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Effects of Timber Harvest Following Wildfire in Western North America

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Abstract

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Timber harvest following wildfire leads to different outcomes depending on the biophysical setting of the forest, pattern of burn severity, operational aspects of tree removal, and other management activities. Fire effects range from relatively minor, in which fire burns through the understory and may kill a few trees, to severe, in which fire kills most trees and removes much of the organic soil layer. Postfire logging adds to these effects by removing standing dead trees (snags) and disturbing the soil. The influence of postfire logging depends on the intensity of the fire, intensity of the logging operation, and management activities such as fuel treatments. In severely burned forest, timing of logging following fire (same season as fire vs. subsequent years) can influence the magnitude of effects on naturally regenerating trees, soils, and commercial wood value. Removal of snags reduces long-term fuel loads but generally results in increased amounts of fine fuels for the first few years after logging unless surface fuels are effectively treated. By reducing evapotranspiration, disturbing the soil organic horizon, and creating hydrophobic soils in some cases, fire can cause large increases in surface-water runoff, streamflow, and erosion. Through soil disturbance, especially the construction of roads, logging with ground-based equipment and cable yarding can exacerbate this effect, increasing erosion and altering hydrological function at the local scale. Effects on aquatic systems of removing trees are mostly negative, and logging and transportation systems that disturb the soil surface or accelerate road-related erosion can be particularly harmful unless disturbances are mitigated. Cavity-nesting birds, small mammals, and amphibians may be affected by harvest of standing dead and live trees, with negative effects on most species but positive or neutral effects on other species, depending on the intensity and extent of logging. Data gaps on postfire logging include the effects of various intensities of logging, patch size of harvest relative to fire size, and long-term (10+ years) biophysical changes. Uncertainty about the effects of postfire logging can be reduced by implementing management experiments to document long-term changes in natural resources at different spatial scales.

Keywords: Fire effects, postfire logging, postfire management, salvage logging, soil disturbance.

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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 social value of economic benefits versus potential changes in resource conditions has been debated in scientific and policy forums for two decades, but little has been resolved. Recently, several authors have raised concerns about the ecological effects of postfire logging (Beschta et al. 2004, Donato et al. 2006, Lindenmayer and Noss 2006, Noss and Lindenmayer 2006), and McIver and Starr (2001) and Lindenmayer et al. (2008) have synthesized the scientific literature on postfire logging effects. Here, we expand on this literature to provide a broad **scientific basis for decisionmaking about timber harvest as a component of postfire management**.

In this paper, we synthesize scientific findings on the effects of logging 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 decisionmaking. 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 logging in western North America (table 1) are disparate in geographic setting, study design, sampling, and analytical approach (McIver and Starr 2000, 2001), but do provide some basis for evaluation. In addition, the effects of wildland fire on forest ecosystems in western North America have been described in considerable detail in the scientific literature (e.g., Schmoldt et al. 1999), and the effects of logging on forest ecosystems have been well described at spatial scales from stands to landscapes (e.g., Hunter 1999). We draw on all these sources of literature to consider scientific issues relevant to timber harvest in postfire forest landscapes in western North America. We synthesize information relevant to inferences about postfire logging, and confine our analysis to the effects of large wildfires (typically >1000 ha) and subsequent harvest in the postfire environment.

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 (1) dry forests dominated by ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) and Douglas-fir (*Pseudotsuga*

Table 1—Summary of scientific literature on the effects of postfire timber harvest in western North America[a]

Citation	Forest type (location[b])	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 and 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 and Haire (1990)	Conifer forest (Wyoming)	Soil erosion, water runoff, organic matter and litter
Grifantini et al. (1992)	Douglas-fir–hardwood (California)	Shrub and forb cover, plant diversity
Stuart et al. (1993)	Douglas-fir–hardwood (California)	Vegetation cover (shrubs, forbs, ferns, grasses, hardwoods, conifers)
Chou et al. (1994a,1994b)	Montane conifer (California)	Soil erosion
Caton (1996)	Mixed conifer (Montana)	Cavity-nesting birds
Hitchcox (1996)	Mixed conifer (Montana)	Cavity-nesting birds
Hejl and McFadzen (1998)	Mixed conifer (Idaho, Montana)	Cavity-nesting birds
Saab and Dudley (1998)	Ponderosa pine, Douglas-fir (Idaho)	Cavity-nesting birds
Sexton (1998)	Ponderosa pine (Oregon)	Vegetation biomass and diversity, tree and shrub growth and diversity, soil moisture
Haggard and Gaines (2001)	Douglas-fir, ponderosa pine	Cavity-nesting birds
Khetmalas et al. (2002)	Subalpine fir (British Columbia)	Ectomycorrhizae and bacteria in soil
McIver (2004)	Ponderosa pine (Oregon)	Soil disturbance and erosion
Hanson and Stuart (2005)	Douglas-fir–hardwood (California)	Vegetation composition and structure
Donato et al. (2006)	Douglas-fir (Oregon)	Tree regeneration, woody debris
McIver and McNeil (2006)	Ponderosa pine (Oregon)	Soil disturbance and erosion
Macdonald (2007)	Aspen–mixed conifer (Alberta)	Vegetation composition
McIver and Ottmar (2007)	Ponderosa pine (Oregon)	Tree regeneration, woody debris, fuels

[a] From McIver and Starr (2001) and other recent sources.

[b] All locations are in the United States except for Khetmalas et al. (2002) and Macdonald (2007).

menziesii (Mirb.) Franco), (2) moist forests dominated by Douglas-fir, grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), western redcedar (*Thuja plicata* Donn ex D. Don), and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), (3) cold forests dominated by Engelmann spruce (*Picea engelmannii* Parry ex. Engelm.), fir species (*Abies* spp.), and lodgepole pine (*Pinus contorta* var. *latifolia* var. *murrayana* (Grev. & Balf.) Engelm.), and (4) woodlands such as pinyon pine-juniper (*Pinus* spp./*Juniperus* spp.) in the American Southwest.

When these forests burn, the primary effects are mortality of trees and other plants, emissions of carbon dioxide and other greenhouse gases, altered soil organic layers including complete removal in some cases, and mineral soil heating that can cause chemical transformations such as release of soluble nutrients. In many dry forests, fire historically burned through the understory consuming plants, killing seedlings, saplings, and some trees, and creating patches of bare soil. In many of these forests, dense stands of ponderosa pine and/or Douglas-fir and understory species such as grand fir have developed following grazing, removal of larger pines, and/or fire exclusion (Agee 1993, Taylor and Skinner 1998). 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 cold and moist forests, crown fire remains the norm, as it was historically, killing large patches of vegetation and altering soil properties over large areas. Forests intermediate between dry and cool/moist often burn with a mix of surface and crown fire.

The ecological effects of postfire logging are influenced by various combinations and intensities of the fire itself and management activities that affect (1) ground disturbance by equipment and road use, (2) number of living and dead trees and their spatial pattern following harvest, (3) postharvest fuel treatment, and (4) in some cases, grass seeding and placement of various structures and materials to mitigate the effects of fire and logging. Ground disturbance depends on the yarding system used for logging, with tractor logging generally causing the most soil disturbance, followed by cable yarding (which fully or partially suspends logs above the forest floor during extraction), followed by helicopter logging (Dykstra 1976, McIver and Starr 2001, Rice et al. 1972). The number and arrangement of residual trees, slash treatment, and postlogging treatments are determined during planning of postfire logging operations.

Effects of postfire logging on natural resources may depend on the scale at which the effects are measured.

Another factor influencing the effects of postfire logging is the spatial scale of both the fire and the logging operation (Peterson and Parker 1998). Fires may be large or small, as may be the area affected by logging. Measuring the combined effects of fire and management activities on soils and water is difficult because (1) erosion and sedimentation are in some cases high at the scale of forest stands but undetectable at the watershed scale, and (2) it is difficult to distinguish fire effects from management effects (Chou et al. 1994a). Similarly, deleterious effects on wildlife species may occur at the stand scale, but effects on viability in the larger landscape may be minimal. As a result, effects of postfire logging on natural resources may depend on the scale at which the effects are measured. Unfortunately, we are aware of no research that allows us to adequately quantify these scale effects or to address comprehensively the range of effects that are produced by all combinations of fire and logging intensity.

We therefore focus most of our discussion on the effects of relatively intense logging on severely burned landscapes. We generally do not address partial harvest of burned forest (with the exception of the wildlife section), and we only briefly consider the effects of postfire or postlogging mitigation. Many possible outcomes exist between intense logging/severe burns and no logging/light burns, and future studies will hopefully improve our understanding of these intermediate outcomes. We refer throughout the paper to a conceptual model of forest growth and succession as a context for temporal patterns of overstory development, understory development, standing dead trees (snags), and woody debris (≈ surface fuels) following postfire logging (fig. 1). This model describes ecological trends for general postfire management scenarios, and actual responses and trends are likely to differ among different forests, fires, and management situations.

Effects of Postfire Timber Harvest on Vegetation

An important effect of fire on vegetation is to kill plants, which, in the case of trees, generally occurs within 2 years following wildfire (Peterson and Arbaugh 1986). In addition to this primary mortality, fire can induce bark beetle-caused mortality in residual and adjacent green trees, which may last for several years following the fire (Amman and Ryan 1991, Edmonds et al. 2005, Furniss and Carolin 1977). When a tree is killed, dead needles not consumed by fire fall to the forest floor within a year or two, followed by branches and eventually boles. Snags provide important habitat features and long-term delivery of carbon (C) and nutrients to the forest floor when they fall (Harmon et al. 1986), which may take several decades.

