *In* Proceedings GIS ’94 Symposium. Reid, Collins, and Associates, Vancouver, B.C.

Church, M., and Eaton, B. 2001. Hydrological effects of forest harvest in the Pacific Northwest. Riparian Decision Tool Tech. Rep. 3. Dept. of Geography, University of British Columbia, Vancouver.

Cockle, K.L., and Richardson, J.S. 2003. Do riparian buffer strips mitigate the impacts of clearcutting on small mammals? Biol. Conserv. **113**:133-140.

Cole, D.W., and Gessel, S.P. 1992. Fundamentals of plant nutrition. *In* Forest fertilization: sustaining and improving nutrition and growth of western forests. *Edited by* H.N. Chappell, G.F. Weetman, and R.E. Miller. Institute of Forest Resources Contribution 73, College of Forest Resources, University of Washington, Seattle. pp. 7-16.

Corn, P.S., and Bury, R.B. 1989. Logging in western Oregon: responses of headwater habitats and stream amphibians. For. Ecol. Manage. **29**: 39-57.

Davies, P.E., and Nelson, M. 1994. Relationships between riparian buffer weidths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. Aust. J. Mar. Freshw. Res. 45: 1289-1305.

DeBano, L.F. 1979. Effects of fire on soil properties. *In* California Forest Soils: a Guide for Professional Foresters and Resource Managers and Planners. *Edited by* R.J. Laacke. Agricultural Sciences Publications, University of California, Berkeley. pp. 109-118.

DeBano, L.F. 1981. Water repellent soils: a state-of-the-art review. USDA For. Serv. Gen. Tech. Rep. PSW-46.

Debano, L.F. 2000. Fire-induced water repellency: an erosional factor in wildland environments. *In* Land Stewardship in the 21st Century: the Contributions of Watershed Management. *Edited by* P.F. Ffolliott, M.B. Baker, C.B. Edminster, M.C. Dillon, K.L. Mora. USDA For. Serv. Proc. RMRS-P-13. pp. 307-310.

DeBano, L.F., Neary, D.G., and Ffolliott, P.F. 1998. Fire effects on ecosystems. John Wiley & Sons, New York.

DeLuca, T.H., and Zouhar, K.L. 2000. Effects of selection harvest and prescribed fire on the soil nitrogen status of ponderosa pine forests. For. Ecol. Manage. **138**: 263-271.

Doerr, S.H., Blake, W.H., Shakesby, R.A., Stagnitti, F., Vuurens, S.H., Humphreys, G.S., and Wallbrink, P. 2004. Heating effects on water repellency in Australian eucalypt forest soils and their value in estimating wildfire soil temperatures. Int. J. Wildl. Fire **13**: 157-163.

deMaynadier, P.G., and Hunter, M.L. 1995. The relationship between forest management and amphibian ecology: a review of the North American literature. Environ. Rev. **3**: 230-261.

Donato, D.C., Fontaine, J.B., Campbell, J.L., Robinson, W.D., Kauffman, J.B., and Law, B.E. 2006. Post-wildfire logging hinders regeneration and increases fire risk. Science **311**: 352.

Dunham, J.B., Young, M., Gresswell, R., and Rieman, B.E. 2003. Effects of fire on fish populations: landscape perspectives on persistence of native fishes and non-native fish invasions. For. Ecol. Manage. **178**: 183-196.

Dwire, K.A., and Kauffman, J.B. 2003. Fire and riparian ecosystems in landscapes of the western USA. For. Ecol. Manage. **178**: 61-74.

Dykstra, D.P. 1976. Cable, balloon, and helicopter yarding compared for partial cutting and clearcutting in old-growth Douglas-fir. USDA For. Serv. Res. Bull. PNW-RP-22.

Dykstra, D.P., and Heinrich, R. 1996. FAO model code of forest harvesting practice. Food and Agriculture Organization of the United Nations, Rome, Italy.

Dyrness, C.T., Van Cleve, K., and Levison, J.D. 1989. The effect of wildfire on soil chemistry in four forest types of interior Alaska. Can. J. For. Res. **19**: 1389-1396.

Edmonds, R.L., Agee, J.K., and Gara, R.I. 2005. Forest health and protection. Waveland Press, Long Grove, Ill.

Everett, R., Lehmkuhl, J., Schellhaas, R., Ohlson, P., Keenum, D., Reisterer, H., and Spurbeck, D. 1999. Snag dynamics in a chronosequence of 26 wildfires on the east slope of the Cascade Range in Washington state, USA. Int. J. Wild. Fire **9**: 223–234.

Everett, R.L., Schellhaas, R., Keenum, D., Spurbeck, D., and Ohlson, P. 2000. Fire history in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. For. Ecol. Manage. **129**: 207-225.

Everett, R.L., Schellhaas, R., Ohlson, P., Spurbeck, D., Keenum, D. 2003. Continuity in fire disturbance between riparian and adjacent sideslope Douglas-fir forests. For. Ecol. Manage. **175**: 31-47.

Farris, K.L., Huss, M.J., and Zack, S. 2004. The role of foraging woodpeckers in the decomposition of ponderosa pine snags. The Condor **106**: 50-59.

Food and Agricultural Organization (FAO). 2005. Global forest resources assessment 2005. Progress towards sustainable forest management. FAO Forestry Paper 147, Food and Agricultural Organization of the United Nations, Rome, Italy.

Forman, R.T.T., and Alexander, L.E. 1998. Roads and their major ecological effects. Ann. Rev. Ecol. Syst. **29**: 207-231.

Franklin, A.B., Anderson, D.R., Gutierrez, R.J., and Burnham, K.P. 2000. Climate, habitat quality, and fitness in northern spotted owl populations in northwestern California. Ecol. Monogr. **70**: 539-590.

Franklin, J.F., Spies, T., Van Pelt, R., Carey, A., Thornburgh, D., Berg, D.R., Lindenmayer, D., Harmon, M., Keeton, W.S., Shaw, D.C., Bible, K., and Chen, J. 2002. Disturbances and the structural development of natural forest ecosystems with some implications for silviculture. For. Ecol. Manage. **155**: 399-423.

Franzreb, K.E., and Ohmart, R.D. 1978. The effects of timber harvesting on breeding birds in a mixed-coniferous forest. The Condor **80**: 431-441.

Frelich, L.E., and Reich, P.B. 1995. Spatial patterns and succession in a Minnesota southern boreal forest. Ecol. Monogr. **65**: 325-346.

Froehlich, H.A., and McNabb, D.H. 1984. Minimizing soil compaction in Pacific Northwest sites. *In* Forest Soils and Treatment Impacts. *Edited by* E.L. Stone. Department of Forestry, Wildlife and Fisheries, University of Tennessee, Knoxville. pp. 159-192.

Furniss, M.J., Ledwith, T.S., Love, M.A., McFadin, B.C., and Flanagan, S.A. 1998. Response of road-stream crossings to large flood events in Washington, Oregon, and northern California. USDA For. Serv. Pub. 9877. San Dimas Technology and Development Center, San Dimas, Calif.

Furniss, R.L., and Carolin, V.M. 1977. Western forest insects. USDA Forest Service Misc. Pub. 1339.

Gaines, W.L., Strand, R., and Piper, S. 1997. Effects of the Hatchery Complex fires on the northern spotted owls in the eastern Washington Cascades. *In* Proc. Fire Effects on Rare and Endangered Species and Habitats Conf, 13-16 November 1995, Coeur d’Alene, Id. International Association of Wildland Fire, Fairfield, Wash. pp. 123–130.

Garrett, K.L, Raphael, M.G., and Dixon, R.D. 1996. White-headed woodpecker (*Picoides albolarvatus*). *In* The Birds of North America, No. 252. *Edited by* A. Poole, and F. Gill. Academy of Natural Sciences, Philadelphia, Penn.

Gillespie, G.R. 2002. Impacts of sediment loads, tadpole density, and food type on the growth and development of tadpoles of the spotted tree from *Litoria spenceri*: an in-stream sediment. Biol. Conserv. **106**: 141-150.

Gomez, G.A., Powers, R.F., Singer, M.J., and Horwath, W.R. 2002. Soil compaction effects on growth of young ponderosa pine following litter removal in California's Sierra Nevada. Soil Sci. Soc. Am. J. **66**: 1334-1343.

Graham, R. T. 1990. Silvics of western white pine. *In* Silvics of North America, Volume 1, Conifers. *Tech. coord. by* R.M. Burns and B.H. Honkala. USDA For. Serv. Agric. Handb. 654. pp. 385-394.

Graham, R.T., Harvey, A.E., Jain, T.B., and Tonn, J.R. 1999. The effects of thinning and similar stand treatments on fire behavior in western forests. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-463.

Graham, R.T., McCaffrey, S., and Jain, T.B. 2004. Science basis for changing forest structure to modify wildfire behavior and severity. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-120.

Gray, A.N., and Franklin, J.F. 1997. Effects of multiple fires on the structure of southwestern Washington forests. Northwest Sci. **71**: 174-185.

Greenberg, C.H., Harris, L.D., and Neary, D.G. 1995. A comparison of bird communities in burned and salvage-logged, clearcut, and forested Florida sand pine scrub. Wilson Bull. **197**: 40-54.

Greenlee, D., and Greenlee, J. 2002. Changes in fire hazard as a result of the Cerro Grande Fire. Fire Manage. Today **62**:15-21.

Gregory, S.V., and Bisson, P.A. 1996. Degradation and loss of anadromous salmonid habitat in the Pacific Northwest. *Edited by* D. Stouder, and R.J. Naiman. *In* Pacific Salmon and Their Ecosystems: Status and Future Options. Springer-Verlag, New York, N.Y. pp. 277-314.

Gresswell, R.E. 1999. Fire and aquatic ecosystems in forested biomes of North America. Trans. Amer. Fish. Soc. **128**: 193-221.

Grifantini, M.C. 1990. Early-seral changes following wildfire, salvage-logging and reforestation, Klamath Mountains, CA. Master's thesis, Humboldt State University, Arcata, Calif.

Grifantini, M.C., Stuart, J.D., and Fox, L. 1992. Deer habitat changes following wildfire, salvage logging and reforestation, Klamath Mountains, California. *In* Proceedings Symposium on Biodiversity of Northwestern California, Report 29. University of California Wildland Resources Center, Berkeley, CA. pp. 163-167.

Gucinski, H., Furniss, M. J., Ziemer, R.R., and Brookes, M.H. 2001. Forest roads: a synthesis of scientific information. USDA For. Serv.Gen. Tech. Rep. PNW-GTR-509.

Haase, S.M., and Sackett, S.S. 1998. Efects of prescribed fire in giant sequoia-mixed conifer stands in Sequoia and Kings Canyon National Parks. *Edited by* T.L. Pruden, and L.A. Brennan. Proc. Tall Timbers Fire Ecol. Conf. **20**: 236-243.

Hadfield, J., and Magelssen, R. 2006. Wood changes in fire-killed tree species in eastern Washington. USDA Forest Service Report, Okanogan and Wenatchee National Forests, Wenatchee, Wash.

Haggard, M., and Gaines, W.L. 2001. Effects of stand-replacement fire and salvage logging on a cavity-nesting bird community in eastern Cascades, Washington. Northwest Sci. **75**: 387-396.

Hanson, J.J., and Stuart, J.D. 2005. Vegetation responses to natural and salvage logged fire edges in Douglas-fir/hardwood forests. For. Ecol. Manage. **214**: 266-278.

Harmon, M.E., Franklin, J.F., Swanson, F.J., Sollins, P., Gregory, S.V., Lattin, J.D., Anderson, N.H., Cline, S.P., Aumen, N.G., Sedell, J.R., Lienkaemper, G.W., Cromack, K., and Cummins, K.W. 1986. Ecology of coarse woody debris in temperate ecosystems. Adv. Ecol. Res. **15**: 133-302.

Harvey, A.E., Graham, R.T., and McDonald, G.I. 2000. Fire/decay: managing codependent forest processes across the landscape. *In* Crossing the Millennium: Integrating Spatial Technologies and Ecological Principles for a New Age in Fire Management, Proceedings from the Joint Fire Science Conference and Workshop, 15-17 June 1999, Boise, Id. *Edited by* L.F. Neuenschwander and K.C. Ryan. International Association of Wildland Fire and University of Idaho, Boise. pp. 179-189.

Hawkins, C.P., Murphy, M.L., and Anderson, N.H. 1982. Effects of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in Cascade Range streams of Oregon. Ecology **63**: 1840-1856.

Hawkins, C. P., Murphy, M.L., Anderson, N.H., and Wilzbach, M.A. 1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. Can. J. Fish. Aquat. Sci. **40**: 1173-1185.

