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Cumulative Watershed Effects of Fuel Management in the Western United States

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Abstract

Fire suppression in the last century has resulted in forests with excessive amounts of biomass, leading to more severe wildfires, covering greater areas, requiring more resources for suppression and mitigation, and causing increased onsite and offsite damage to forests and watersheds. Forest managers are now attempting to reduce this accumulated biomass by thinning, prescribed fire, and other management activities. These activities will impact watershed health, particularly as larger areas are treated and treatment activities become more widespread in space and in time. Management needs, laws, social pressures, and legal findings have underscored a need to synthesize what we know about the cumulative watershed effects of fuel management activities. To meet this need, a workshop was held in Provo, Utah, on April, 2005, with 45 scientists and watershed managers from throughout the United States. At that meeting, it was decided that two syntheses on the cumulative watershed effects of fuel management would be developed, one for the eastern United States, and one for the western United States. For the western synthesis, 14 chapters were defined covering fire and forests, machinery, erosion processes, water yield and quality, soil and riparian impacts, aquatic and landscape effects, and predictive tools and procedures. We believe these chapters provide an overview of our current understanding of the cumulative watershed effects of fuel management in the western United States.

Keywords: cumulative effects, watershed, wildfire, fuel management, water quality, soil erosion

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Foreword

This document is the result of a major interdisciplinary effort to synthesize our understanding of the cumulative watershed effects of fuel management. This document is the product of more than 20 authors and 40 reviewers including scientists from four Forest Service Research Stations and numerous universities. Chapter outlines and contents were first reviewed at a workshop in April 2005. Authors then drafted chapters that were peer-reviewed over the next two years. We edited all chapters twice before submitting them for a third round of editing by RMRS publication specialists. Chapter topics include overviews of the effects of fuel management on both terrestrial and aquatic watershed processes. The other editors and I are grateful to all authors and reviewers for their considerable efforts in the development of this document over the past four years. We wish to acknowledge the Stream Team and the National Fire Plan for financial assistance. As with all syntheses, science will continue to generate new knowledge, which will in time supersede the contents of this document. Readers are encouraged to seek updated and locally derived information to supplement the contents of this document. My personal thanks go to all the authors, reviewers, my coeditors and RMRS publishing staff for the considerable effort necessary to develop and publish this synthesis.

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Contents

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Chapter 1.

Introduction to Synthesis of Current Science Regarding Cumulative Watershed Effects of Fuel Reduction Treatments

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Introduction

This report was produced by a group of scientists who were invited by the U.S. Forest Service to synthesize the current scientific literature to answer an important question facing the managers of federal and private lands in many parts of the country. The question was: What potential cumulative environmental effects at the watershed scale might be caused by implementing land management activities that reduce forest fuels on large scales? The main body of this report is a compilation of what they found, including both what can and cannot be concluded from the current science.

The scientific principles reviewed in this report are intended to be general. The examples of fire environments, land management practices, and vegetation types are drawn primarily from the western continental United States, roughly the region west of 100° W longitude. A follow-up report is planned to address the same question but with a focus on examples from the Eastern section of the continental United States (Lafayette and others, under review).

In this chapter, I broadly describe fuel reduction treatments on wildlands and the concept of analyzing cumulative watershed effects. For perspective, I have referred to some of the primary legislative and policy direction that influences the way that federal land managers apply fuel reduction treatments and analyze cumulative effects.

Fuel Reductions Treatments and Policies and Laws Related to Them

Fuel reduction treatments are land management actions taken to reduce the threat posed by severe and/or intense wildland fire by manipulating live and dead vegetation to reduce the loading of fuel on the landscape that can support wildland fire. Fuel reduction can be accomplished in a number of ways with the most common involving mechanical removal of fuel material (usually brush or trees) and/or consumption of fuel using prescribed fire. These two types of treatments may be applied alone, sequentially, or in various combinations. Treatment of wildlands to reduce fuel was given a national mandate when Congress passed the Healthy Forests Restoration Act of 2003 (P.L. 108-148) (HFRA). Efforts to reduce the risk of severe wildland fire may receive additional impetus from studies that attribute recent increases in the frequency of large wildland fires to changing climate (Westerling and others 2006), raising the possibility that this threat may further increase in western forests with expected climate changes in future years.