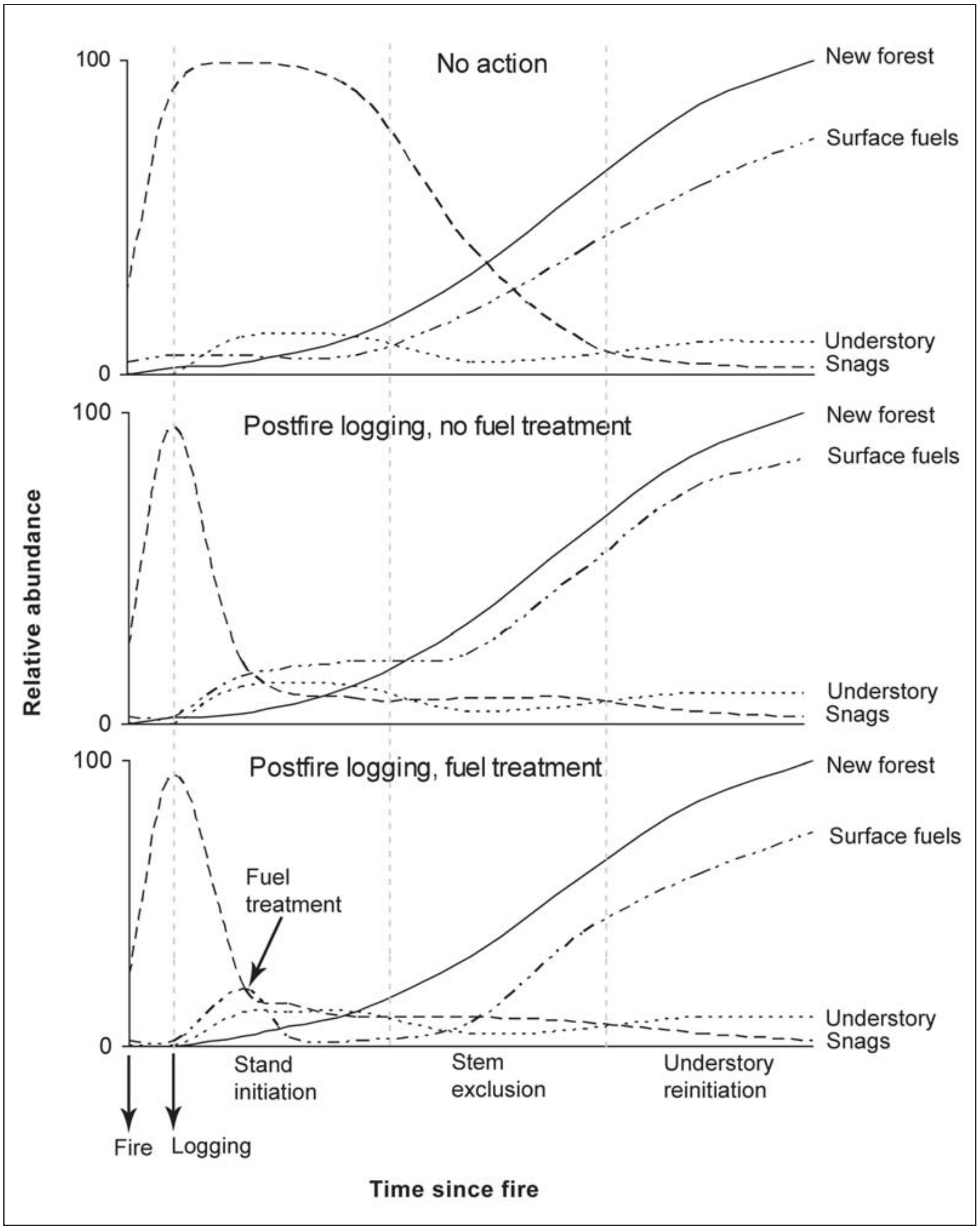

Figure 1—Conceptual diagram of temporal progression in biomass accumulation following wildfire with various management scenarios. General stand structural stages are indicated on the x-axis. **No management action** (upper diagram): no logging or surface fuel treatment; new forest grows through natural regeneration or planting; surface fuel accumulates primarily as a result of falling snags and dead understory. **Postfire logging, no fuel treatment** (middle diagram): most snags of commercial value removed, leaving unmerchantable trees in the stand; new forest grows through natural regeneration or planting; slash created by the logging operation not removed or treated. **Postfire logging, fuel treatment** (lower diagram): most snags of commercial value removed, leaving unmerchantable trees in the stand; new forest regrows through natural regeneration or planting; slash created by the logging operation is removed from the site. Actual biomass trajectories will differ considerably, depending on vegetation characteristics and management actions.

Trees killed or badly damaged by fire or insects begin to deteriorate immediately, and commercial value declines as wood decays or is blemished owing to insects, stain fungi, and decay fungi (Lowell et al. 1992). Insect and woodpecker activity provides a mechanism for introducing fungal agents into sapwood (Farris et al. 2004). Once initiated, sapwood decay progresses quickly, and sapwood of some species may be completely stained within a few months. Insects and staining fungi generally reduce product grade (and value) but seldom render wood unusable. Decay fungi also infect sapwood within the first year, and sapwood usually deteriorates beyond commercial use by the second or third year (Kimmey 1955).

By the second year, the heartwood is colonized by decay fungi, and heart rot begins to spread. Decay fungi affect wood properties, especially strength, thus reducing volume of wood that can be converted into structural products. 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. Weather deterioration causes longitudinal splits in wood by differential shrinkage where bark is missing or thin, or on the ends of logs after felling. This damage is usually minor in large standing dead trees, but smaller trees or species with thin bark may be affected more extensively, reducing the grade of smaller logs (Lowell and Cahill 1996). Progression of decay is similar in most conifer species, although decay rate differs considerably: ponderosa pine, grand fir (fast) > Douglas-fir > subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.), Engelmann spruce, western larch (*Larix occidentalis* Nutt.), and lodgepole pine (slow) (Hadfield and Magelssen 2006). Rate of volume loss is considerably lower for large logs than for small logs (fig. 2).

Postfire logging modifies snag dynamics by lowering the density and average size of dead trees (size distribution differs from site to site) and by altering site conditions such as soil bulk density and windflow. Following fires in the Blue Mountains of Oregon, snag biomass on harvested areas was generally <50 percent of that on unharvested sites (McIver and Ottmar 2007). Following fires in Idaho, the predicted half-life of ponderosa pine snags was 7 to 8 years (harvested) and 9 to 10 years (unharvested), and predicted half-life of Douglas-fir snags was 12 to 13 years (harvested) and 15 to 16 years (unharvested) (Russell et al. 2006). In this case, postfire logging reduced the persistence of snags, at least partially because the remaining snags were smaller and fell to the ground sooner than large snags.

Initial responses of the plant community to wildfire depend on prefire vegetation, seed banks, and the ability of plants to colonize the postburn environment. After severe fires, **invaders** (highly dispersive, pioneering fugitives), **endurers**

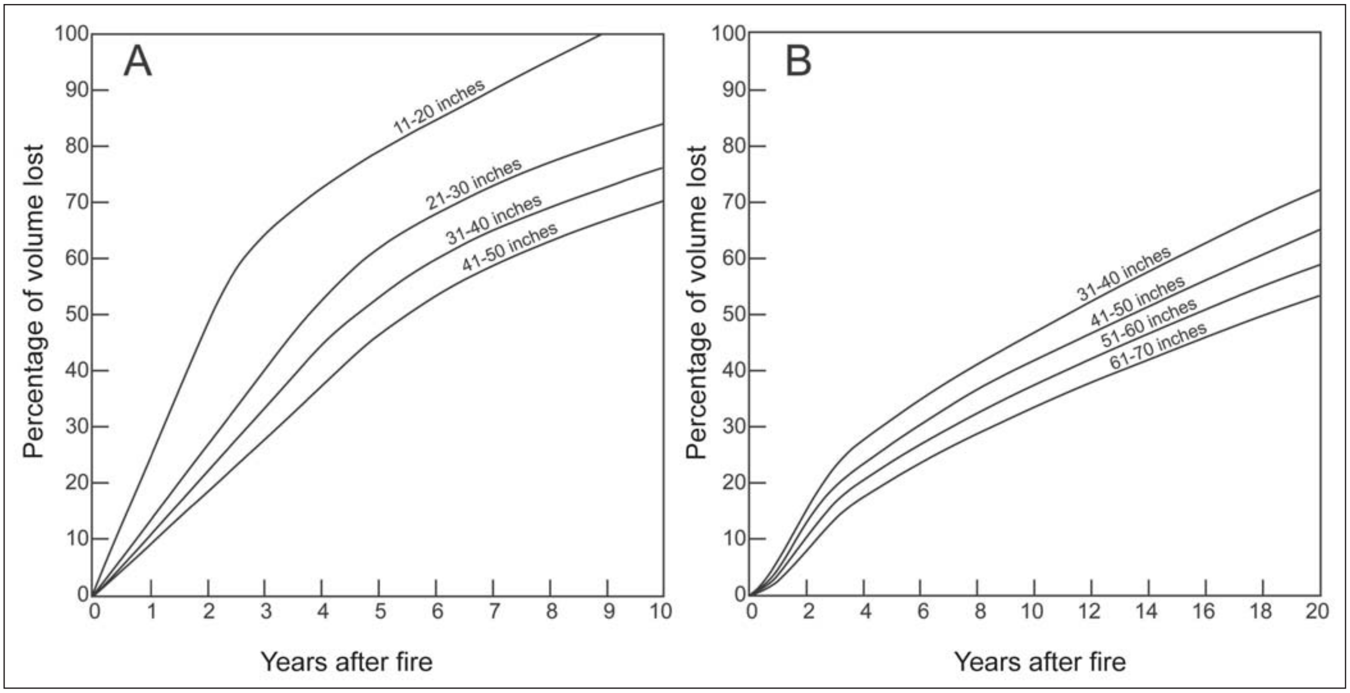

Figure 2—Average rate of loss from general deterioration of the original wood volume in Douglas-fir trees of several diameter at breast height classes for (**A**) second-growth wood (60 to 200 years old) and (**B**) old-growth wood (200 to 400 years). Note that the horizontal axis for A covers only half 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).