Hejl, S. and McFadzen, M. 1998. Maintaining fire-associated bird species across forest landscapes in the northern Rockies. Summary report on file at USDA Forest Service, Rocky Mountain Research Station, Missoula, Mont.

Helvey, J.D. 1980. Effects of a north central Washington wildfire on runoff and sediment production. Water Resour. Bull 16: 627-634.

Helvey, J.D., Tiedemann, A.R., and Anderson, T.D. 1985. Plant nutrient losses byh soil erosion and mass movement after wildfire. J. Soil Water Conserv. **40**:168-173.

Hessburg, P.F., Smith, B.G., and Salter, R.B. 1999. Detecting change in forest spatial patterns from reference conditions. Ecol. Applic. **9**: 1232–1252.

Hitchcox, S.M. 1996. Abundance and nesting success of cavity-nesting birds in unlogged and salvage-logged burned forest in northwestern Montana. Master's thesis. University of Montana, Missoula.

Hobson, K.A., and Schieck, J. 1999. Changes in bird communicates in boreal mixedwood forest: harvest and wildfire effects over 30 years. Ecol. Applic. **9**: 849-863.

Horton, S.P., and Mannan, R.W. 1988. Effects of prescribed fire on snags and cavity-nesting birds in southeastern Arizona pine forests. Wildl. Soc. Bull. **16**:37-44.

Huff, M.H., and Smith, J.K. 2000. Fire effects on animal communities: effects of fire on fauna. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-42-vol. 1.

Hunter, M.L. 1999. Maintaining Biodiversity in Forest Ecosystems. Cambridge University Press, Cambridge, U.K.

Hutto, R.L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountain (USA) conifer forests. Conserv. Biol. **9**: 1041-1058.

Hutto, R.L. 2006. Toward meaningful snag-management guidelines for postfire salvage logging in North American conifer forests. Conserv. Biol. **20**: 984-995.

Imbeau, L., Savard, J-P.L., and Gagnon, R. 1999. Comparing bird assemblages in successional black spruce stands originating from fire and logging. Can. J. Zool. **77**: 1850-1860.

Isaac, L. 1940. Vegetation succession following logging in the Douglas-fir region with special reference to fire. J. For. **38**: 716-721.

Jain, T.B., Graham, R.T., and Morgan, P. 2004. Western white pine growth relative to forest openings. Can. J. For. Res. 34: 2187-2198.

Johnson, M.C., Peterson, D.L., and Raymond, C.L. 2006. Guide to fuel treatments in dry forests of the western United States: assessing forest structure and fire hazard. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-686.

Johnson, S.L., and Jones, J.A. 2000. Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. Can. J. Aquat. Sci. **57** (Suppl. 2): 30-39.

Johnson, T., and Wauer, R.H. 1996. Avifaunal response to the 1977 La Mesa fire. *In* Proceedings of the Second La Mesa Fire Symposium. *Edited by* C.D. Allen. USDA Forest Service Gen. Tech. Rep. RM-GTR-286. pp. 70-94.

Jones, J.A., Swanson, F.J., Wemple, B.C., and Snyder, K.U. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conserv. Biol. **14**: 76-85.

Kauffman, J.B. 1990. Ecological relationships of vegetation and fire in Pacific Northwest forests. *In* Natural and prescribed fire in Pacific Northwest forests. *Edited by* J. Walstad, S.R. Radosevich, and D.V. Sandberg. Oregon State University Press, Corvallis. Chapter 4.

Keeley, J.E. 2004. Ecological impacts of wheat seeding after a Sierra Nevada wildfire. Int. J. Wildl Fire **13**: 73-78.

Keller, G., and Sherar, J. 2003. Low-volume road engineering: best management practices: field guide. U.S. Agency for International Development, USDA Forest Service, and Conservation Management Institute, Virginia Polytechnic Institute and State University. Available from http://ntl.bts.gov/ruraltransport/card_view.cfm?docid=24650 [accessed 4 May 2006].

Ketcheson, G.L., and Megahan, W.F. 1996. Sediment production and downslope sediment transport from forest roads in granitic watersheds. USDA For. Serv. Res. Pap. INT-RP-486.

Khetmalas, M.B., Egger, K.N., Massicotte, H.B., Tackaberry, L.E., and Clapperton, M.J. 2002. Bacterial diversity associated with subalpine fir (*Abies lasiocarpa*) ectomycorrhizae following wildfire and salvage-logging in central British Columbia. Can. J. For Res. **48**: 611-625.

Kimmey, J.W. 1955. Rate of deterioration of fire-killed timber in California. USDA For. Serv. Circ. 962.

Kimmey, J.W., and Furniss, R.L. 1943. Deterioration of fire-killed Douglas-fir. USDA Tech. Bull. 851.

Klock, G.O. 1975. Impact of postfire salvage logging systems on soils and vegetation. J. Soil Water Conserv. **30**: 78–81.

Koenigs, E., Shea, P.J., Borys, R., and Haverty, M.I. 2002. An investigation of the insect fauna associated with coarse woody debris of *Pinus ponderosa* and *Abies concolor* in northeastern California. *In* Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests. *Tech. coord. by* W.F. Laudenslayer, P.J. Shea, B.E. Valentine, C.P. Weatherspoon, and T.E. Lisle. USDA For. Serv. Gen. Tech. Rep. PSW-GTR-181. pp. 97-110.

Kotliar, N.B., Hejl, S.J., Hutto, R.L., Saab, V.A., Melcher, C.P., and McFadzen, M.E. 2002. Effects of fire and postfire salvage logging on avian communities in conifer-dominated forests of the western United States. Stud. Avian Biol. **25:** 49–64.

Kreisel, I.J., and Stein, S.J. 1999. Bird use of burned and unburned coniferous forests during winter. Wilson Bull. 111: 243-250.

Landres, P.B., Morgan, P., and Swanson, F.J. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecol. Applic. **9**: 1179–1188.

Lee, D., Sedell, J., Rieman, B., Thurow, R., and Williams, J. 1997. Broad-scale assessment of aquatic species and habitats. *In* An Assessment of Ecosystem Components in the Interior Columbia River Basin and Portions of the Klamath and Great Basins. *Edited by* T.M. Quigley, and S.J. Arbelbide. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-405. Chapter 4.

Lehmkuhl, J.F. 2004. Epiphytic lichen diversity and biomass in low-elevation forests of the eastern Washington Cascade range, USA. For. Ecol. Manage. **187**: 381-392.

Lehmkuhl, J.F. 2005. Wildlife adaptations and management in Eastside interior forests with mixed-severity fire regimes. *In* Proc. Symp. mixed-severity fire regimes: ecology and management. Misc. Pub. 3. Association for Fire Ecology, Davis, Calif.

Lehmkuhl, J.F., Everett, R.L., Schellhaas, R., Ohlson, P., Keenum, D., Riesterer, H., and Spurbeck, D. 2003. Cavities in snags along a wildfire chronosequence in eastern Washington. J. Wildl. Manage. **67**: 219-228.

Lehmkuhl, J.F., Kistler, K., Begley, J., and Boulanger, J. 2006. Demography of northern flying squirrels informs ecosystem management of western interior forests. Ecol. Applic. **16**: 584-600.

Lindenmayer, D.B., and Noss, R.E. 2006. Salvage logging, ecosystem processes, and biodiversity conservation. Conserv. Biol. **20**:949-958.

Lotan, J.E., and Critchfield, W.B. 1990. Silvics of lodgepole pine. *In* Silvics of North America, volume 1, conifers. *Tech. coord. by* R.M. Burns, and B.H. Honkala. U.SDA For. Serv. Agric. Handb. 654. pp. 302-315.

Lowell, E.C., and Cahill, J.M. 1996. Deterioration of fire-killed timber in southern Oregon and northern California. West. J. Appl. For. **11**: 125-131.

Lowell, E.C., Willits, S.A., and Krahmer, R.L. 1992. Deterioration of fire-killed and fire-damaged timber in the western United States. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-292.

Lyon, L., and Smith, J. 2000. Management and research implications. *In* Wildland fire in ecosystems: effects on fauna. *Edited by* J.K. Smith. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-42-vol 1. Chapter 8.

Marston, R.A., and Hare, D.H. 1990. Runoff and soil loss following the 1988 Yellowstone fires. Great Plains-Rocky Mtn. Geograph. J. **18**: 1-8.

Maser, C., Anderson, R.G., Cromack, K., Williams, J., and Martin, R.E. 1979. Dead and down woody material. *In* Wildlife habitats in managed forests – the Blue Mountains of

Oregon and Washington. *Edited by* J.W. Thomas. USDA For. Serv. Agric. Handbook 553. pp. 78-90.

May, C.L., and Gresswell, R.E. 2003. Processes and rates of sediment and wood accumulation in headwater streams of the Oregon Coast Range, USA. Earth Surf. Proc. Land. **28**: 409-424.

McClelland, D.E., Foltz, R.B., Falter, C.M., Wilson, W.D., Cundy, T., Schuster, R.L., Saurbier, J., Rabe, C., and Heinemann, R. 1999. Relative effects on a low-volume road system of landslides resulting from episodic storms in northern Idaho. *In* Proc. Seventh Int. Conf. Low-Volume Roads, 23-26 May 1999, Baton Rouge, LA. Transport. Res. Rec. **1652**: 235–243.

McColl, J.G., and Powers, R.F. 1984. Consequences of forest management on soil-tree relationships. *In* Nutrition of Plantation Forests. *Edited by* G.D. Bowen, and E.K. S. Nambiar. Academic Press, London. pp. 379-412.

McDonald, L.H., and Huffman, E.L. 2004. Postfire soil water repellency: persistance and soil moisture thresholds. Soil Sci. Soc. Am. J. **68**: 1729-1734.

McIver, J.D. 2004. Sediment transport and soil disturbance after postfire logging. Hydrol. Sci. Technol. **20**: 101-111.

McIver, J.D., and McNeil, R. 2006. Soil disturbance and hill-slope sediment transport after logging of a severely burned site in northeastern Oregon. West. J. Appl. For. **21**: 123-133.

McIver, J., and Ottmar, R. 2006. Fuel mass and stand structure after postfire logging of a severely burned ponderosa pine forest in northeastern Oregon. For. Ecol. Manage. In press.

McIver, J.D., and Starr, L. 2001. Environmental effects of postfire logging: literature review and annotated bibliography. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-486.

McIver, J.D., and Starr, L. 2001. A literature review on the environmental effects of postfire logging. West. J. Appl. For. **16**:159-168.

McKenzie, D., Peterson, D.L., and Alvarado, E. 1996. Extrapolation problems in modeling fire effects at large spatial scales: a review. Int. J. Wildl. Fire **6**: 165-176.

McKenzie, D., Gedalof, Z., Peterson, D.L., and Mote, P. 2004. Climatic change, wildfire, and conservation. Conserv. Biol. **18**: 890-902.

Megahan, W.F. 1980. Nonpoint source pollution from forestry activities in the western United States: results of recent research and research needs. *In* U.S. forestry and water quality: what course in the 80s?, Proc. Water Pollution Control Federation, 19 June 1980, Richmond, Va. pp. 93-151.

Mellen, K., Marcot, B.G., Ohmann, J.L., Waddell, K., Livingston, S., Willhite, E., Hostetler, B., Ogden, C., and Dreisbach, T. 2002. DecAID, the decayed wood advisor for managing snags, partially dead trees, and down wood for biodiversity in forests of Washington and Oregon. USDA Forest Service, Pacific Northwest Region and Pacific Northwest Research Station; USDI Fish and Wildlife Service, Oregon State Office, Portland, Ore. Available from http://wwwnotes.fs.fed.us:81/pnw/DecAID/DecAID.nsf [accessed 29 May 2006].

Minshall, G.W., Robinson, C.T., and Lawrence, D.E. 1997. Immediate and mid-term responses of lotic ecosystems in Yellowstone National Park, USA to wildfire. Can. J. Fish. Aquat. Sci. **54**: 2509-2525.

Minshall, G.W. 2003. Responses of stream benthic invertebrates to fire. For. Eco. Manage. **178**: 155-161.

Murphy, M. L., and J. D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Can. J. Fish. Aquat. Sci. **38**: 137-145.

Murphy, M.L., Hawkins, C.P., and Anderson, N.H. 1981. Effects of canopy modification and accumulated sediment on stream communities. Trans. Amer. Fish. Soc. **110**: 469-478.

Murphy, M.L. and Koski, K.V. 1989. Input and depletion of large woody debris in Alaska streams and implications for streamside management. N. Amer. J. Fish. Manage. **9**: 427-436.

Neary, D.G., Ffolliott, P.F., and Landsberg, J.D. 2005a. Fire and streamflow regimes. *In* Wildland fire in ecosystems: effects of fire on soils and water. *Edited by* D.G. Neary, K.C. Ryan, and L.F. DeBano. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-42-vol.4. Chapter 8.