To effectively reduce the risk of wildland fires, fuel reduction treatments will need to be applied to large areas of federal lands each year in the form of mechanical fuel removal and/or prescribed fire. In most cases, vegetation can regrow after treatments, meaning that, in many areas, maintaining low fuel stocking will require repeated treatments to reduce fuel stocking at intervals ranging from several years to a few decades. Where they are well designed and implemented, fuel reduction treatments will probably create a relatively low intensity of disturbance, for example, by disturbing only a small amount of surface soil per acre. However, because they will be carried out over many acres each year and entail return treatments at regular intervals, there remains the possibility that local, project-level impacts of fuel reduction treatments may add up to significant impacts at larger watershed scales and thus result in cumulative effects. These cumulative effects on watersheds might be caused by activities directly related to removing fuels (for example, felling, skidding, and/or chipping to mechanically remove fuel and/or prescribed fire to consume it). Cumulative effects might also include impacts of operations or infrastructure that support fuels reduction that may occur in areas at some distance from the actual site of fuel reduction. Examples of these supporting functions could include the movement from logging, fire control, and other vehicles used in fuel management. Examples of supporting infrastructure could include roads that provide access to large areas of the landscape where fuel management activities take place and the drainage ditches, culverts, and stream crossings associated with roads.

In recognition of the critical role of wildland fire in forest ecosystems and the risk that high fuel loads pose in forests, Congress and the Forest Service have taken several actions to accelerate fuel reduction treatments. Several policy initiatives by the Forest Service and other federal agencies (Federal Wildland Fire Policy of 1995, the Cohesive Fire Strategy of 2000, National Fire Plan of 2000, and Ten-year Comprehensive Strategy and Implementation Plan of 2001/2002) were strengthen when the President announced the Healthy Forest Initiative (HFI) (White House 2002). HFI streamlined administrative procedures required to implement fuel reduction treatments, including approval for using the Categorical Exclusion for fuel reduction treatments, a process for complying with the National Environmental Policy Act of 1969 (P. L. 91-190) (NEPA) that reduces the documentation required for planning these activities. In addition, HFRA reduced the number of alternatives that must be considered in NEPA analysis and added specific public involvement and collaboration requirements. Although these initiates streamlined consideration of fuel reduction treatments under NEPA, they did not exempt these treatments from the requirements of the Act.

What Are Cumulative Watershed Effects?

When a federal agency proposes a land management action, its potential cumulative environmental effects must be considered, along with direct and indirect effects, and documented in a public report that the proposing agency prepares to comply with the NEPA. This report may be an Environmental Impact Statement (EIS), Environmental Assessment (EA), or Categorical Exclusion (CE), depending on the nature of the action and the likelihood of significant effects. Thus, to comply with NEPA, cumulative effects are among the environmental impacts of proposed major federal actions and any alternative actions that federal agencies must present for public review.

The basic concept of a cumulative effects analysis is to identify and consider the total effects of actions that overlap temporally or spatially and might be missed by evaluating each action individually. The goal of cumulative effects analysis is to provide government decision makers and the public considering proposed federal projects with comprehensive information about "the impact on the environment which results from the incremental impact of the action when added to other past, present and reasonably foreseeable future actions"(40 CFR 1508.7). In general terms, cumulative effects may arise from single or multiple actions that may result in additive, interactive, direct, or indirect effects. Cumulative watershed effects are the net impact on watersheds of multiple management activities that may coincide geographically and temporally.