(sprouting from the crown, roots, and stem base), and **evaders** (with adaptations such as serotinous cones that open following fire) are likely to be the dominant species (Rowe 1981). **Resisters** have adaptations (e.g., thick bark and high, open crowns) that reduce fire injury during fires of low to moderate intensity. **Avoiders** (species with no fire adaptations) may be locally extirpated from the site for undetermined periods. Vegetation development following fire depends on the prevalence in prefire vegetation of species with each of these strategies, site conditions following fire, postfire management actions, and outside seed sources.

Many historically low-severity fire regimes in western dry forests were dominated by **resisters** like ponderosa pine and Douglas-fir. In mixed-severity fire regimes, **resister** and **evader** species (e.g., knobcone pine, *Pinus attenuata* Lemm.) are common, and in 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 **evader** species may be able to dominate early successional vegetation. Where severe fire kills all the trees in low-severity fire regimes, **resister**-type conifers behave like **avoiders**; regeneration (without replanting) may take a long time, because seed must be blown in from adjacent unburned forest.

The effects of postfire management on vegetation recovery are complex because of multiple successional pathways among forest species and structures following wildfires.

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. 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.

The effects of postfire management on vegetation recovery are complex because of multiple successional pathways among forest species and structures following wildfires (Frelich and Reich 1995). For example, Stickney (1986) recognized four distinct successional sequences after the Sundance Fire (Idaho) of 1967, with short- and long-duration herb- or shrub-dominated phases. Logging and mitigation for the effects of fire and logging create additional successional pathways for vegetation and fuels (Macdonald 2007). Nonetheless, some generalizations about vegetation development following postfire logging can be made.

An immediate effect of postfire logging is soil disturbance. In unburned forest, soil-disturbing activities can, in some cases, enhance establishment of species that require mineral soil (Sessions et al. 2004). However, in the postfire environment, logging is more likely to reduce resisters and evaders through direct mortality, if it occurs **after** significant establishment has occurred (Donato et al. 2006, McIver and Starr 2001, Roy 1956). Alternatively, logging may reduce serotinous evaders if it results in the removal of cones prior to seed dispersal (Greene et al. 2006). The functional effect of logging depends on subsequent establishment and mortality of tree seedlings, and on timing of logging (same season as fire vs. subsequent years; winter vs. other seasons); in many cases tree planting (and associated site preparation) replaces or supplements natural regeneration (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 burned/logged sites than on burned/unlogged sites (2 years following treatments). In contrast, hardwood cover was higher and shrub cover lower on burned/logged sites than on burned/unlogged sites 12 years following treatments. Douglas-fir regeneration (which included planting) was inhibited in both cases, but by different competing understory species. In boreal mixed-wood forest in Alberta, Macdonald (2007) found that single-tree and patch-retention harvest following wildfire had no significant effects on plant species richness and between-habitat (**beta**) diversity. Several environmental factors related to plant composition reflected variation in burn severity rather than logging, and they concluded that the legacy of prefire plant species composition combined with burn severity had a greater influence on postfire plant communities than the effects of logging.

In addition to the impacts of soil disturbance, the removal of trees themselves can affect the postfire plant community (e.g., Maser and Trappe 1984). Trees and large woody debris provide perches for seed-dispersing birds, and their removal may change the composition of the seed rain and therefore the plant community. Woody debris also provides protected "safe sites" for germination and establishment of some species, especially in the postfire environment, and their eventual decay can facilitate recruitment of understory species long after the initial disturbance. When snags fall over, they create tip-up mounds, an important source of seedbed diversity that can affect understory species composition.

Because postfire logging occurs at various locations and times and applies a variety of standards, effects on fuels and subsequent fire behavior may be specific to local conditions.

Effects of Postfire Timber Harvest on Fuels

Fuels and potential fire behavior in forests burned by wildfire differ over time, with or without logging (e.g., Graham et al. 2004) (fig. 1). Surface fuel dynamics following wildfires are a function of (1) prefire live and dead biomass; (2) tree species, which have different rates of litterfall, tree fall, and decay; (3) time, because standing fuels decay and become surface fuels; and (4) events following fire such as logging, windstorms, and delayed mortality owing to insects (Agee 1993). Fuel mass, fuelbed depth, and moisture content of available fuels contribute to the postfire and postharvest fuel complex. Because postfire logging occurs at various locations and times and applies a variety of standards, effects on fuels and subsequent fire behavior 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. These actions, in addition to tree species, affect the amount and timing of fuels reaching the ground, as well as the successional trajectory (Johnson et al. 2007).

Larger fuels (>7.6 cm diameter) follow a predictable pattern after crown fire (Agee 2002b) (fig. 1). Following initial mortality, snag biomass decreases over time (Agee and Huff 1987, Spies et al. 1988), and smaller snags usually fall first (Everett et al. 1999, Lehmkuhl et al. 2003). Biomass from fallen trees correspondingly increases for decades, especially in dry and cold environments, but may decline as the original snags that fell and created the woody debris begin to decay. Tree mortality from self-thinning in the new forest increases the amount of smaller, more easily decayed material (Harmon et al. 1986).

Postfire logging, without subsequent fuel treatment, has two effects on fuels: (1) the deposition of fine fuels (<7.6 cm diameter) on the forest floor, resulting in an increase in fire hazard in the short term, and (2) the removal of snags, resulting in a reduction of large fuels in the long term (figs. 1 and 3). Using empirical data for three stand treatments in the Blue Mountains (Oregon), McIver and Ottmar (2007) projected fine fuel loading over 30 years (fig. 4). Whole-tree yarding increased fine fuel loading above that in unlogged stands, an increase that persisted over 30 years, although levels in all treatments were similar after 20 years. Using empirical data for northern California forests, Weatherspoon and Skinner (1995) found that when wildfire in natural stands spreads to adjacent plantations, fire intensity and damage to the overstory are much lower in plantations where slash has been removed following logging.