Neary, D.G. Klopatek, C.C., DeBano, L.F., and Ffolliott, P.F. 1999. Fire effects onf belowground sustainability: a review and synthesis. For. Ecol. Manage. **122**: 61-71.

Neary, D.G., Landsberg, J.D., Tiedemann, A.R., and Ffolliott, P.F. 2005b. Water quality. *In* Wildland fire in ecosystems: effects of fire on soils and water. *Edited by* D.G. Neary, K.C. Ryan, and L.F. DeBano. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-42-vol.4. Chapter 6.

Neary, D.G., Ryan, K.C., and DeBano, L.F. 2005c. Wildland fire in ecosystems: effects of fire on soils and water. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-42-vol.4.

Newland, J.A., and DeLuca, T.H. 2000. Influence of fire on native nitrogen-fixing plants and soil nitrogen status in ponderosa pine-Douglas-fir forests in western Montana. Can. J. For. Res. **30**: 274-282.

Newton, M., Fitzgerald, S., Rose, R.R., Adams, P.W., Tesch, S.D., Sessions, J., Atzet, T., Powers, R.F., and Skinner, C. 2006. Comment on "Post-wildfire logging hinders regeneration and increases fire risk." Science **313**: 615.

Nickless, R., Boldt, E., and Neesvig, C. 2002. Rainfall events and their effect on severely burned areas of western Montana following the forest fires of 2000. National Weather Service. Western Reg. Tech. Attach. 02-12. Available from http://www.wrh.noaa.gov/wrh/02TAs/0212/index.html [accessed 21 March 2006].

Norman, S.N., and Taylor, A.H. 2003. Tropical and north Pacific teleconnections influence fire regimes in pine-dominated forests of northeastern California. J. Biogeog. **30**: 1081-1092.

Oliver, C.D., and Larson, B.C. 1996. Forest Stand Dynamics. John Wiley and Sons, New York, N.Y.

Oregon Department of Forestry. 1983. From Tillamook Burn to Tillamook State Forest. Oregon Department of Forestry, Salem.

Parrett, C., Cannon, S.H., and Pierce, K.L. 2004. Wildfire-related floods and debris flows in Montana in 2000 and 2001. U.S. Geol. Surv. Water Resour. Invest. Rep. 03-4319.

Passovoy, M.D., and Fulé, P.Z. 2006. Snag and woody debris dynamics following severe wildfires in northern Arizona ponderosa pine forests. For. Ecol. Manage. **223**: 237-246.

Pearson, S.F., and Manuwal, D.A. 2001. Breeding bird response to riparian buffer width in managed Pacific Northwest Douglas-fir forests. Ecol. Applic. **11**: 840-853.

Peterson, D.L., and Arbaugh, M.J. 1986. Postfire survival in Douglas-fir and lodgepole pine: comparing the effects of crown and bole damage. Can. J. For. Res. **16**: 1175-1179.

Peterson, D.L., and V.T. Parker. 1998. Ecological scale: theory and applications. Columbia University Press, New York.

Peterson, D.L., Evers, L., Gravenmier, R.A., and Eberhardt, E. 2006. Analytical and decision support for science-based management of vegetation and fuels. USDA For. Serv. Gen. Tech. Rep. PNW-GTR-xxx.

Peterson, D.L., Johnson, M.C., McKenzie, D., Agee, J.K., Jain, T.B., and Reinhardt, E.D. 2005. Forest structure and fire hazard in dry forests of the western United States. USDA For. Serv. Gen. Tech. Rep. GTR-PNW-628.

Pilliod, D.S., Bury, R.B., Hyde, E.J., Pearl, C.A., and Corn, P.S. 2003. Fire and amphibians in North America. For. Ecol. Manage. **178**: 163-181.

Poff, R.J. 1989. Compatibility of timber salvage operations with watershed values. *In* Proceedings Symposium on Fire and Watershed Management. USDA Forest Service Gen. Tech. Rep. PSW-109. pp. 137-140.

Pollet, J., and Omi, P.N. 2002. Effect of thinning and prescribed burning on crown fire severity in ponderosa pine forest. Int. J. Wildl. Fire **11**: 1-10.

Potts, D.F., Peterson, D.L., and Zuuring, H.R. 1985. Watershed modeling for fire management planning in the northern Rocky Mountains. USDA For. Serv. Res. Pap. PSW-177.

Powers, R.F. 1989. Do timber management practices degrade long-term site productivity? *In* Eleventh Annual Forest Vegetation Management Conference. Forest Vegetation Management Conference, Redding, Calif. pp. 87-106.

Powers, R.F. 2002. Effects of soil disturbance on fundamental, sustainable productivity of managed forests. *In* The Kings River Sustainable Ecosystem Project. *Edited by* J. Verner. USDA For. Serv. Gen. Tech. Rep. PSW-GTR-183. pp. 63-82.

Powers, R.F. 2006. Long-Term Soil Productivity: genesis of the concept and principles behind the program. Can. J. For. Res. **36**: 519-528.

Powers, R.F., Scott, D.A., Sanchez, F.G., Voldseth, R.A., Page-Dumroese, D., Elioff, J.D., Stone, D.M. 2005. The North American long-term soil productivity experiment: findings from the first decade of research. For. Ecol. Manage. **220**: 31-50.

Powers, R.F., and Edmonds, R.L. 1992. Nutrient management of subalpine *Abies* forests. *In* Forest fertilization: sustaining and improving nutrition and growth of western forests. *Edited by* H.N. Chappell, G.F. Weetman, and R.E. Miller. Institute of Forest Resources Contribution 73, College of Forest Resources, University of Washington, Seattle. pp. 28-42.

Raphael, M.G., and White, M. 1984. Use of snags by cavity-nesting birds in the Sierra Nevada. Wildl. Monogr. **86**: 1-66.

Raymond, C.L., and Peterson, D.L. 2005. Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. Can. J. For. Res. **35**: 2981-2995.

Reeves, G.H., Benda, L., Burnett, K., Bisson, P., and Sedell, J. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. Amer. Fish. Soc. Symp. **17**: 334-349.

Reeves, G.H., Bisson, P.A., Rieman, B.E., and Benda, L.E. 2006. Postfire logging in riparian areas. Conserv. Biol. **20**: 994-1004.

Reinhardt, E.D., and Crookston, N.L. 2003. The Fire and Fuels Extension to the Forest Vegetation Simulator. USDA For. Serv. Gen. Tech. Rep. GTR-RMRS-116.

Rice, R.M., Rothacher, J.S., and Megahan, W.F. 1972. Erosional consequences of timber harvesting: an appraisal. *In* Proceedings National Symposium on Watersheds in Transition. American Water Resources Association, Fort Collins, Colo. pp. 321-329.

Rieman, B.E., and Clayton, J. 1997. Wildfire and native fish: issues of forest health and conservation of sensitive species. Fisheries **22**: 6-15.

Rieman, B.E., Lee, D., Burns, D., Gresswell, R., Young, M., Stowell, R., and Howell, P. 2003. Status of native fishes in the Western United States and issues for fire and fuels management. For. Ecol. Manage. **178**: 19-212.

Robichaud, P.R., Byers, J.L., and Neary, D.G. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-63.

Rose, C., Marcot, B., Mellen, T.K., Ohmann, J.L., Waddell, K., Lindley, D., and Schreiber, B. 2001. Decaying wood in Pacific Northwest Forests: concepts and tools for habitat management. *In* Wildlife-habitat relationships in Oregon and Washington. Edited by D.H. Johnson and T.A. O'Neil. Oregon State University Press, Corvallis. pp. 580-623.

Rothermel, R.C. 1972. A mathematical model for predicting fire spread in wildland fuels. USDA For. Serv. Res. Pap. INT-115.

Rowe, J.S. 1981. Concepts of fire effects on plant individuals and species. *In* The role of fire in northern circumpolar ecosystems. *Edited by* R.W. Wein, and D.A. Maclean. John Wiley, New York.

Roy, D. 1956. Salvage logging may destroy Douglas-fir reproduction. USDA For. Serv. Res. Note 107.

Russell, R.E., Saab, V.A., Dudley, J.G., and Rotella, J.J. 2006. Snag longevity in relation to wildfire and postfire salvage logging. For. Ecol. Manage. **232**: 179-187.

Ryan, K.C., and Reinhardt, E.D. 1988. Predicting postfire mortality of seven western conifers. Can. J. For. Res. **18**: 1291-1297.

Saab, V., and Dudley, J. 1998. Response of cavity-nesting birds to stand-replacement fire and salvage logging in ponderosa pine/Douglas-fir forests of southwestern Idaho. USDA For. Serv. Res. Pap. RMRS-RP-11.

Saab, V., and Powell, H. (editors). 2005. Fire and avian ecology in North America. Stud. Avian Biol. **30**:1-193.

Saab, V., Dudley, J., and Thompson, W. 2004. Factors influencing occupancy of nest cavities in recently burned forests. Condor **106**: 20-36.

Saab, V.A., Powell, H.D., Kotliar, N.B., and Newlon, K.R. 2005. Variation in fire regimes of the Rocky Mountains: implications for avian communities and fire management. Stud. Avian Biol. **30**: 76-96.

Saab, V., Brannon, R., Dudley, J., Donohoo, L., Vanderzanden, D., Johnson, V., and Lachowski, H. 2002. Selection of fire-created snags at two spatial scales by cavity-nesting birds. *In* Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests. *Tech. coord. by* W.F. Laudenslayer, P.J. Shea, B.E. Valentine, C.P. Weatherspoon, and T.E. Lisle. USDA For. Serv. Gen. Tech. Rep. PSW-GTR-181. pp. 835-848

Schmoldt, D.L., Peterson, D.L., Keane, R.E., Lenihan, J.M., McKenzie, D., Weise, D.R., and Sandberg, D.V. 1999. Assessing the effects of fire disturbance on ecosystems: a scientific agenda for research and management. USDA For. Serv. Gen. Tech. Rep. GTR-PNW-455.

Schoennagel, T.L., and Waller, D.M. 1999. Understory responses to fire and artificial seeding in an eastern Cascades *Abies grandis* forest. Can. J. For. Res. **29**: 1393-1401.

Schultz, A.M., Launchbaugh, J.L., and Biswell, H.H. 1955. Competition among grasses reseeded on burned brushlands in California. J. Range Manage. **5**: 338-345.

Scott, J.H., and Burgan, R.E. 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-153.

Scott, J.H., and Reinhardt, E.D. 2003. Assessing crown fire potential by linking models of surface and crown fire behavior. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-153.

Sessions, J., Bettinger, P., Buckman, R., Newton, M., and Hamann, J. 2004. Hastening the return of complex forests following fire: the consequences of delay. J. For. **102**: 38-45.

Sexton, T.O. 1998. Ecological effects of post-wildfire salvage logging on vegetation diversity, biomass, and growth and survival of *Pinus ponderosa* and *Purshia tridentata*. Master's thesis, Oregon State University, Corvallis, Ore.

Shakesby, R.A., Boakes, D.J., Coelho, C.deO.A., Gonçalves, A.J.B., and Walsh, R.P.D. 1996. Limiting soil degradational impacts of wildfire in pine and eucalyptus forest in Portugal: a comparison of alternative postfire management practices. *Appl. Geogr*. **16**: 337-355.

Shepperd, W.D., and Battaglia, M.A. 2002. Ecology, silviculture, and management of Black Hills ponderosa pine. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-97.

Sidle, R.C. 1980. Impacts of forest practices on surface erosion. Oreg. State Univ. Exten. Serv. Pub. PNW 195.

Simon, J., Christy, S., and Vessels, J. 1994. Clover-Mist Fire recovery: a forest management response. J. For. **92**: 41-44.

Skinner, C.N. 2002. Influence of fire on the dynamics of dead woody material in forests of California and southwestern Oregon. *In* Proceedings of the Symposium on the Ecology and Management of Dead Wood in Western Forests. *Tech. coord. by* W.F. Laudenslayer, P.J. Shea, B.E. Valentine, C.P. Weatherspoon, and T.E. Lisle. USDA For. Serv. Gen. Tech. Rep. PSW-GTR-181. pp. 445-454.

Smith, J.K. 2000. Wildland fire in ecosystems: effects of fire on fauna. USDA For. Serv. Gen. Tech. Rep. RMRS-GTR-42 vol. 1.

Smith, T.G., and Maguire, C.C. 2004. Small-mammal relationships with down wood and antelope bitterbrush in ponderosa pine forests of central Oregon. For. Sci. **50**: 711-728.

Smucker, K.M., Hutto, R.L., and Steele, B.M. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecol. Applic. **15**: 1535-1549.

Society for Ecological Restoration. 2004. The SER international primer on ecological restoration [online]. Available from http://www.ser.org/content/ecological_restoration_primer.asp [accessed 3 August 2006].