Although cumulative effects are defined by NEPA, concepts related to cumulative watershed effects also come into play in the application of other more restricted environmental laws. The Clean Water Act of 1972 (P.L. 92-500) (CWA) requires that both point sources (in other words, having a readily identifiable origin) and nonpoint sources (such as, lacking a readily identifiable origin) of water pollution be controlled, especially in waters that have been designated by states as not meeting water quality standards under section 303b of CWA. A common reason that nonpoint sources are not easily attributable to distinct locations is that they are spatially and/or temporally dispersed and thus may be the result of cumulative watershed effects. In addition, the Endangered Species Act of 1973 (P.L. 93-205) (ESA) protects species that are at risk of extinction (listed under the Act as "threatened" or "endangered") from actions by federal agencies that could reduce the number of these organisms or their habitat. Where species listed under ESA dwell in aquatic or riparian habitats, these species or their habitat may be at risk from multiple management activities occurring at a watersheds scale, that is, as a result of cumulative watershed effects. These examples are not exhaustive. Laws such as the Clean Air Act of 1970 (P.L. 91-604), National Historic Preservation Act of 1966 (P. L. 89-665, section 106), and others have requirements of their own that, under some conditions, may call for a cumulative effects analysis. If, for a given landscape, the consideration of cumulative watershed effects for complying with multiple laws becomes an issue for fuel reduction treatments or other land management practices, a more comprehensive watershed-scale analysis that meets the requirement of all these laws simultaneously might be warranted. This synthesis could provide a scientific basis for developing such a comprehensive watershed analysis to address multiple laws should federal land managers find it necessary.

Evidence of Cumulative Watershed Effects

Implementation of fuel reduction treatments at large scales has only begun recently so direct evidence of its cumulative watershed effects is likely to be scarce. However, cumulative effects of other land management activities have been measured at watershed scales. A classic example comes from a study of fish habitat in streams from across the Columbia River Basin (McIntosh and others 2000). In this study, habitat for anadromous fish in 122 streams that had been originally surveyed in the 1930s and 1940s was remeasured in the 1980s and 1990s. With the exception of streams in roadless watersheds, the prime habitat features in these streams showed significant losses over the intervening 60 years (for example "large pools" decreased by 24% and "deep pools" decreased 65%). An analysis of land management practices in these watersheds showed that no single practice or project was clearly responsible for the loss of habitat in these watersheds. Instead, they found that a wide spectrum of land uses had occurred within the watersheds of the degraded streams, including forestry, grazing, urbanization and road construction. It was the aggregate impact of all these practices, that is, the cumulative effect of all the land uses that had caused the habitat loss. By comparison, in watersheds with little or no change in land use over the period (which the authors represented by examining watersheds with no roads), aquatic habitat condition had remained constant or improved, reinforcing the conclusion that cumulative effects of multiple land management activities had caused the degradation. The lesson from this study for large-scale fuel management is that wide-spread land management activities have the potential to cause significant, real impacts on aquatic systems even where the impacts of individual local projects may be small or difficult to measure.

Considering Cumulative Watershed Effects in Fuel Management

It is critical that cumulative watershed effects be considered early as part of planning and implementing fuel reduction treatments in the current legal and policy environment. Under HFI and HFRA, NEPA analysis is not waived. HFRA streamlined NEPA analysis by reducing the number of alternatives that must be considered and added requirements for public collaboration, but did not exempt or waive any projects from NEPA analysis. While HFI included Categorical Exclusions for fuel reductions treatments, the intent was to use an approach that more efficiently complies with NEPA requirements and likewise does not waive consideration under NEPA. Although a brief discussion of NEPA requirements follows, it is the purpose of this synthesis to assess what valid scientific information is available to assess the cumulative watershed effects of fuels reduction treatments, rather than explain in detail the legal requirements for documenting these effects. Planning teams may use information in this report to produce environmental documents at the appropriate level.