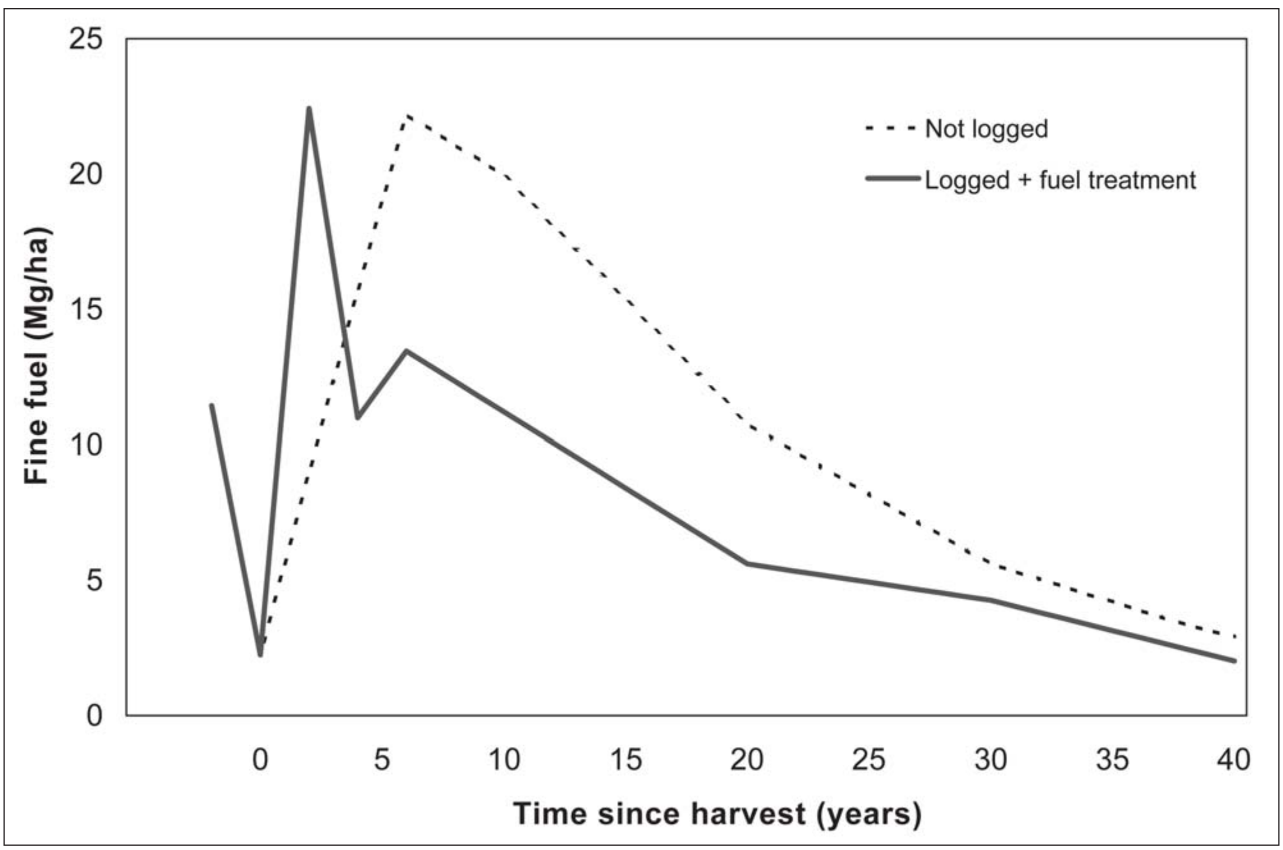

Figure 3—Patterns of fine fuel (<7.6 cm) after stand-replacing wildfire simulated by the Fire and Fuels Extension of the Forest Vegetation Simulator (Reinhardt and Crookston 2003) for a dry forest stand on the Bitterroot National Forest (Montana). The harvest treatment here consisted of removing snags >30 cm d.b.h. and <15 cm d.b.h., with fuels treated by slashing, piling, and burning. From Brown et al. (2003).

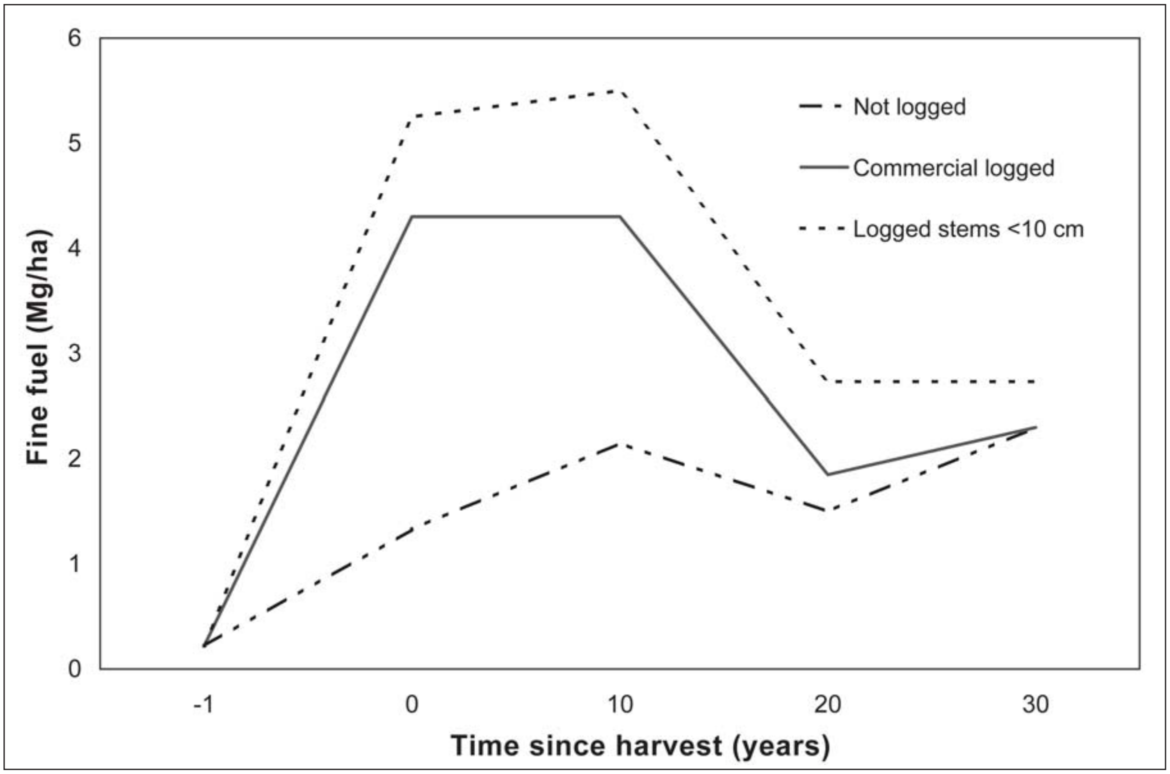

Figure 4—Simulated fine fuel load (<7.6 cm) in a dry forest in the Blue Mountains, Oregon. “Commercial” harvest was defined as removal of most merchantable trees, while “10 cm” harvest was defined as removing most trees <10 cm d.b.h. No slash fuel treatment was applied to either harvest option, but whole trees (including tops and branches) were cable-winched to landings. Scale of x-axis has been altered for display purposes. From McIver and Ottmar (2007).

In dry forests of interior western North America, large woody debris was limited historically by frequent fire that consumed logs (Agee 2003, Skinner 2002). Following a stand-replacement wildfire in a contemporary ponderosa pine forest, large woody debris of 10 to 100 Mg/ha may be present (Agee 2002b, McIver and Ottmar 2007, Passovoy and Fulé 2006) compared to historical woody debris biomass of 5 Mg/ha for dry ponderosa pine forest (Agee 2002b). Even higher woody debris has been measured in Douglas-fir forest with a mixed-severity fire regime (Wright 1998). Without treatment or removal, such high fuel loads may complicate the reintroduction of low-severity fire to the recovering forest.

The forest developing after wildfire or postfire logging may, 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 creates torching potential (cf. Scott and Reinhardt 2003). If the stand is dense (e.g., 10-cm d.b.h. trees at a density of >1200 per ha), 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. For example, in the Olympic Mountains (Washington), fine fuel mass following fire at a productive site (Agee and Huff 1987) was higher than short-term fine fuel mass following fire on drier sites (table 2). 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 post-fire was twice that of unburned forest primarily owing to branch fall caused by a windstorm during the first postfire winter.

Some historical evidence suggests that fires block the spread of subsequent fires in low-severity and mixed-severity fire regimes. In ponderosa pine forest, large historical low-severity fires were generally followed by fires of smaller extent (Everett et al. 2000), implying a fuel-limiting 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. In the Entiat River watershed in the Cascade Range (Washington), a series of large fires has burned much of the dry forest in the lower watershed since 1970 (fig. 5), yet the pattern of fire spread does

Table 2—Fine fuel mass (<7.6 cm diameter) differs by site, over time, and by treatment[a]

		Treatment		
Location (reference)	**Time since fire**	**No logging**	**Logging**	**Unburned forest**
	Years	– – – – – – – – – – *Mg/ha*	– – – – – – – – – –	
Olympic Mountains, NW Washington (Agee and Huff 1987)	1	13.5		6.1
	3	9.2		6.1
	19	5.3		6.1
	110	5.2		6.1
Siskiyou Mountains, SW Oregon (Raymond and Peterson 2005)	1	1.2		6.1
	1	3.6		29.7[b]
Siskiyou Mountains, SW Oregon (Donato et al. 2006)	3	1.0	6.3	5.9
Blue Mountains, NE Oregon (McIver and Ottmar 2007)	3	1.5	4.3 / 5.4[c]	

[a] Each line indicates a separate site within the geographic location.

[b] Postthinning windthrow event.

[c] Two harvest options (see text).

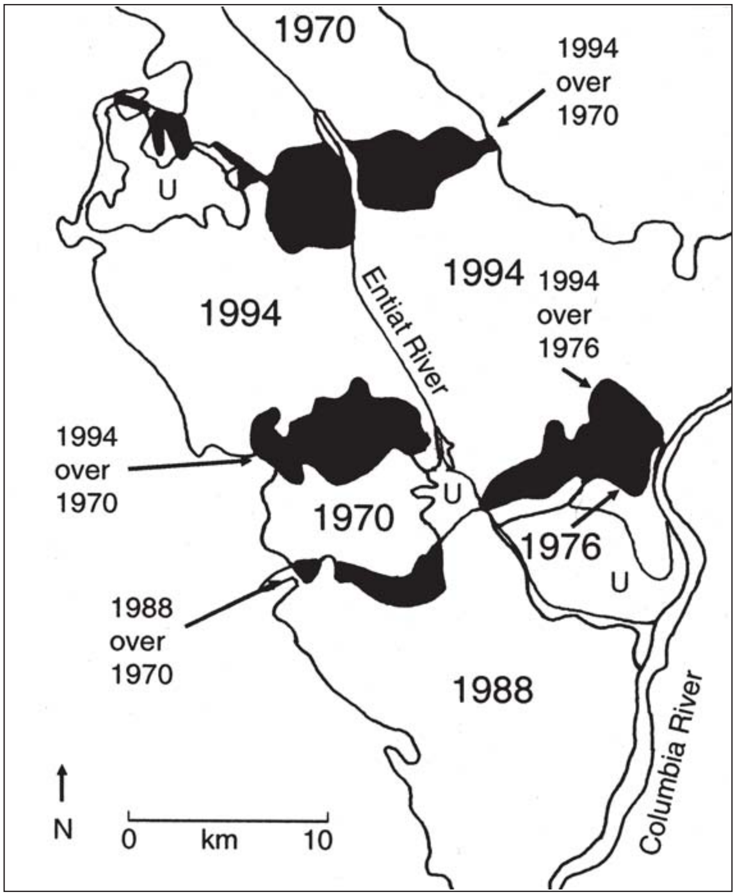

Figure 5—Map of the lower Entiat River, eastern Cascade Range (Washington), showing dates of wildfires and reburns. Small unburned patches within the matrix of burned areas are identified with "U." Black areas are reburns.

not support reburn potential in previously burned areas. It appears that previously unburned areas have the highest potential for burning, and that fires have stopped at or near old burn boundaries.