Soil Classification Working Group. 1998. The Canadian System of Soil Classification, 3rd edition. Research Branch, Agriculture and Agri-Food, Canada, Ottawa.

Soil Survey Staff. 1996. Keys to Soil Taxonomy. U.S. Department of Agriculture, Washington, D.C.

Spies, T.A., Franklin, J.F., and Thomas, T.B. 1988. Coarse woody debris in Douglas-fir forests of western Oregon and Washington. Ecol. **69**: 1689-1702.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. J. Hydrol. **176**: 79-95.

Stein, S.J. 1988. Explanations of imbalanced age structure and scattered distribution of ponderosa pine within a high-elevation mixed coniferous forest. For. Ecol. Manage. **25**: 139-153.

Stepnisky, D. 2003. Response of *Picoides* woodpecders to salvage harvesting of burned, mixedwood boreal forest: exploration of pattern and process. Master's thesis, University of Alberta, Edmonton.

Stickney, P.F. 1986. First decade plant succession following the Sundance forest fire, northern Idaho. USDA For. Serv. Gen. Tech. Rep. INT-197.

Stuart, J.D., Grifantini, M.C., and Fox, L. 1993. Early successional pathways following wildfire and subsequent silvicultural treatment in Douglas-fir/hardwood forests, NW California. For. Sci. **39**: 561-572.

Taylor, A.H., and Skinner, C.N. 1998. Fire history and landscape dynamics in a late-successional reserve in the Klamath Mountains, California, USA. For. Ecol. Manage. **11**: 285-301.

Taylor, A.H., and Skinner, C.N. 2003. Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecol. Applic. **13**: 704-719.

Thomas, J., Anderson, R., Maser, C., and Bull, E.L. 1979. Snags. *In* Wildlife habitats in managed forests – the Blue Mountains of Oregon and Washington. *Edited by* J.W. Thomas. USDA For. Serv. Agric. Handbook 553. pp. 60–77.

Tobalske, B.W. 1997. Lewis' woodpecker (*Melanerpes lewis*). *In* Birds of North America, no. 284. *Edited by* A. Poole and F. Gill. Academy of Natural Sciences, Philadelphia, Penn., and American Ornithologists Union, Washington, D.C.

Troendle, C.A., Wilcox, M.S., Bevenger, G.S., and Porth, L.S. 2001. The Coon Creek water yield augmentation project: implementation of timber harvesting technology to increase streamflow. For. Ecol. Manage. **143**: 179-187.

Trombulak, S.C., and Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conserv. Biol. **14**: 18-30.

Turner, M.G., Romme, W.H., and Gardner, R.H. 1999. Prefire heterogeneity, fire severity, and early postfire plant reestablishment in subalpine forests of Yellowstone National Park, Wyoming. Int. J. Wildl. Fire **9**: 21-36.

USDA Forest Service. 1976. Resistance to control calculation matrix. USDA For. Serv. Manual, Region 5 Supplement 5151. San Francisco, Calif.

USDA Natural Resources Conservation Service. 1986. Urban hydrology for small watersheds. USDA Nat. Res. Conserv. Serv. Conserv. Engin. Div. Tech. Rel. 55.

Van Cleve, K., and Powers, R.F. 1995. Soil carbon, soil formation, and ecosystem development. *In* Carbon Forms and Functions in Forest Soils. *Edited by* W.W. McFee, and J.M. Kelly. Soil Science Society of America, Madison, Wis. pp. 155-200.

van Wagtendonk, J.W. 1985. Fire suppression effects on fuels and succession in short-fire-interval wilderness ecosystems. In Proc. Symp. and Workshop on Wilderness Fire. Tech. coord. by J.E. Lotan, B.M. Kilgore, W.C. Fischer, and R.W. Mutch. USDA For. Serv. Gen. Tech. Rep. INT-182. pp. 119–126.

Wemple, B.C., Jones, J.A., and Grant, G.E. 1996. Channel network extension by logging roads in two basins, western Cascades, Oregon. Water Resour. Bull. **35**: 555-567.

Wondzell, S.M., and King, J.G. 2003. Postfire erosional processes in the Pacific Northwest and Rocky Mountain region. For. Ecol. Manage. **178**: 75-87.

Wright, C.S., and Agee, J.K. 2004. Fire and vegetation history in the eastern Cascade mountains, Washington. Ecol. Applic. **14**: 443-459.

Wright, P.J. 1998. The effect of fire regime on coarse woody debris in the west central Cascades, Oregon. Master’s thesis, Oregon State University, Corvallis.

Zedler, P.H., Gauthier, C.R., and McMaster, G.S. 1983. Vegetation changes in response to extreme events: the effect of a short interval between fires in California chaparral and coastal scrub. Ecology **64**: 809-818.

Table 1. Summary of scientific literature on the effects of postfire timber harvest. From McIver and Starr (2001) and other recent sources.

Citation	Forest type (location*)	Effect studied
Roy (1956)	Douglas-fir (California)	Tree regeneration
Klock (1975)	Ponderosa pine – mixed conifer (Washington)	Soil erosion, vegetation cover
Helvey (1980)	Ponderosa pine – mixed conifer (Washington)	Soil erosion, water runoff
Blake (1982)	Ponderosa pine (Arizona)	Shrub abundance & cover
Helvey et al. (1985)	Ponderosa pine – mixed conifer (Washington)	Soil erosion, soil nutrients
Potts et al. (1985)	Conifer forest (Rocky Mountains)	Water yield, soil erosion
Grifantini (1990)	Douglas-fir – hardwood (California)	Grass cover
Marston & Haire (1990)	Conifer forest (Wyoming)	Soil erosion, water runoff, organic matter & litter
Grifantini et al. (1992)	Douglas-fir – hardwood (California)	Shrub & forb cover, plant diversity
Stuart et al. (1993)	Douglas-fir – hardwood (California)	Vegetation cover (shrubs, forbs, ferns, grasses, hardwoods, conifers
Chou et al. (1994a,b)	Montane conifer (California)	Soil erosion
Caton (1996)	Mixed conifer (Montana)	Cavity-nesting birds
Hitchcox (1996)	Mixed conifer (Montana)	Cavity-nesting birds

Hejl & McFadzen (1998)	Mixed conifer (Idaho, Montana)	Cavity-nesting birds
Saab & Dudley (1998)	Ponderosa pine, Douglas-fir (Idaho)	Cavity-nesting birds
Sexton (1998)	Ponderosa pine (Oregon)	Vegetation biomass & diversity, tree & shrub growth & diversity, soil moisture
Haggard & Gaines (2001)	Douglas-fir, ponderosa pine	Cavity-nesting birds
Khetmalas et al. (2002)	Subalpine fir (British Columbia, Canada)	Ectomycorrhizae & bacteria in soil
Hanson & Stuart (2005)	Douglas-fir – hardwood (California)	Vegetation composition & structure
Donato et al. (2006)	Douglas-fir (Oregon)	Tree regeneration, woody debris
McIver & McNeil (2006)	Ponderosa pine (Oregon)	Soil disturbance & erosion
McIver & Ottmar (2006)	Ponderosa pine (Oregon)	Tree regeneration, woody debris, fuels

* All locations are in the United States except for Khetmalas et al. (2002).

Table 2. Site and standing biomass characteristics of several mature forest stands in western North America. Canadian soil classification based on Soil Classification Working Group (1998); U.S. soils classification based on Soil Survey Staff (1996). Modified from Powers (2006).

Location	Forest type	Elev. (m)	Annual precip. (cm)	Soil origin	Soil family	Stand age (yr)	Standing biomass ($kg\ ha^{-1}$)		
							Overstory	Understory	Forest floor
British Columbia	Aspen	720	48	Glacial fluvial	Gleyed Gray Luvisol	100	520,000	nd	54,913
British Columbia	Mixed conifer	785	62	Glacial till	Orthic Humo-Ferric Podzol	140	174,800	nd	78,230
British Columbia	Mixed conifer	1050	43	Glacial till	Orthic Gray Luvisol	110	64,400	nd	41,065
British Columbia	Mixed conifer	1100	53	Glacial till	Orthic Gray Luvisol	140	169,600	nd	75,329
California	Mixed conifer	1798	67	Volcanic ash	Frigid Ultic Haploxeralfs	262	170,815	nd	nd
California	Mixed conifer	1320	165	Volcanic mudflow	Mesic Ultic Haploxeralfs	65	352,224	240	78,724
California	Mixed conifer	1130	190	Volcanic mudflow	Mesic Ultic Haploxeralfs	115	357,453	511	65,587
California	Mixed conifer	1524	91	Vilcanic ash	Frigid Typic Vitrixerands	242	206,663	nd	nd
California	Mixed conifer	1685	114	Granodiorite	Mesic Tyouc Dystroxerepts	117	422,111	94	80,455
California	Mixed conifer	790	173	Metabasalt	Mesic Typic Palexerults	108	473,348	576	60,926
California	Mixed conifer	1959	69	Volcanic ash	Frigid Ultic Haploxeralfs	258	212,633	nd	nd
California	Mixed conifer	1270	173	Volcanic mudflow	Mesic Ultic Haploxeralfs	117	438,176	674	83,820
California	Mixed conifer	1805	114	Granodiorite	Mesic Tyoic Dystroxerepts	115	576,071	34	72,233
California	Mixed conifer	1200	170	Granodiorite	Mesic Pachic Xerumbrepts	112	493,934	324	76,901
California	Mixed conifer	1560	76	Granodiorite	Mesic Typic Dystroxerepts	132	373,609	43	72,567
California	Mixed conifer	1575	178	Volcanic ash	Mesic Andic Xerumbrepts	230	450,193	83	115,757
Idaho	Mixed conifer	900	77	Volcanic ash	Frigid Andic Xerochrepts	120	191,250	1,750	68,000
Idaho	Mixed conifer	1575	68	Basalt	Mesic Typic Hapludand	100	252,000	157	72,450

Note: nd = no data.

Table 3. Absolute (and proportional) amounts of biomass and nitrogen affected by removal through timber harvest in mature forests of western North America. Over 50% of the biomass is in the boles. Between one-third and three-quarters of the nitrogen is in the forest floor. Modified from Powers (2006).

		Biomass removed (Mg ha^{-1}) (*% of above-ground total*)			Nitrogen removed (kg ha^{-1}) (*% of above-ground total*)		
Location	Forest type	Forest floor	Crown	Bole	Forest floor	Crown	Bole
British Columbia	Sub-boreal spruce	65 *(29)*	32 *(14)*	126 *(57)*	815 *(76)*	58 *(5)*	195 *(18)*
Idaho	Mixed conifer	70 *(27)*	31 *(12)*	160 *(61)*	436 *(52)*	220 *(26)*	190 *(22)*
California	Mixed conifer	59 *(11)*	221 *(42)*	252 *(47)*	455 *(43)*	391 *(37)*	218 *(20)*

Table 4. A life history classification of fire adaptations. From Rowe (1981).

Life history type	Fire adaptations
Invaders	Highly dispersive, pioneering fugitives
Evaders	Serotinous cones, refractive seed
Endurers	Sprouters (crown, roots, stem base)
Resisters	Thick bark
Avoiders	None

Table 5. Fine fuel mass (<7.6 cm diameter) differs by site, over time, and by treatment. Each line indicates a separate site within the geographic location.

Location (Reference)	Time since fire (yr)	No harvest (Mg ha^{-1})	Harvest (Mg ha^{-1})	Unburned forest (Mg ha^{-1})
Olympic Mountains,	1	13.5		6.1
NW Washington	3	9.2		6.1
(Agee & Huff 1987)	19	5.3		6.1
	110	5.2		6.1
Siskiyou Mountains,	1	1.2		8.9
SW Oregon	1	3.6		29.7*
(Raymond & Peterson 2005)				
Siskiyou Mountains, SW Oregon (Donato et al. 2006)	3	1.0	6.3	5.9
Blue Mountains, NE Oregon (McIver & Ottmar 2006)	3	1.5	4.3 / 5.4**	

*Post-thinning windthrow event

**Two harvest options (see text)

Table 6. Summary of changes in hydrologic processes produced by wildland fires. After Neary et al. (2005a).

Hydrologic process	Type of change	Specific effect
Interception	Reduction	Moisture storage smaller Greater runoff in small storms Increased water yield
Litter storage of water	Reduced	Less water stored Overland flow increased
Transpiration	Temporary elimination	Stream flow increased Soil moisture increased
Infiltration	Reduced	Overland flow increased Storm flow increased
Stream flow	Changed	Increased in most ecosystems Decreased in snow systems Decreased in fog-drip systems
Base flow	Changed	Decreased (less infiltration) Increased (less evapotranspiration) Summer low flows (+ and -)
Storm flow	Increased	Volume greater Peak flows larger Time to peak flow shorter Flashflood frequency greater Flood levels higher Stream erosive power increased
Snow accumulation	Changed	Fires <4 ha, increased snowpack Fires >4 ha, decreased snowpack Snowmelt rate increased Evaporation and sublimation greater

Table 7. Summary of responses of cavity-nesting bird species to fire, timber harvest, and postfire timber harvest, including supporting evidence. Number of studies listed in parentheses.