On National Forest lands, the forest planning rule approved in 2008 required that forest plans be written at a broad strategic scale. The Forest Service has determined that, at this broad scale, forest plans do not have environmental effects. Therefore, forest plans qualify for a Categorical Exclusion and do not require documentation in an EIS. As has been previously true, NEPA analysis will continue to be done for individual projects that are on the scale at which ground-disturbing activities are analyzed. In determining the scope of a proposed action, and the level of NEPA analysis needed, it is the responsible public official who is required to consider the action's environmental effects, including direct, indirect, and cumulative impacts (see 40 CFR 1508.25). In other words, the responsible public official decides the appropriate level of detail for the cumulative watershed effects analysis for the fuels reduction project(s) in question. Under the 2008 planning rule, site-specific actions are required to comply with NEPA, and cumulative watershed assessments will be a key part of the strategic resource decisions on these forests in the future.

The Council on Environmental Quality (CEQ) gave guidance on when to include cumulative effects in NEPA analysis (CEQ 1997). A recent memo (CEQ 2005) stated that "except in extraordinary circumstances, proposed actions that are categorically excluded from NEPA analysis do not involve cumulative effects analysis." This means that to be categorically excluded, a project must fit within specifically defined categories and must not involve extraordinary circumstances. For the Forest Service, extraordinary circumstances are defined as the degree of environmental impact to seven specific resource conditions listed in the Forest Service Handbook 1909.15, chapter 30, section 30.3. Fuel reductions treatments that meet these requirements may not have to undergo detailed cumulative watershed effects analyses.

Ultimately, it is the responsibility of the leadership and employees of the agency to find a balance between the broad set of laws and policies designed to produce the intended effect of protecting the public from wildland fire, while at the same time complying with other sets of laws intended to protect natural resources and the environment. If members of the public disagree with the balance that is struck, they have the right to challenge these management decisions through administrative procedures in the courts where these differences will be resolved.

Within this changing legal and policy arena, tools for analyzing cumulative watershed effects are likely to remain important for managers of natural resources. Cumulative effects are real, and sustaining multiple natural resources over the long run will require that they be considered. Streamlining requirements for analysis under NEPA or other rules assumes that these practices have impacts that are either insignificant or small compared to the long-term benefits from the proposed action. Courts and public opinion will likely place the burden on land managers to demonstrate that those assumptions are valid. The standard for showing that kind of validity usually requires predictions that are supported by the current science.

The Need for Science-Based Decisions

The unprecedented scope and scale of fuels reduction treatments being undertaken by Federal land managers also makes a strong argument for developing available and scientifically based tools to estimate their potential cumulative effects. A recent, tragic example illustrates the vital role that tools play in bringing scientific information to bear on large scale problems. An analysis of the structural failure that caused the collapse of the World Trade Center Towers (WTCT) on September 11, 2001 was presented on Public Television (PBS, 2002). Paradoxically, the analysis showed that the buildings were not brought down by the physical impact of the airplanes that crashed into them. Instead, the buildings failed because of damage caused by the heat of the subsequent fires that were fueled primarily by the contents of the building themselves and not by the aircraft fuel that had only served to start the fire.

In an interview, the lead engineer who designed the buildings pointed out with some pride that at the time they were designed, the WTCT were not only the tallest buildings ever constructed but were also the first designed to withstand the physical impact of a large airplane. His design had succeeded to the degree that the initial impacts had not brought the buildings down. When asked why the buildings had failed to survive the effects of the subsequent fire, the engineer admitted that he was not able to consider the effects of such a fire in the buildings' design. At the time, there were no models available of how a fire of that magnitude would affect the structure of such a large building. In effect, the science of building fires was not available in the form and format (that is as a useful tool) that would permit practitioners to use it to evaluate this critical question at the scale of the problem they faced. The lesson from this example is that when decision makers must tackle projects outside of the previous scope and scale that they have experienced, the relevant body of science, where it exists, may be of little practical use if it has not been interpreted and articulated as tools that are directly useful for addressing problems at the proper scale.

Applying this lesson to fuels reduction treatments, land managers are planning these treatments at unprecedented geographic and temporal scales. If tools based on current science are not available, it is possible that planners and managers may not be able to include cumulative watershed effects in a meaningful way in designing and implementing fuel reduction treatments. Alternatively, if courts decide that fuel reduction treatments cannot proceed unless adequate cumulative watershed effects analysis are part of their planning, these treatments may be delayed until useful tools are developed that incorporate the latest science. In either case, developing tools that predict cumulative watershed effects of fuels reductions based on current science will be important.