Although recent crown fire often reduces the probability of burning in the next fire, it does not guarantee protection. For example, an analysis of fire hazard conducted after the Cerro Grande Fire (New Mexico) of 2000 concluded that much of the area burned would be at lower risk as a result of the fire, but that some areas would be exposed to higher fire hazard, at least temporarily, as a result of the buildup of fallen trees (Greenlee and Greenlee 2002). In addition, large wildfires (>4000 ha) were found to occur at intervals of 15 to 20 years across any given location in pine-dominated forests of northeastern California (Norman and Taylor 2003). The Biscuit Fire (Oregon) of 2002 burned through the Silver Fire of 1987 (mostly wilderness without postfire logging), which suggests that fuels had accumulated sufficiently in 15 years to reburn.

The Tillamook fires of 1933-1951 in the Coast Range of Oregon are another example of "reburn potential" but in a high-precipitation environment. The first fire in 1933, started by logging 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 percent of the area a reburn of the 1933 fire. In 1945, 73 000 ha burned with 50 percent of the area a reburn of the 1933 and 1939 fires. In 1951, 13 000 ha burned within the area burned in 1933 and 1939 (Oregon Department of Forestry 1983). 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. Slash from the harvest operations likely helped carry some of the subsequent fires; other sources of fire spread were snags left from previous wildfires (Oregon Department of Forestry 1983) and a vigorous understory of bracken fern (*Pteridium aquilinum* (L.) Kuhn) (Isaac 1940). The Tillamook cycle of reburns was broken after the 1951 fire–more than 300 km of snag-free fuelbreaks were constructed, some snags were harvested and others were dropped on site, road access was improved, and fire prevention was promoted.

Understory response by forbs and shrubs is a less appreciated factor in postfire fuel dynamics. If fire creates a favorable environment for annual grasses, substantial fine fuel can be present within a year of the fire, and the potential for subsequent fire exists (Weatherspoon and Skinner 1995, Zedler et al. 1983). If response is by shrubs and perennial grasses, moisture content generally remains high enough during the fire season that potential fire behavior may be reduced (Agee et al. 2002, Raymond and Peterson 2005).

These findings suggest several conclusions about the effects of fire and subsequent logging on fuels. First, past fire generally reduces the probability of a site burning again for a period of years to decades but can result, at least temporarily, in fuel hazards above prefire conditions. Second, postfire logging creates activity fuels that increase fire hazard, unless those fuels are effectively reduced or removed from the site. Finally, the potential for reburn may remain high, regardless of treatment of woody fuels, wherever regrowth of highly flammable fine fuels, such as bracken fern and annual grass, is rapid.

Effects of Postfire Timber Harvest on Soils and Hydrology

Forests accumulate organic matter in the absence of major disturbance, and western forests often exceed 500 Mg/ha in standing biomass, of which roughly half is organic C. At maturity, 50 to 75 percent of aboveground C is in the boles of trees, followed by the forest floor and tree crowns (table 3). Soil C concentrations are generally high near the surface and decline with depth. Soil organic matter from litter decay and root turnover is the main source of soil nitrogen (N), which is concentrated near the soil surface and declines with depth, and N concentrations in the forest floor of western forests commonly are much higher than in the mineral soil (Cole and Gessel 1992).

Intense wildfire typically oxidizes or volatilizes soil C and N in the forest floor and surface layers of mineral soil (Dyrness et al. 1989), with heat duration and depth of penetration dependent on fuel characteristics, weather conditions, and fire behavior (Neary et al. 1999). Heating effects of fire on soil depend on temperature of the fire, duration of heating, and soil moisture, with effects lasting a few years to a few decades (Robichaud et al. 2000). Fire typically results in higher short-term nutrient availability to plants (Haase and Sackett 1998), lower total nutrient content in the soil, altered soil biota, and altered physical condition (Neary et al. 1999).

Organic matter holds mineral soil in place, and crown fires typically eliminate surface organic matter (Graham et al. 2004, Jain et al. 2004, Peterson et al. 2005), leading to potential soil erosion. Erosive losses of surface soil from overland flow can range between 1 and 85 Mg/ha or more in the first year, depending on fire severity, slope, and precipitation (Baird et al. 1999). Depending on wildfire intensity, much of the organic matter remaining in a forest stand may exist in standing live and dead trees, and biomass removed during logging therefore alters the C cycle. The potential for wood products (versus dead trees left to decay on site) to sequester C (Johnson et al. 2005) depends on the life cycle of the products and energy used for transportation and manufacturing.

Nitrogen loss is a linear function of the amount of material consumed by fire (McColl and Powers 1984). On drier sites in the Washington Cascades, surface erosion accounted for N losses of up to 22 kg/ha in the first year after fire, although much of it was redistributed downslope in the burned area (Baird et al. 1999). The greatest effect on soil fertility and N loss is caused by the loss of the forest floor, the largest reservoir of aboveground N and other nutrients (DeLuca and Zouhar 2000, Powers et al. 2005).

Table 3—Biomass and nitrogen contained in aboveground components of mature forests typical of western North America [a]

		Biomass[b]			Nitrogen[c]		
Location	**Forest type**	**Forest floor**	**Crown**	**Bole**	**Forest floor**	**Crown**	**Bole**
		Mg/ha (percentage of aboveground total)			*kg/ha (percentage of aboveground total)*		
British Columbia	Subboreal white spruce (*Picea glauca* (Moench) Voss, *Pinus contorta*)	65 (29)	32 (14)	126 (57)	815 (76)	58 (5)	195 (18)
Colorado	Subalpine fir (*Abies lasiocarpa*)	103 (31)	54 (16)	173 (52)	598 (46)	302 (23)	394 (30)
Idaho	Mixed conifer (*Abies, Picea, Pinus*)	70 (27)	31 (12)	160 (61)	436 (52)	220 (26)	190 (22)
Washington	Douglas-fir (*Pseudotsuga menziesii*)	14 (8)	31 (17)	134 (75)	187 (37)	164 (32)	161 (31)
California	Mixed conifer (*Abies, Pinus*)	59 (11)	221 (42)	252 (47)	455 (43)	391 (37)	218 (20)
Arizona	Pine (*Pinus ponderosa*)	47 (24)	31 (16)	121 (61)	291 (48)	176 (29)	145 (24)

[a] Modified from Kimmins et al. 1985; Powers 2006.
[b] More than 50 percent of the biomass is in the boles.
[c] Approximately 40 to 75 percent of the nitrogen is in the forest floor.

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 2004, McIver and McNeil 2006, McIver and Starr 2001). Skidding logs across bare ground disturbs and compacts soil more than other methods (Beschta et al. 2004, Klock 1975), particularly when soil moisture is near the plastic limit. Depending on soil texture, compaction can be detrimental or favorable to plant growth (Froehlich and McNabb 1984). On clayey soils, compaction degrades soil quality by increasing strength and reducing aeration porosity, whereas on sandy soils, compaction may enhance plant growth by improving soil water availability (Gomez et. al 2002, Powers et al. 2005).

In either case, the immediate concern about soil compaction from postfire logging is that the size of surface soil pores can be reduced, and infiltration can be impeded, potentially causing increased surface erosion. This is especially a concern following fires in which surface litter has been consumed. The presence of even a thin litter layer can substantially reduce soil erosion (Powers 2002). A study conducted in Portugal suggests that when soils are bared by wildfire, slash produced during logging can be used to cover the soil surface and reduce erosion (Shakesby et al. 1996).

As noted above, different logging systems have different effects on soil compaction and fertility. Tractor logging and ground-based equipment on relatively level areas (<30 percent slope) cause the most soil compaction, although some effects can be mitigated by avoiding wet soils, logging over snow, and 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, which suspends logs above the ground, avoids most physical abrasion of the forest floor and mineral soil. Helicopter logging greatly reduces soil impacts by minimizing movement of logs along the ground.

Soil water repellency (hydrophobicity) occurs where hydrophobic substances produced by organic matter coat soil particles, thereby impeding water infiltration into the soil profile, creating 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). When soil heating from fire results in temperatures between 176 and 288 °C, transfer of hydrophobic substances into the mineral soil can be enhanced, resulting in increased water repellency (DeBano 1981), Fire-induced hydrophobicity is transient and often patchy. 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 percent, and lightly burned soils were repellent to soil moisture of 14 percent. However, repellency persisted on severely burned sites even at soil moisture contents of >26 percent. Poff (1989) observed in the Sierra Nevada that disturbance of the soil surface by logging can in some cases disrupt hydrophobic layers and reduce water repellency, but MacDonald (1989) cautioned that this potential benefit must be weighed against the detrimental effects of logging. In some cases, managers have subsoiled their lands along contours following postfire logging to mitigate the effects of compaction on fine-textured soils and to break up hydrophobic layers (Webster and Fredrickson 2005).