Species	Fire	Timber harvest	Fire and timber harvest	Citation
Black-backed woodpecker	Positive (12), especially within 10 yr postfire	Negative (10)	Negative (3)	Hutto 1995, Caton 1996, Hitchcox 1996, Saab & Dudley 1998, Hobson & Schieck 1999, Imbeau et al. 1999, Kreisel & Stein 1999, Kotliar et al. 2002, Stepnisky 2003, Saab et al. 2004
Downy woodpecker	Neutral (6), occasionally positive (2)	Neutral (11), occasionally negative (1)	Neutral (4), occasionally negative (1)	Franzreb & Ohmart 1978, Hutto 1995, Greenberg et al. 1995, Caton 1996, Hitchcox 1996, Hobson & Schieck 1999, Stepnisky 2003
Hairy woodpecker	Positive (11), especially 1-2 yr postfire (2)	Positive (9)	Negative (4)	Franzreb & Ohmart 1978, Hutto 1995, Caton 1996, Hitchcox 1996, Johnson & Wauer 1996, Saab & Dudley 1998, Hobson & Schieck 1999, Kreisel & Stein 1999, Kotliar et al. 2002, Stepnisky 2003, Saab et al. 2004
Lewis' woodpecker	Positive (1)		Positive (2)	Hitchcox 1996, Johnson & Wauer 1996, Saab & Dudley 1998, Saab et al. 2004
Northern flicker	Neutral (7), occasionally positive (5) or negative (1)	Neutral (9), occasionally positive (1)	Neutral (1)	Franzreb & Ohmart 1978, Horton & Mannan 1988, Greenberg et al. 1995, Hutto 1995, Hobson & Schieck 1999, Kotliar et al. 2002
Three-toed woodpecker	Positive (10), especially 1-2 yr postfire (2)	Negative (9), occasionally neutral (1)	Negative (2)	Franzreb & Ohmart 1978, Hutto 1995, Caton 1996, Hitchcox 1996, Johnson & Wauer 1996, Hobson & Schieck 1999, Imbeau et al. 1999, Kreisel & Stein 1999, Kotliar et al. 2002, Stepnisky 2003
White-headed woodpecker	Negative (1)	Negative (1)	Negative (1)	Garrett et al. 1996
Mountain bluebird	Positive (9), especially within 2-10 yr postfire (2)	Positive (7) or neutral (1)	Negative (4)	Raphael & White 1984, Hutto 1995, Caton 1996, Hitchcox 1996, Johnson & Wauer 1996, Saab & Dudley 1998, Hobson & Schieck 1999, Kotliar et al. 2002, Saab et al. 2004
Western bluebird	Neutral (5), occasionally positive or negative (1)	Neutral (9)		Hutto 1995, Johnson & Wauer 1996

Note: All responses indicate number of studies that found an effect (usually $p<0.05$) on abundance or occurrence compared with unburned or unharvested plots. Neutral indicates no significant effect ($p>0.05$) of fire or timber harvest; this category is likely under-represented in the literature. See Hutto (1995) and Kotliar et al. (2002) for additional references.

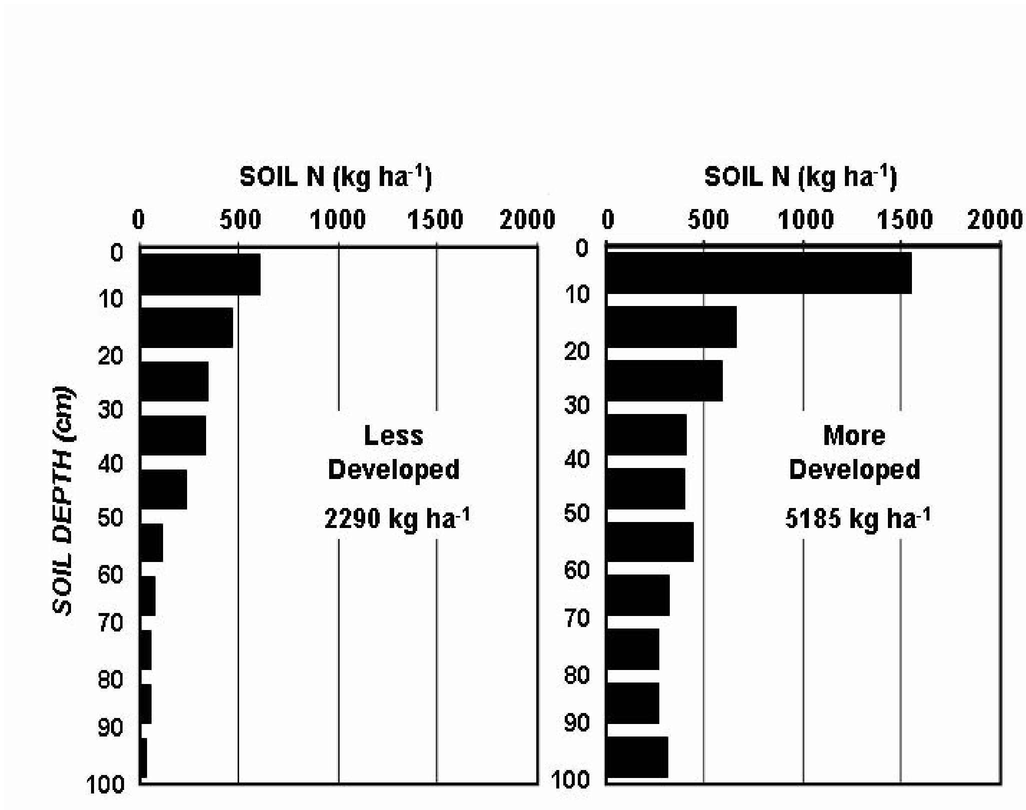

Figure 1. Distribution of organic nitrogen in typical forest soils of the west to a depth of 1 meter. Less-developed soils (Entisols and Inceptisols) usually contain less N than more developed soils (Alfisols and Ultisols), but all show the same pattern of high amounts of N near the surface, declining with depth. Modified from Powers (1989).

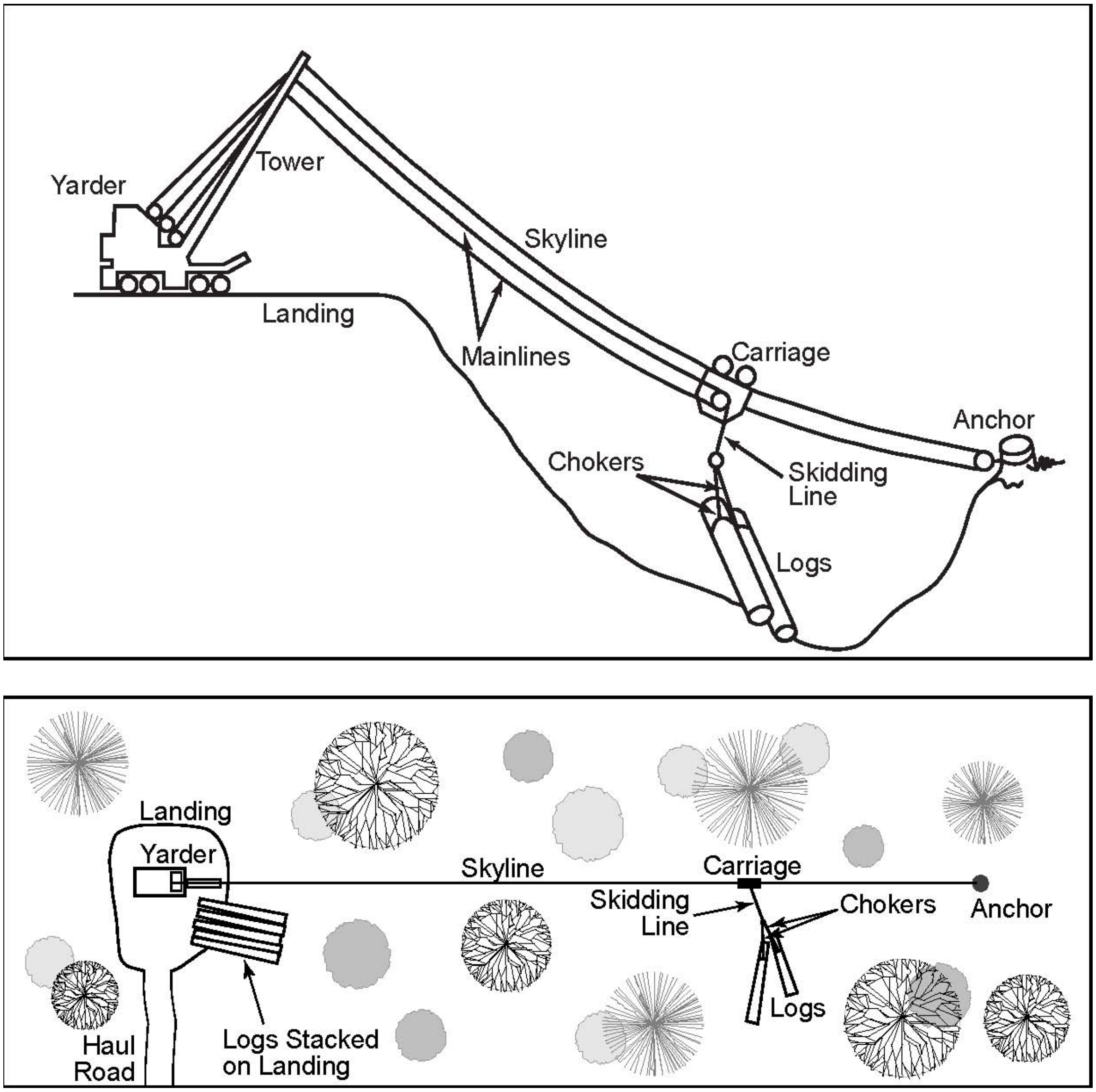

Figure 2. A typical layout for a running skyline yarding system, one of many configurations possible for skyline yarding. The lower image is a map view showing a load of logs being yarded laterally to the skyline corridor in preparation for main yarding to the landing. Neither image is drawn to scale. The type of yarder shown here would typically have a tower 8 to 13 m high, and the distance from the yarder to the anchor could be 600 m or more.

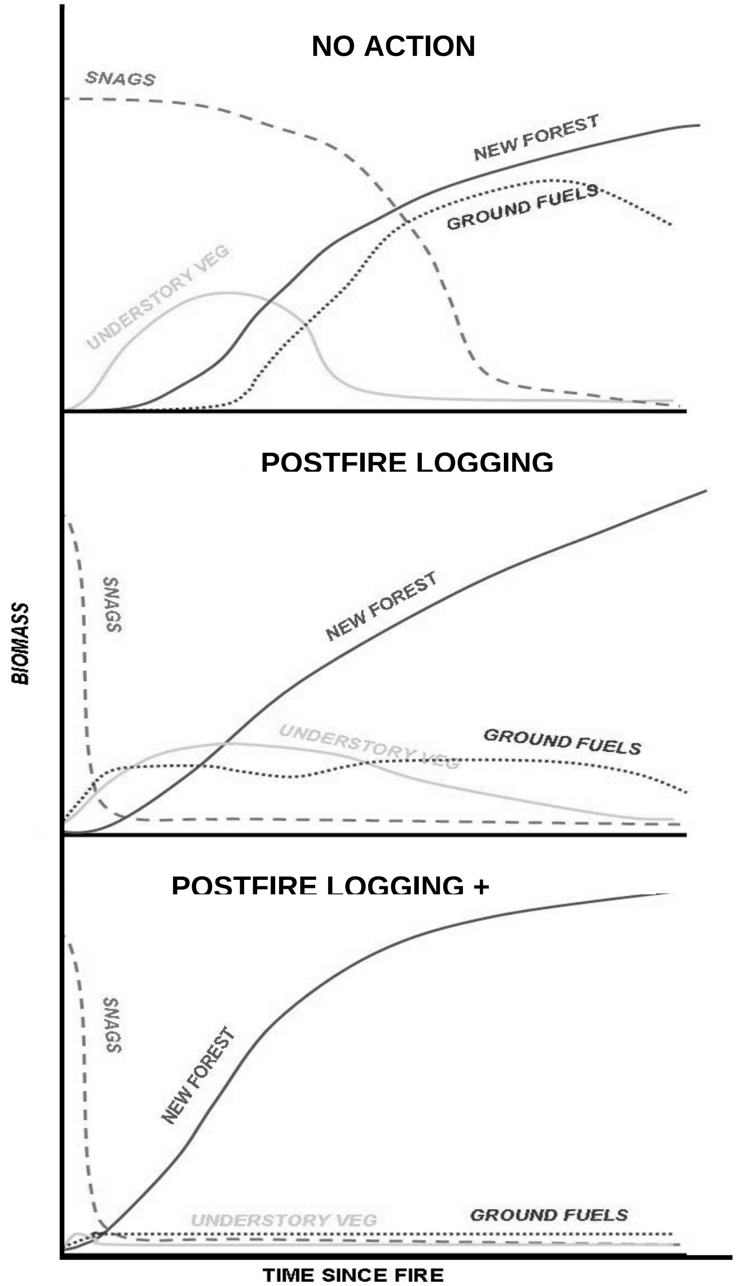

Figure 3. Conceptual diagram of temporal progression in biomass accumulation following wildfire with no management action (top), postfire timber harvest (middle), postfire timber harvest plus mitigation for fire hazard through fuel treatment (bottom). Original figure developed by the authors.

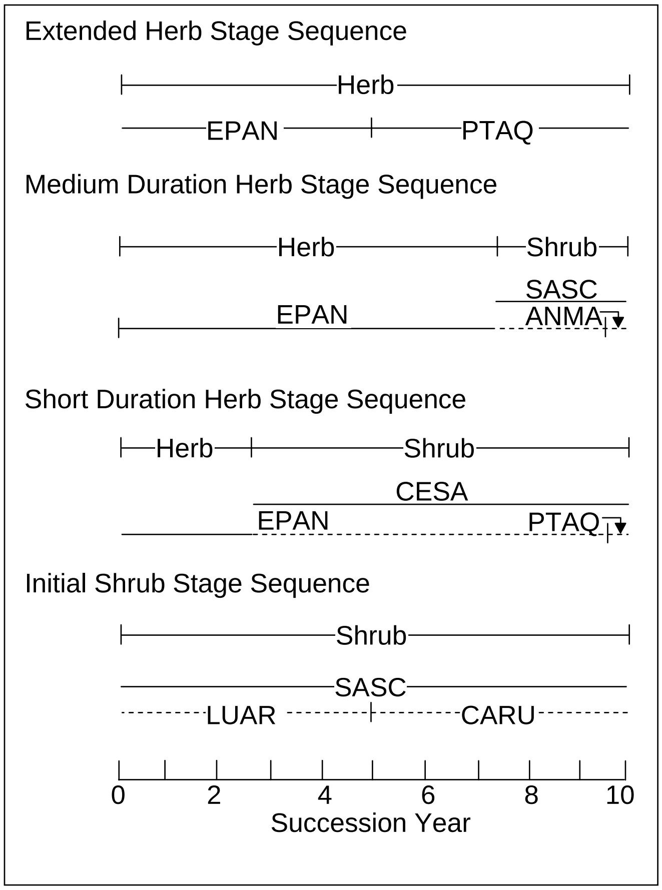

Figure 4. Multiple successional pathways after the Sundance Fire in northern Idaho, where EPAN = fireweed, PTAQ = bracken fern, SASC = Scouler willow, ANMA = pearly everlasting, LUAR = lupine, and CARU = pinegrass. Adapted from Stickney (1986).

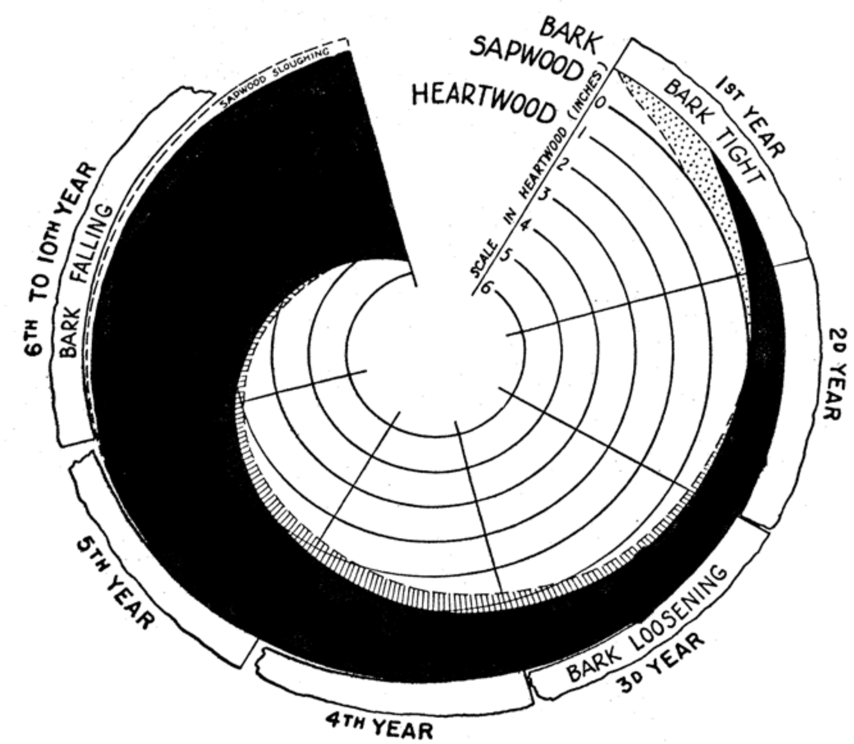

Figure 5. Progress of deterioration in the trunk of an average second-growth Douglas-fir tree killed by fire. Stippling represents deterioration of the sapwood by staining fungi and beetles. The black area represents general deterioration of the heartwood by all causes, primarily decay fungi. Cross hatching shows limited deterioration of the heartwood by borers. This image represents an idealized average for fire-killed trees 60 to 200 years old. From Kimmey and Furniss (1943).

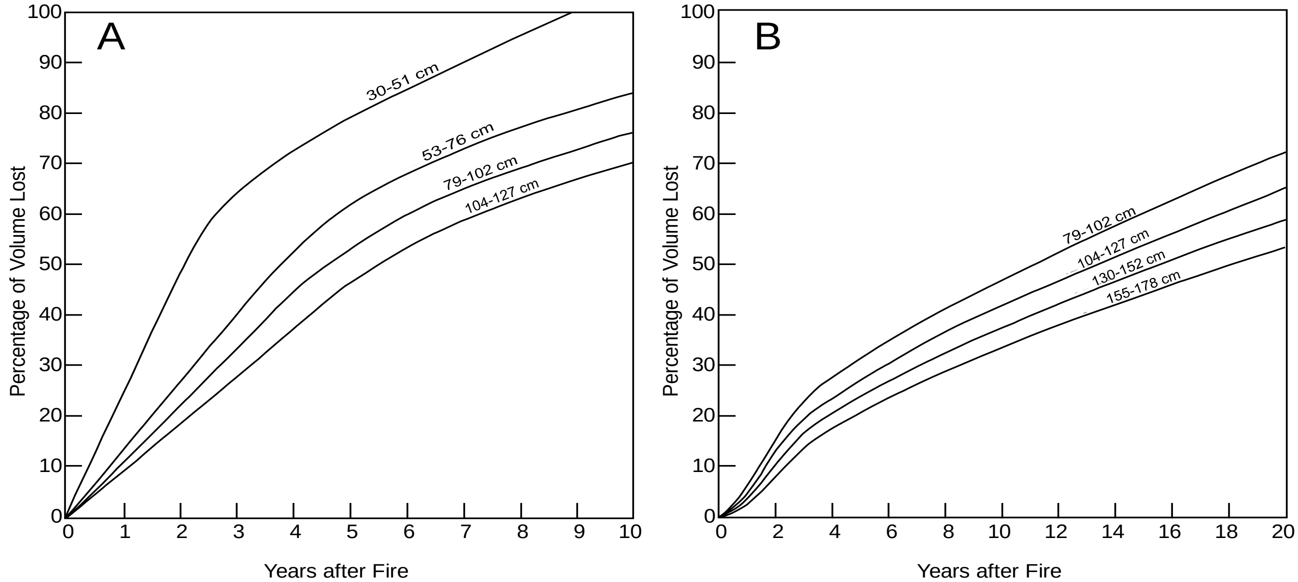

Figure 6. Average rate of loss from general deterioration of the original wood volume in Douglas-fir trees of several dbh classes for (A) second-growth wood (60-200 years old) and (B) old-growth wood (200-400 years). Note that the horizontal axis for A covers only half as many years as that for B, indicating a much faster average deterioration rate for young trees as compared to old-growth trees. From Kimmey and Furniss (1943).

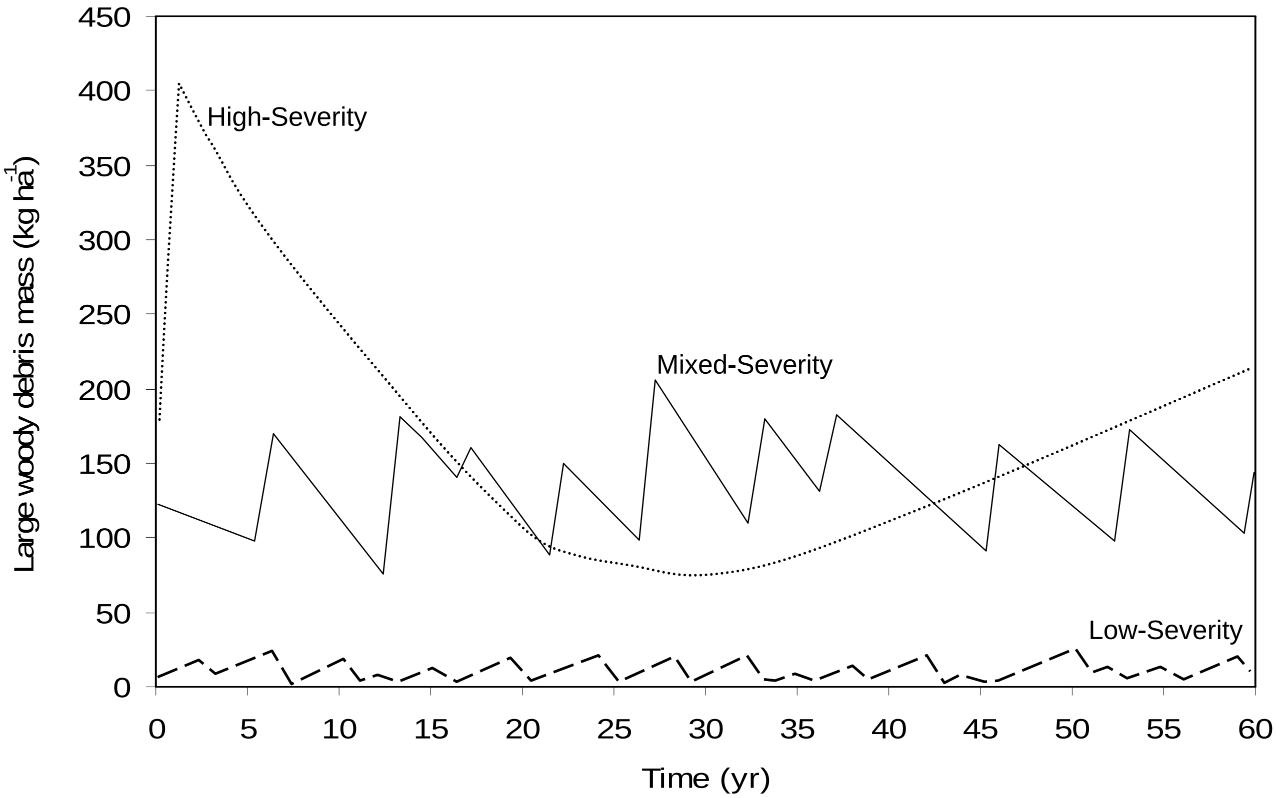

Figure 7. Historical patterns of large woody debris in the three major fire regimes of western North America. Dry forests typically exhibit a low-severity pattern, as frequent fires consume available woody debris every 10 to 20 years. Mesic forests have less frequent fires with patches of low- mixed-, and high-severity fire consuming and creating large woody debris every 25-75 years. Cold and wet forests have a "boom and bust" pattern of large woody debris following infrequent but severe wildfire. This pattern is now more typical of stand-replacement fires in all forest types as a result of several decades of fire exclusion. From Agee (2002a).