Removal of living forest overstory increases water yield, although increased water yield is not detectable unless 20 to 40 percent of watershed or forest basal area is removed.

Fire affects hydrology through the removal of aboveground canopy, removal of litter, and, in some cases, development of a hydrophobic soil layer. Direct measurements of the effects of fire on hydrology are rare, whereas the effects of vegetation removal are well studied and can be used cautiously to infer fire effects (Neary et al. 2005a, 2005b, 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).

Removal of living forest overstory increases water yield, although increased water yield is not detectable unless 20 to 40 percent 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 water yield is higher in wet years than dry years. Increased water yields in coastal forests of western North America are greater than those in dry, interior forests.

Water yield typically increases significantly in the first year following fire or logging, then decreases with time as vegetation reoccupies a watershed. This “hydrologic recovery” is related to site quality, which in turn is determined by temperature regime, moisture regime, and soil fertility. Recovery may take 60 to 80 years in high-elevation (ca. >1000 m), interior watersheds of western North America. It may take <25 years in coastal watersheds, although analysis at Alsea Experimental Watershed (Oregon) revealed that mechanisms of streamflow generation and routing had not recovered to predisturbance levels 28 years following clearcut logging (although water yield had recovered) (Stednick 1996).

Soil water storage, interception, and evapotranspiration are reduced when vegetation is killed and organic material on the soil surface is consumed by wildfire (DeBano et al. 1998, Neary et al. 2005b, 2005c). Changes in annual water yields are about the same as those caused by forest harvest, although perhaps greater because fire reduces absorptive soil organic material and kills some of the understory vegetation. Normal patterns of snow accumulation and melt may also be changed. Infiltration may be reduced for several reasons, which in turn may increase overland flow and alter subsurface flow. Changes in these processes result in increased peak flow, altered runoff timing, altered base flow, and probable changes in water quality (table 4). For example, following fires in the Bitterroot Mountains (Montana) in 2000, the 24.5-km^2 Laird Creek watershed, of which 30 percent burned with high severity (Nickless et al. 2002), experienced two flood events within 2 days that would have been considered 200- to 500-year floods based on unburned conditions (fig. 6) (Parrett et al. 2004), triggering significant debris flows (fig. 7).

Table 4—Summary of changes in hydrologic processes produced by wildland fires

Hydrologic process	Type of change	Specific effect
Interception	Reduced	Less moisture stored Higher runoff in small storms Higher water yield
Litter storage of water	Reduced	Less water stored Higher overland flow
Transpiration	Temporary elimination	Higher streamflow Higher soil moisture
Infiltration	Reduced	Higher overland flow Higher stormflow
Streamflow	Changed	Higher in most ecosystems Lower in snow-ominated systems Lower in fog-drip systems
Base flow	Changed	Lower (less infiltration) Higher (less evapotranspiration) Summer low flows (+ and -)
Stormflow	Increased	Higher volume Higher peak flows Less time to peak flow Higher flash flood frequency Higher flood levels Higher stream erosive power
Snow accumulation	Changed	Fires <4 ha, increased snowpack Fires >4 ha, decreased snowpack Higher snowmelt rate Higher evaporation and sublimation

Source: Neary et al. 2005a.

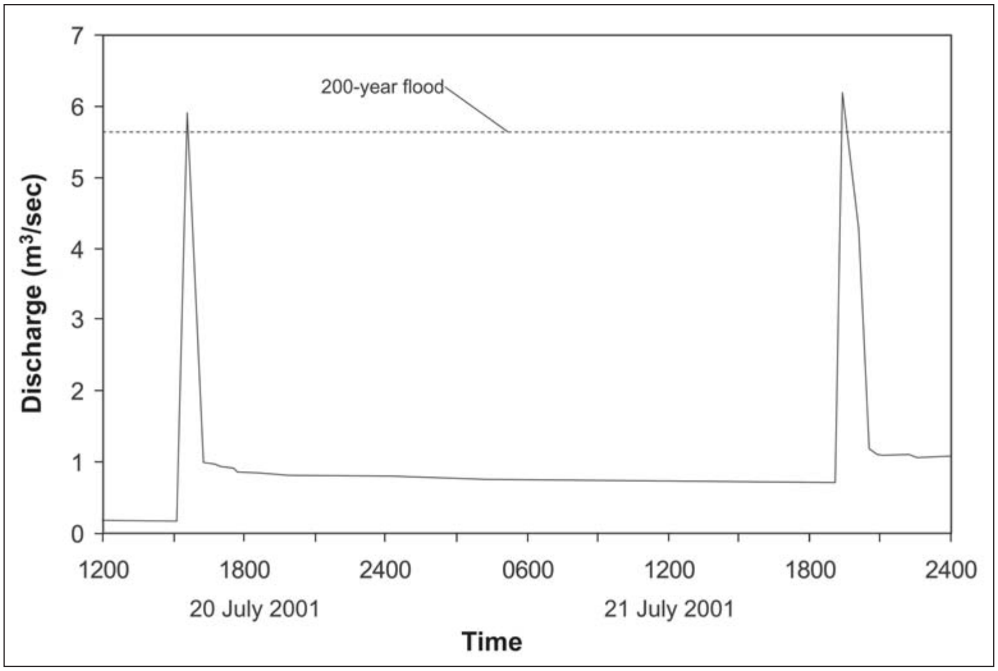

Figure 6—Storm hydrographs from Laird Creek, near Sula, Montana. Adapted from data from the U.S. Geological Survey.

U.S. Geological Survey

Figure 7—Flood deposits from a Laird Creek tributary blocked the Laird Creek channel following storms of 20–21 July 2001, near Sula, Montana.

Unfortunately, little research has been done that directly addresses the additional hydrologic impact of logging beyond the fire itself. Because postfire logging takes place in an environment in which the canopy and soil have already been modified, it is reasonable to conclude that logging will not add significantly to the altered hydrology. However, to the extent that logging results in soil compaction, it may exacerbate enhanced runoff, and soil disturbance is likely to add to the potential for debris movement.

Forest roads, whether built to facilitate postfire logging or not, can exacerbate hydrological effects by concentrating and channelizing surface and subsurface flow. 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 percent 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 cut banks and fill slopes have revegetated and stabilized. This may be especially important when roads are built to provide access for postfire logging, because landscape capacity to trap sediments above streams can be greatly reduced following wildfire.

Management of forest roads presents special challenges in the postfire environment. Streamflows typically increase for the first several years following vegetation removal by fire, and a hydrophobic soil layer, if present, can accelerate 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).

Effects of Postfire Timber Harvest on Riparian Systems and Aquatic Ecology

Riparian areas play a critical role in the postfire environment. They are often the least severely burned part of the landscape owing to topography and the cooler temperatures of canyon bottoms. Intact riparian buffers reduce sediment delivery to streams, maintain cooler stream temperatures through shading, and minimize

Protection of residual structures following fire in riparian areas (including smaller streams) is critical for minimizing deleterious effects.

changes in aquatic habitat for fish and macroinvertebrates (Davies and Nelson 1994), all of which are especially important following wildfire. Vegetation in riparian areas also mitigates the effects of denuded slopes on wildlife by providing habitat for small mammals (Cockle and Richardson 2003), birds (Pearson and Manuwal 2001), and amphibians (Pilliod et al. 2003).

A recent summary of the effects of fire and logging on riparian and aquatic systems (Reeves et al. 2006) concluded that protection of residual structures following fire in riparian areas (including smaller streams) is critical for minimizing deleterious effects. Few data exist on the effects of postfire logging on aquatic ecosystems, although effects can be inferred from the literature on riparian fire (Dwire and Kauffman 2003, Everett et al. 2003), fire effects, and logging effects. Information on the effects of fire on aquatic vertebrates has focused primarily on salmonid fishes (Dunham et al. 2003, Gresswell 1999, Rieman et al. 2003) and to a lesser extent on amphibians (Pilliod et al. 2003). Short-term (<3 years) effects on aquatic vertebrates are generally negative. Severe fires that burn through riparian areas can cause mortality or emigration of fish and other organisms owing to 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 soil infiltration, increased erosion, changes in timing and amount of runoff, elevated stream temperature, and altered channel morphology (Wondzell and King 2003).

Substantial information exists on the effects of logging on fish and amphibians (Corn and Bury 1989; deMaynadier and Hunter 1995; Hawkins et al. 1982, 1983; Murphy and Hall 1981; Murphy and Koski 1989; Murphy et al. 1981). Most of this literature documents adverse responses, although relationships of specific effects to harvest area and to proximity of logging to streams are poorly quantified. Removal of snags following fire may affect macroinvertebrate (Minshall 2003), fish, and amphibian populations, because large wood recruitment into streams following fire alters channel morphology (e.g., steps, pool habitats), sediment transport, and nutrient cycling (Gregory and Bisson 1996, May and Gresswell 2003). Long-term (>10 years) effects on aquatic systems and biota depend on the context and scale of disturbance (Reeves et al. 2006). Erosion following fires and logging can provide wood and coarse sediment that maintain productive habitats (Reeves et al. 1995), create heterogeneity in channel structure and complexity, and temporarily increase aquatic productivity through nutrient transfer (Minshall 2003). Benthic macroinvertebrate populations recover within a few years after fire (Minshall 2003), and fish communities in Idaho recovered within 10 years after severe fires (Rieman and Clayton 1997).

Postfire logging activities such as road building and log skidding can increase surface soil erosion, resulting in increased sedimentation of stream substrates (Church and Eaton 2001, Ketcheson and Megahan 1996, Lee et al. 1997, Rieman and Clayton 1997). 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, Church and Eaton 2001, Gillespie 2002, Minshall et al. 1997). Mass wasting and debris flows, often associated with structural failure of roads or removal of trees near drainage headwalls are the most damaging to stream habitat. If postfire management activities alter aquatic system recovery processes in terms of physical and biological landscape elements (e.g., wood in streams) and frequency and severity of natural disturbance, those activities can delay development of late-successional conditions (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).

Effects of Postfire Timber Harvest on Terrestrial Wildlife

The effects of postfire logging on terrestrial wildlife species depend on characteristics of the animal as well as characteristics of the environment. Most studies infer causes of observed postfire and postharvest patterns from life history traits and habitat relationships, with changes in wildlife abundance mediated by changes in vegetation composition and structure (Bunnell 1995, Pilliod et al. 2003, Saab and Powell 2005, Smith 2000). 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, Kotliar et al. 2002, Pilliod et al. 2006, Saab et al. 2005). Terrestrial species dependent on shrub and herb understories for food and cover (e.g., ungulates) generally benefit from increased diversity of understories following fire, although species associated with woody debris may decrease in the short term until new down wood is recruited (Lyon and Smith 2000, Pilliod et al. 2006). Depending on species tolerance to postfire conditions, recolonization of a burned forest may occur within days or years (Lehmkuhl 2005). If only low-quality habitat is available, an animal may remain in unburned forest patches or move elsewhere until conditions become favorable.

In general, fire has a positive or neutral effect on cavity nesting birds, whereas postfire logging has a negative effect on most of these species (table 5). There is evidence that Lewis' woodpecker (*Melanerpes lewis*) may benefit from limited postfire logging that accelerates the development of open stands (Haggard and

Table 5—Summary of responses of cavity-nesting bird species to fire, timber harvest, and postfire timber harvest, including supporting evidence[a]

Species	Fire	Timber harvest	Fire and timber harvest	Citation
Black-backed woodpecker (*Picoides arcticus*)	Positive (12)	Negative (10)	Negative (3)	Caton 1996, Hitchcox 1996, Hobson and Schieck 1999, Hutto 1995, Imbeau et al. 1999, Kotliar et al. 2002, Kreisel and Stein 1999, Saab and Dudley 1998, Saab et al. 2004, Stepnisky 2003
Downy woodpecker (*P. pubescens*)	Neutral (6), occasionally positive (2)	Neutral (11), occasionally negative (1)	Neutral (4), occasionally negative (1)	Caton 1996, Franzreb and Ohmart 1978, Greenberg et al. 1995, Hitchcox 1996, Hobson and Schieck 1999, Hutto 1995, Stepnisky 2003
Hairy woodpecker (*P. villosus*)	Positive (11), especially 1 to 2 years postfire (2)	Positive (9)	Negative (4)	Caton 1996, Franzreb and Ohmart 1978, Hitchcox 1996, Hobson and Schieck 1999, Hutto 1995, Johnson and Wauer 1996, Kotliar et al. 2002, Kreisel and Stein 1999, Saab and Dudley 1998, Saab et al. 2004, Stepnisky 2003
Three-toed woodpecker (*P. tridactylus*)	Positive (10), especially 1 to 2 years postfire (2)	Negative (9), occasionally neutral (1)	Negative (2)	Caton 1996, Franzreb and Ohmart 1978, Hitchcox 1996, Hobson and Schieck 1999, Hutto 1995, Imbeau et al. 1999, Johnson and Wauer 1996, Kotliar et al. 2002, Kreisel and Stein 1999, Stepnisky 2003
White-headed woodpecker (*P. albolarvactus*)	Negative (1)	Negative (1)	Negative (1)	Garrett et al. 1996
Lewis' woodpecker (*Melanerpes lewis*)	Positive (1)		Positive (2)	Hitchcox 1996, Johnson and Wauer 1996, Saab and Dudley 1998, Saab et al. 2004
Mountain bluebird (*Sialia currucoides*)	Positive (9)	Positive (7) or neutral (1)	Negative (4)	Caton 1996, Hitchcox 1996, Hobson and Schieck 1999, Hutto 1995, Johnson and Wauer 1996, Kotliar et al. 2002, Raphael and White 1984, Saab and Dudley 1998, Saab et al. 2004
Western bluebird (*S. mexicana*)	Neutral (5), occasionally positive or negative (1)	Neutral (9)		Hutto 1995, Johnson and Wauer 1996
Northern flicker (*Colaptes auratus*)	Neutral (7), occasionally positive (5) or negative (1)	Neutral (9), occasionally positive (1)	Neutral (1)	Franzreb and Ohmart 1978, Greenberg et al. 1995, Hobson and Schieck 1999, Horton and Mannan 1988, Hutto 1995, Kotliar et al. 2002

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 underrepresented in the literature. See Hutto (1995) and Kotliar et al. (2002) for additional references.

[a] Number of studies listed in parentheses.

Gaines 2001, Saab et al. 2002), although Hutto and Gallo (2006) found no significant effect. In contrast, black-backed woodpeckers (*Picoides arcticus*) and three-toed woodpeckers (*P. tridactylus*) are associated with dense stands of snags, and postfire logging likely would be detrimental to their occupation of the site (Hutto and Gallo 2006, 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 logging, may increase the abundance and diversity of cavity-nesting species over the short term (ca. 5 year) (Haggard and Gaines 2001, Saab and Dudley 1998).

Traditional snag retention guidelines are based on the notion that snag harvest that targets tree species and smaller trees 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), although they appear to have value for feeding of beetles and birds (Hutto 2006). Snags <25 cm d.b.h. are generally too small for nests of cavity-excavating birds (Rose et al. 2001). If snags are very dense, then removal of some snags may have little effect (Mellen et al. 2002), especially where the density of primary cavity-nesting birds is limited by territorial behavior (Bunnell 1995, Raphael and White 1984). However, that relationship would not be expected to hold for subsequent use by secondary cavity-nesting birds and mammals (Raphael and White 1984) or where nest density is limited by insect food sources during the first 5 years after fire when bark beetles and wood-boring beetles attract insectivorous birds (Hutto and Gallo 2006, Lehmkuhl et al. 2003). Snag density guidelines for individual species closely associated with postfire habitats could facilitate management of multiple species (Hutto 2006).

Postfire logging can alter the abundance of large woody debris (fig. 1), which provides important habitat for many wildlife species (Bull 2002; Bull et al. 1997; Bunnell et al. 2002; Maser et al. 1979; Pilliod et al. 2003, 2006). 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. Postfire logging 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. 1). Lack of large woody debris may alter species occurrence or abundance of some wildlife species in regenerating forests

Good postfire snag habitat occurs where (1) management actions promote tree species and large size classes favored for cavity excavation, (2) patchiness in stand densities provides postfire habitats for different species, and (3) defective trees that provide immediate postfire cavity excavation opportunities are created and retained.

following postfire logging, depending on stand conditions and number of snags removed, although it is difficult to quantify thresholds of down wood abundance relative to wildlife habitat quality. For example, 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) than with high levels of down wood (118 m^3/ha), although this pattern was not observed for yellow-pine chipmunk (*Tamias amoenus*) or deer mouse (*Peromyscus maniculatus*) (Smith and Maguire 2004).

Good postfire snag habitat occurs where (1) management actions promote tree species and large size classes favored for cavity excavation, (2) patchiness in stand densities provides postfire habitats for different species, and (3) defective trees that provide immediate postfire cavity excavation opportunities are created and retained. Attaining desired size, density, and distributions of snags and down wood is an important consideration for short-term postfire management (fig. 8). 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 (Everett et al. 1999, McIver and Ottmar 2007, Saab and Dudley 1998).