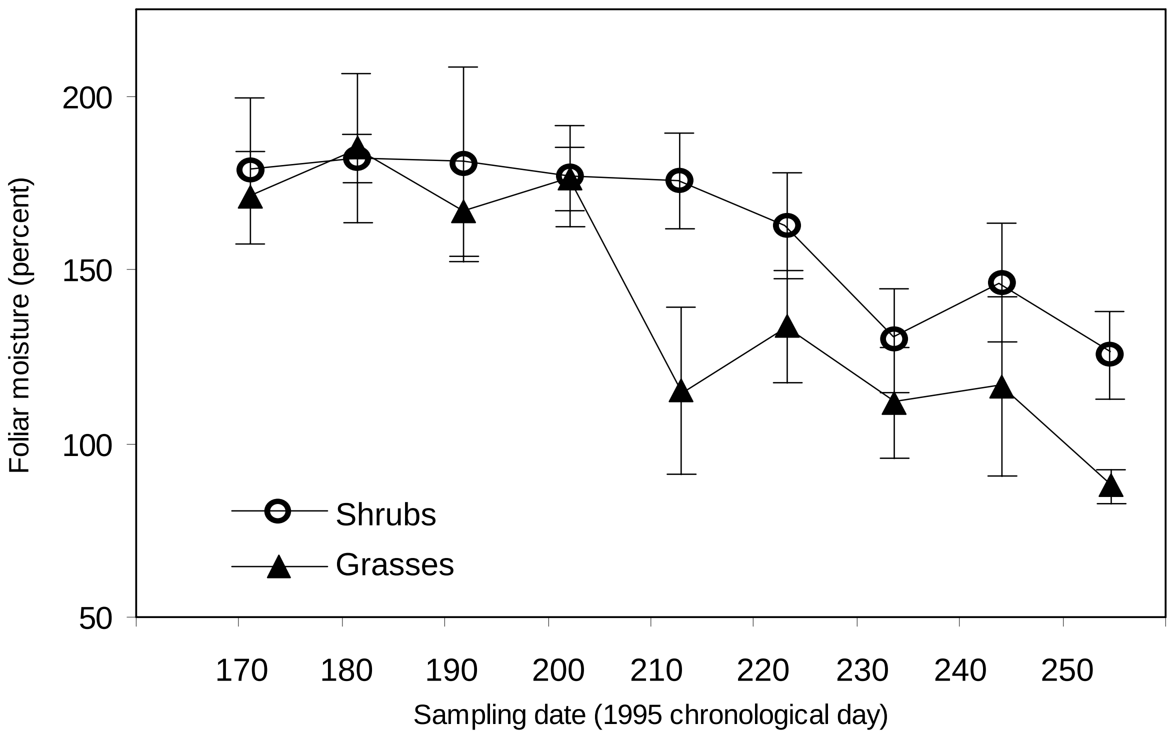

Figure 8. Foliar moisture content of shrubs and grasses (± 1 SD) during a typical summer in the Cascade Range of eastern Washington. From Agee et al. (2002).

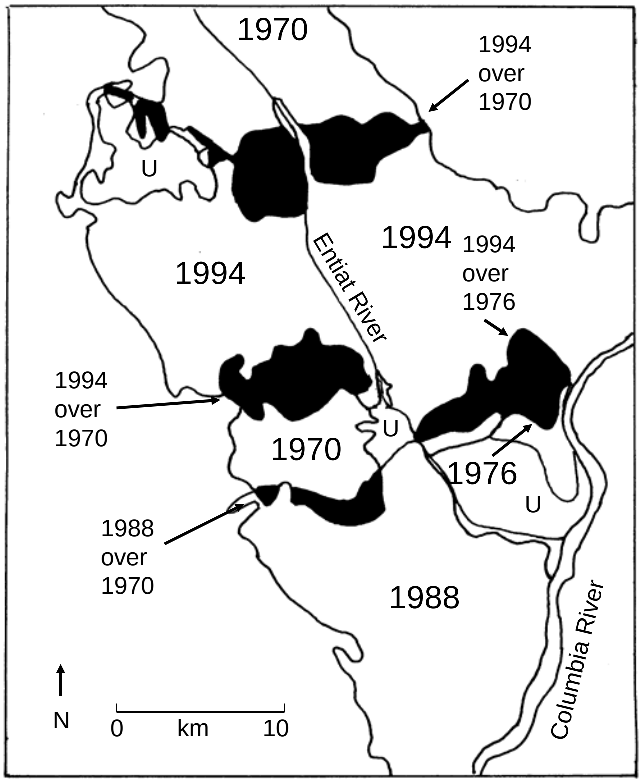

Figure 9. Map of the lower Entiat River, eastern Cascade Range, Washington,vbg showing dates of wildfires and reburns. Small unburned patches within the matrix of burned areas are identified with “U”. Black areas are reburns.

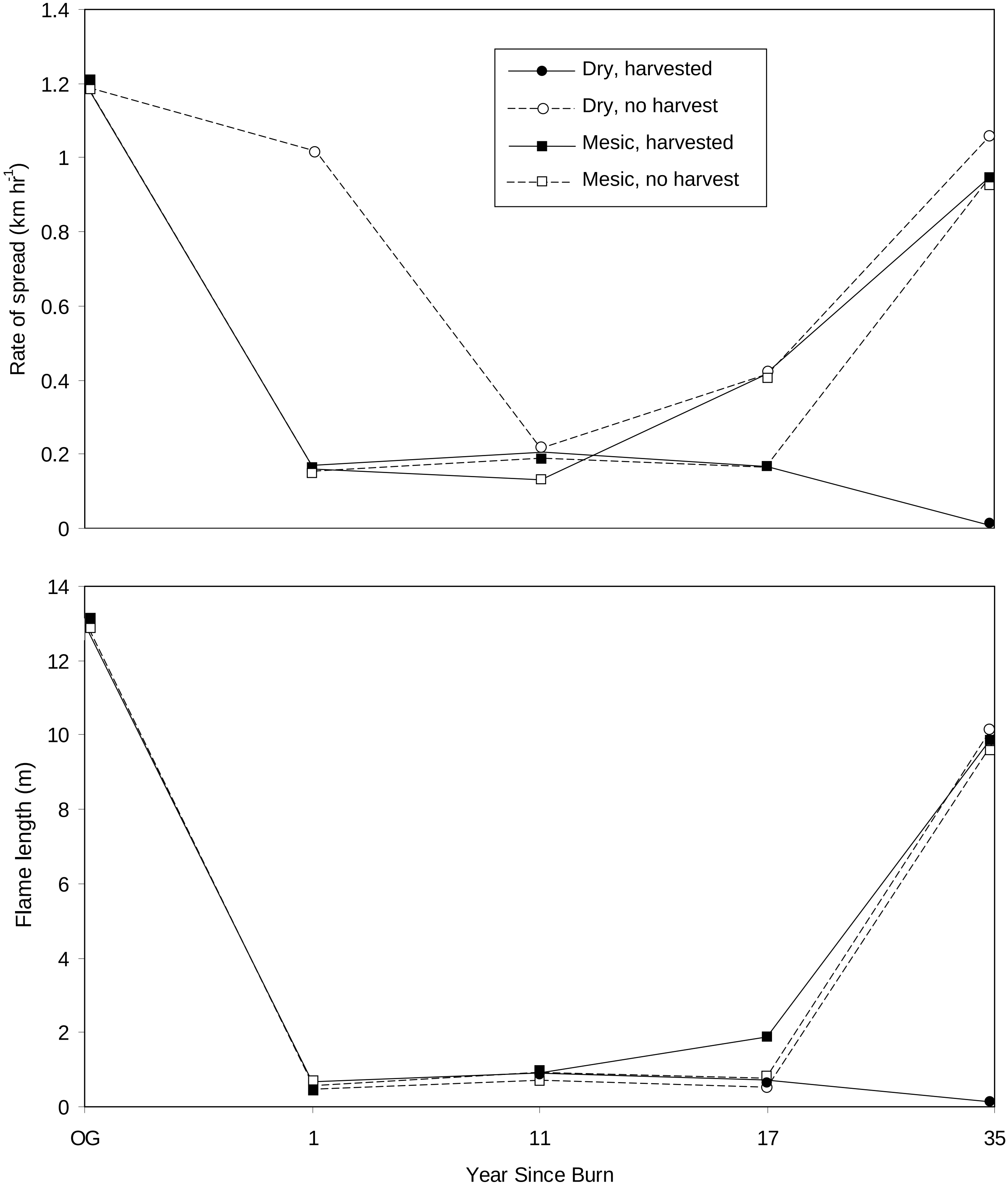

Figure 10. Simulated wildfire rates of spread and flame lengths from unburned forest and forest of postfire ages 1-35 in the Entiat and Wenatchee watersheds, Washington. OG = old growth.

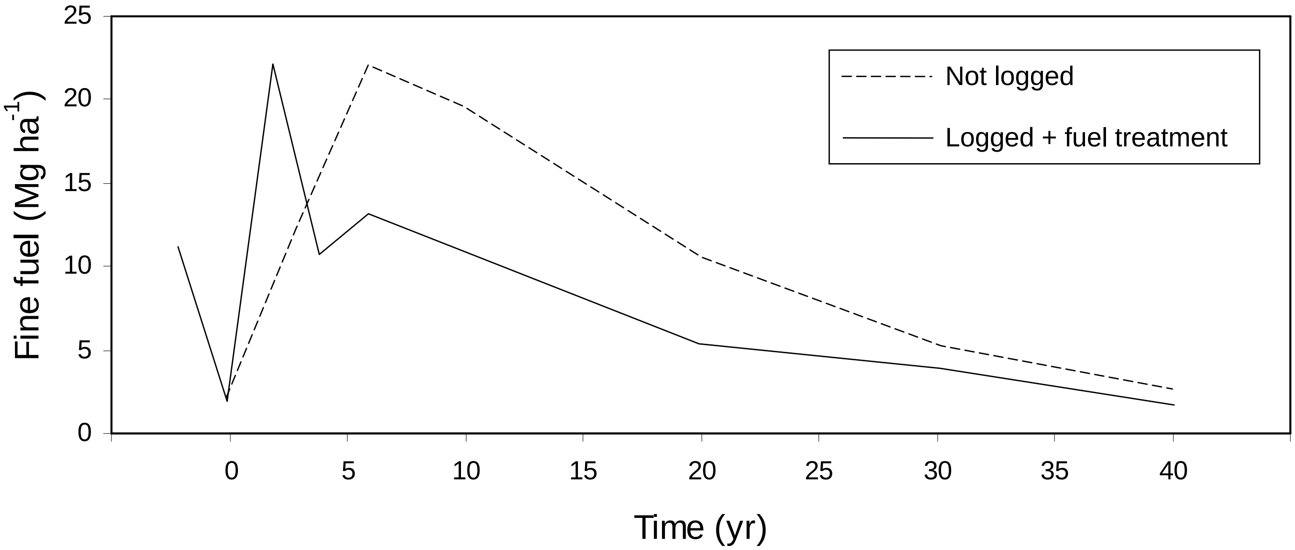

Figure 11. Patterns of fine fuel (< 7.6 cm) after stand-replacing wildfire simulated by FFE-FVS for a dry forest stand on the Bitterroot National Forest (Montana). The harvest treatment here consisted of removing snags >30 cm dbh and <15 cm dbh, with fuels treated by slashing, piling, and burning. From Brown et al. (2003).