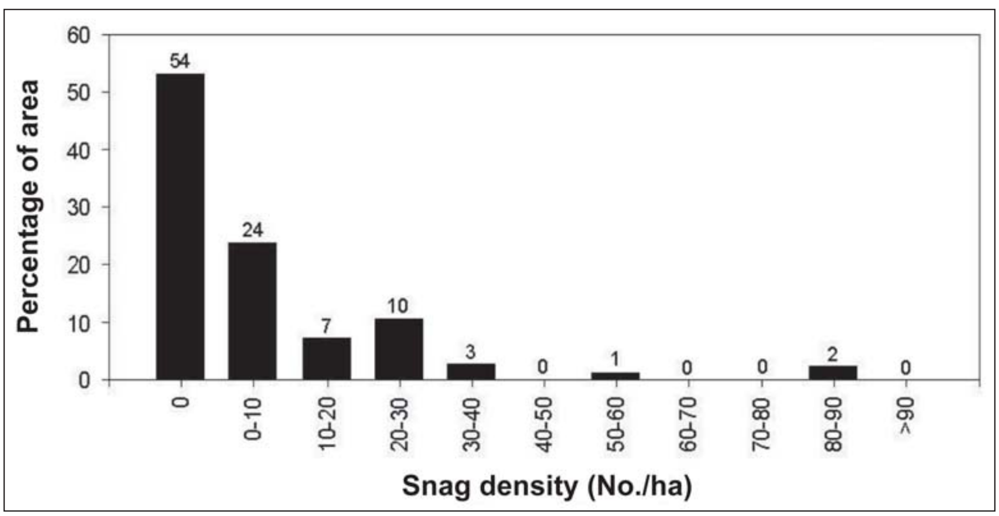

Figure 8—Distribution of density classes for snags ≥25 cm d.b.h. in unlogged and unburned ponderosa pine/Douglas-fir forest wildlife habitat and large-tree structural class based on 73 inventory plots. Derived with the program DecAID (Mellen et al. 2002; accessed online May 2006).

Managing for forest conditions characteristic of different fire severities may support the most bird species because different species respond positively to different fire severities (Smucker et al. 2005). For example, large-scale stand-replacement fires may negatively affect spotted owl (*Strix occidentalis*) populations in dry forests (Gaines et al. 1997), but habitat patchiness created by small stand-replacement fires or fires with both high and low severity may be beneficial (Bond et al. 2002). Assuming that species are adapted to persist across large landscapes, 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) (fig. 9) within a known range of variability (Agee 2003, Hessburg et al. 1999, Landres et al. 1999).

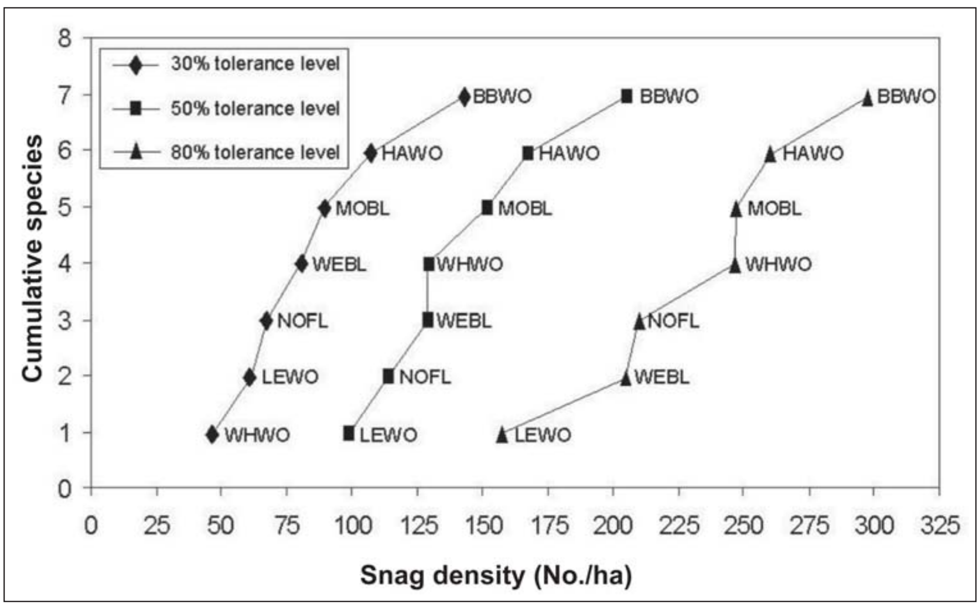

Figure 9—Cumulative bird species curves for density of snags ≥25 cm d.b.h. at 30 percent, 50 percent, and 80 percent tolerance levels for the post wildfire ponderosa pine/Douglas-fir wildlife habitat of eastern Washington and Oregon. Tolerance levels are statistical estimates of the percentage of individuals in the population that use areas with less than or equal to the indicated snag density. They define the range of management options for maintaining a given percentage (i.e., 30 percent, 50 percent, 80 percent) of the population for different species. For example, the 80 percent tolerance level for BBWO indicates that 80 percent of the individuals in the population have been found to use areas with 300 snags/ha or less. In contrast, 80 percent of LEWO individuals have been found in areas with up to 155 snags/ha. 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 the program DecAID (Mellen et al. 2002; accessed online May 2006).

Conclusions

Possible outcomes ranging from negligible to highly significant can be expected in response to logging following large wildfires. Variation in the biophysical setting of forest landscapes affects the ways in which fire and logging alter physical and biological components of forest ecosystems and natural resource values. Diverse fire phenomena and logging activities further add to variability in specific effects. Despite this variability, several principles about the effects of postfire timber harvest emerge from our review.

- Timing of timber harvest following fire affects the magnitude of ecological and economic effects. Specifically, logging can cause mortality of naturally regenerating trees if it disturbs the soil *after* trees have established, or it can prevent establishment if it removes the seed source prior to dispersal. Mortality of natural regeneration is less consequential if trees are planted to obtain desired species and stem densities. Logging soon after fire provides the highest value of wood for commercial use.
- Crown fire typically reduces the probability of future fire for years to decades. Logging reduces fuel loads over the long term, although its effects on subsequent fire potential may depend on time scale: fine fuels from decaying snags affect potential fire behavior in the short term, whereas large fuels from decaying and falling snags affect the nature of potential fire effects (e.g., smoke production) in the long term. Activity fuels created by postfire logging generally increase fire hazard in the first few years after logging unless those fuels are effectively treated.
- Fire and logging, individually and combined, affect physical, biological, and nutritional properties of soil. Fire removes the protective organic layer and exposes the soil to disturbance and compaction from ground-based logging equipment. Unless logging occurs when soils are dry or mitigation measures are applied, this can exacerbate erosion.
- Large, severe fire reduces water uptake by vegetation, causing streamflow to increase and water quality to decrease, with increased potential for debris movements. Postfire logging and accompanying roads can exacerbate these effects through increased surface-water runoff and soil erosion.
- Short-term effects of removing trees near aquatic systems are mostly negative, and logging and transportation systems that disturb the soil surface or accelerate road-related erosion may be particularly harmful.

- Most cavity-nesting birds and other cavity-nesting vertebrates are negatively affected by harvesting large standing dead trees, and possibly small trees in the short term, but effects differ with the habitat requirements of each species and the intensity, pattern, and extent of tree removal.

Incorporating postfire management of forest ecosystems in the broader task of sustainable resource management will reduce perceptions of large fires as "emergencies" or aberrations in long-term planning.

Incorporating postfire management of forest ecosystems in the broader task of sustainable resource management will reduce perceptions of large fires as "emergencies" or aberrations in long-term planning (Agee 2002a) Forest ecosystems in western North America will continue to experience large fires, and larger expanses of forests will likely burn in a warmer climate (McKenzie et al. 2004). Therefore, management of postfire environments will 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. Comparing risks associated with no action and with alternative management actions can help clarify postfire decisionmaking. It is especially important that fire management and silviculture are integrated with respect to scientific concepts and on-the-ground applications, because management of forest structural characteristics affects long-term patterns of fuels and fire hazard (Graham et al. 1999, Peterson et al. 2005). Similarly, coordination with wildlife biologists and vegetation biologists is helpful because management of forest structure affects wildlife habitat and biological diversity.

No single decision-support system exists for selecting 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, preferably in consultation 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 is usually an economic undertaking and rarely a restorative activity in the sense of ecological restoration (Society for Ecological Restoration International Science and Policy Working Group 2004). Postfire harvest may fit into an effective restoration strategy if management pathways for attaining desired combinations of species, forest structure, and ecological functions are specified. Clearly stated management objectives–which may include short- and long-term physical, biological, and economic components–are an important guide for logging or any other activity following a large wildfire.

A better understanding is needed of scale-related issues to reduce scientific uncertainty about the effects of postfire logging. First, more data are needed on the effects of different levels of (live and dead) tree retention, including variable densities (clumps). Second, more information is needed on how the proportion of harvest area relative to total fire area affects the magnitude of change in different resources. Finally, longer time series (10+ years after fire) of data on the effects of postfire logging are needed to accurately quantify long-term resource responses.

Large wildfires are opportunities to implement long-term management experiments to improve the scientific basis for decisionmaking. 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. Potential treatments would include time since fire (e.g., <1 year, 3 years), postharvest stand density (e.g., high [no harvest], moderate, low), and fuel treatment (e.g., none vs. effective removal of fine fuels). 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, small mammals, amphibians, and aquatic organisms. Such experiments would ideally continue for at least 10+ years (and preferably much longer) to capture temporal trends in postfire and postharvest responses. Sampling intensity and frequency would initially be high, but could decrease over time depending on trends in the data.

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English Equivalents

When you know:	Multiply by:	To find:
Centimeters (cm)	0.394	Inches
Kilometers (km)	.621	Miles
Square kilometers (km^2)	10.76	Square feet
Hectares (ha)	2.47	Acres
Cubic meters per second	35.31	Cubic feet per second
Cubic meters per hectare	35.3	Cubic feet per acre
Kilograms per cubic meter (kg/m^3)	.0624	Pounds per cubic foot
Kilograms per hectare (kg/ha)	.893	Pounds per acre
Megagrams per hectare (Mg/ha)	.446	Tons per acre

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