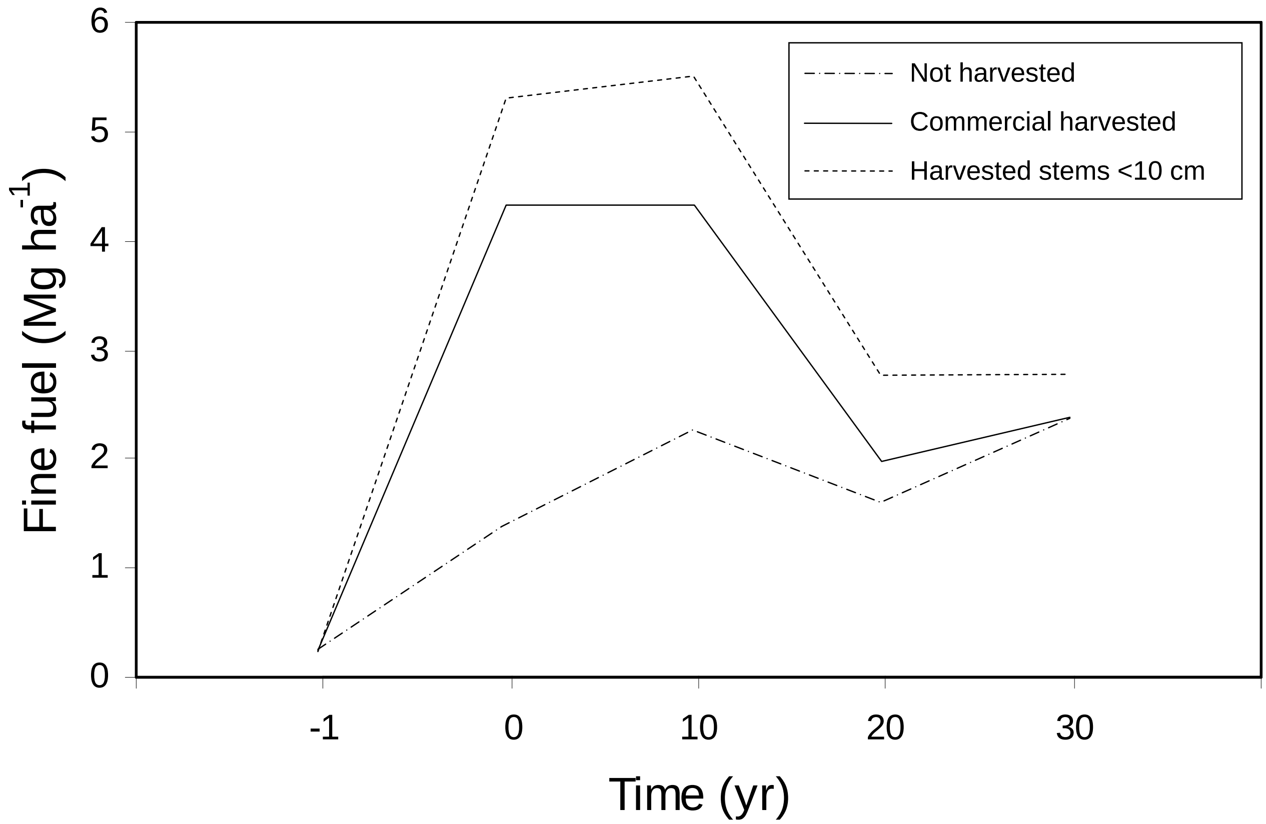

Figure 12. Simulated fine fuel load (<7.6 cm) in a dry forest in the Blue Mountains, Oregon. “Commercial” harvest was defined as removal of most merchantable trees, while “10 cm” harvest was defined as removing most trees <10 cm dbh. No *in situ* slash fuel treatment was applied to either harvest option, but whole trees (including tops and branches) were cable-winched to landings. From McIver and Ottmar (2006).

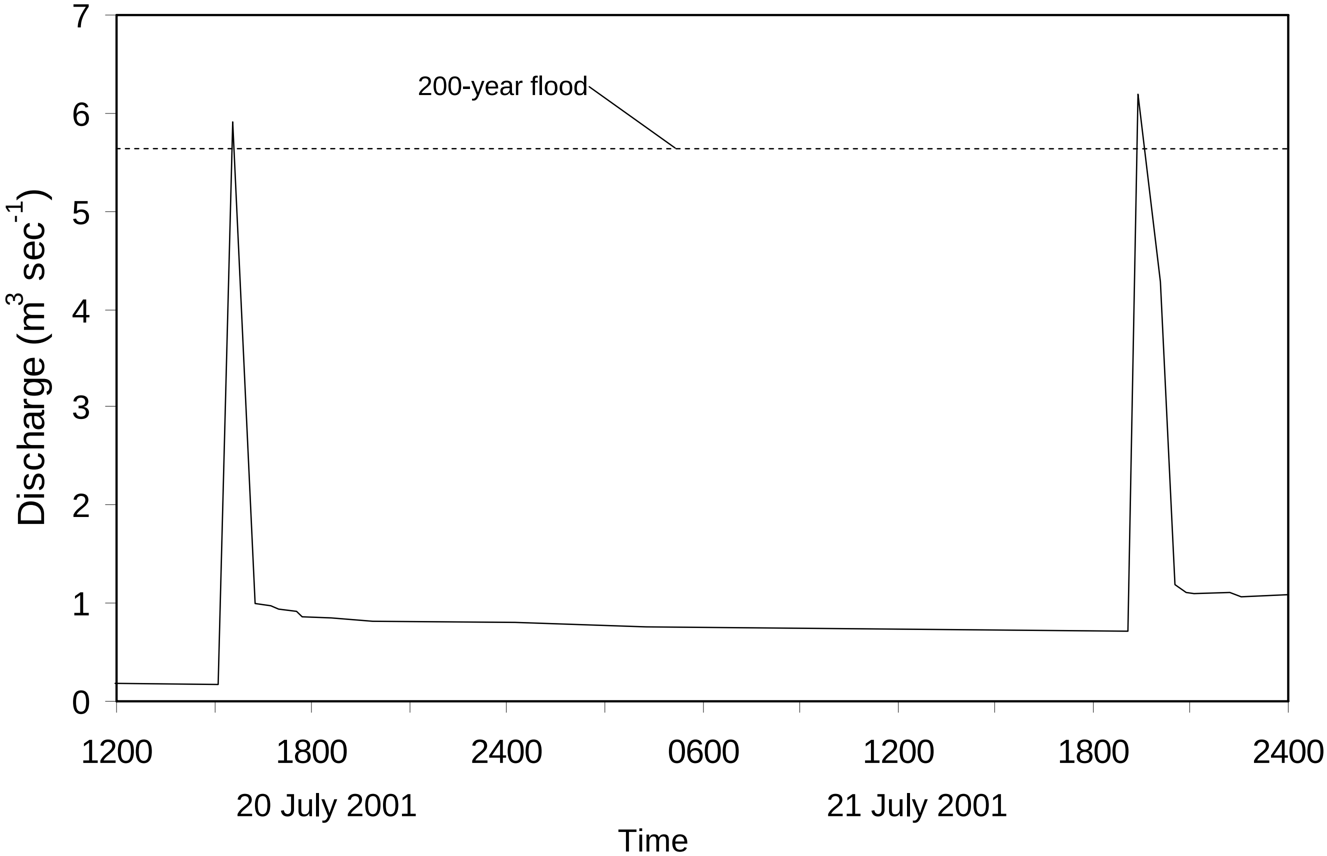

Figure 13. Storm hydrographs from Laird Creek, near Sula, Montana. Adapted from data from the U.S. Geological Survey.

Figure 14. Flood deposits from a Laird Creek tributary blocked the Laird Creek channel following storms of 20-21 July 2001, near Sula, Montana. Photo by U.S. Geological Survey.

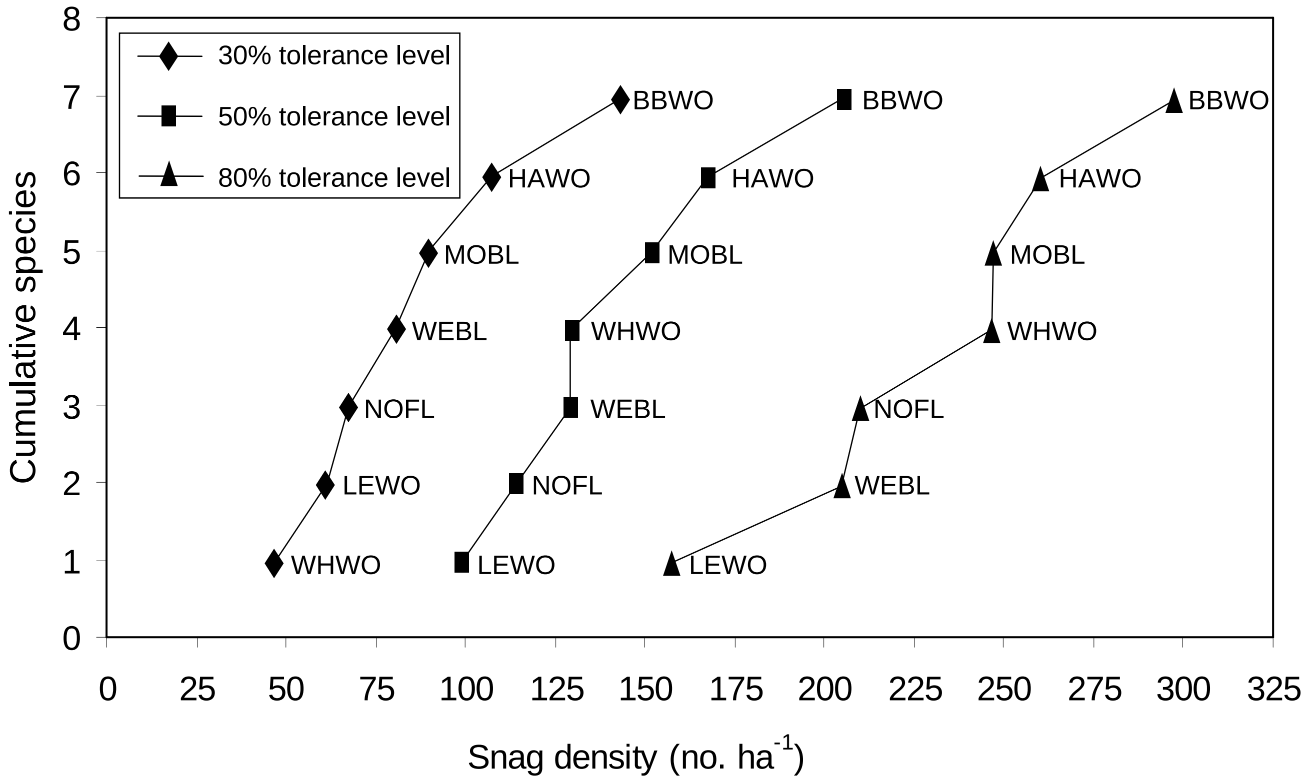

Figure 15. Cumulative bird species curves for density of snags ≥25 cm dbh at 30%, 50%, and 80% tolerance levels for the ponderosa pine/Douglas-fir wildlife habitat type and postfire structural condition class. Letter codes mark values for cavity-excavating bird species: WHWO = white-headed woodpecker, LEWO = Lewis' woodpecker, NOFL = northern flicker, WEBL = western bluebird, MOBL = mountain bluebird, HAWO = hairy woodpecker, BBWO = black-backed woodpecker. Derived with Program DecAID (Mellen et al. 2002; accessed online May 2006).

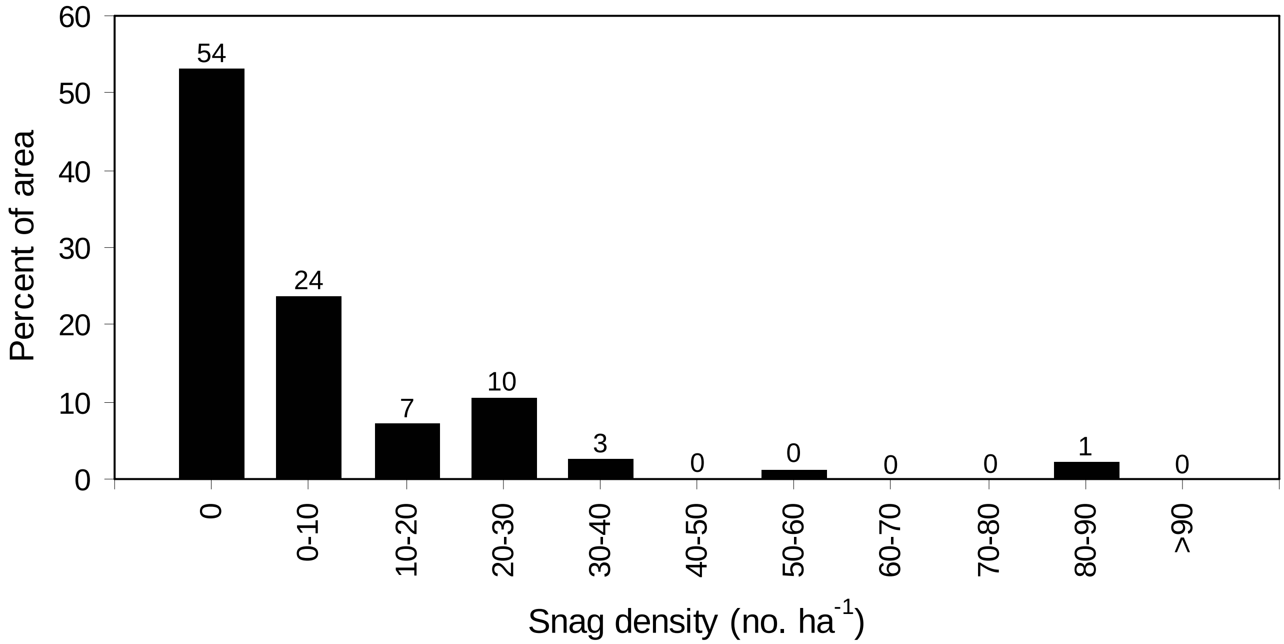

Figure 16. Distribution density classes for snags ≥25 cm dbh in unharvested and unburned ponderosa pine/Douglas-fir forest wildlife habitat type and large-tree structural class based on 73 inventory plots. Derived with Program DecAID (Mellen et al. 2002; accessed online May 2006).