Potential Uses of This Synthesis

This synthesis of the current literature on cumulative watershed effects is a first step toward developing useful tools for managers to consider these effects when planning and implementing fuel reduction treatments. It assembles in one place the current state of thinking that was previously scattered across many outlets in the scientific literature. At the minimum, it should provide managers, planners, and policy makers with a place to start when they have questions about this topic.

This synthesis, however, goes beyond being a central source of scientific information on this subject. Although cataloguing and summarizing the literature are useful, this report tries to go further to anticipate questions that are likely to be posed by managers, planners and policy makers by asking them of the literature. By doing this, the audience can be informed about what relevant questions the current science can and cannot answer. The "science gaps" are at least as important as the current knowledge because it is in these gaps that management and policy may lack science for guidance. These are areas in which managers and policy makers should be cautious because the outcomes of their actions in these areas may not be reliably predicted with a scientific basis, produce unforeseen outcomes, or be vulnerable to legal challenges. Identifying critical knowledge gaps also performs an important function for the science community. Future research and development that fill these gaps could potentially have a high payoff for managers and policy makers. This science synthesis may only be a first step toward providing useful tools. If managers, planners, or policy makers find that they need more detailed or explicit tools, the peer-reviewed knowledge gathered in a synthesis can provide an information base from which to start developing tools that meet those further needs.

The value of this synthesis will depend strongly on how well it reinterprets existing knowledge in the face of a new question. While it is true that a synthesis of the current literature may be a reworking of existing information, asking new questions of old data often casts them in new light. When done thoughtfully, new questions may suggest new insights that have not previously been considered. The questions considered here are indeed new because they involve the implications of a new management practice being imposed on the landscape at unprecedented scales of space and time.

Conclusion

In the end, this synthesis will be judged by its usefulness to future policy and management decision makers and as a starting point for future researchers and tools developers. We will leave it to you, our readers, to decide how well this document meets your particular needs. As you use this document, if you find other pressing questions that we did not anticipate or address, we urge you to ask the science community to answer them. If this report is used in these ways, it will have served its purpose to advance the state of land management and policy making and set the stage for future research and tools development.

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Fire Regimes and Ecoregions[1]

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Introduction

The public land management agencies are phasing in a radically new approach to land management. They are shifting from their focus on individual resources to a more holistic approach of managing *whole ecosystems*. Fire-excluded systems are prone to changes in composition and density and are susceptible to catastrophic fire and invasion by non-native species. The cause of the problem in many areas includes more than a century of fire exclusion and suppression along with increased human development at the wildland-urban interface. Grazing and logging have also contributed to this problem.

To correct this problem, fire and land management must return ecosystems to a healthier, *sustainable* condition. One way to do this is to modify the current structure of ecosystems to mimic natural structures (Bailey 2002).

Ecosystem Structure and Process

Ecosystem structure and process are related. For example, riparian forests evolved with flooding, and fire-adapted forests evolved with fire of varying frequency and intensity. For ecosystems to be able to sustain natural structures, they will need to experience the same kinds of processes in which they evolved (Allen and others 2002; Savage 2003).

Range of Variation

Restoration works best if ecosystems are returned to within a "natural range of variation" (Landres and others 1999). Ecosystems, for example, have variability not only through time because of climate change, but across the landscape and the nation because of disturbance events, successional processes, and natural climatic variation in addition to climate change. From forests, to desert, to steppe, the continent's ecosystems vary vastly. It is not possible to reconstruct how each system looked in the past. Instead, we can reset altered ecosystems back to within a range of natural variability. As Melissa Savage (2003) puts it, "If we can restore the natural processes, the natural structure should follow."

[1] This chapter is derived from a presentation at the annual meeting of the Association of American Geographers, Denver, Colorado USA, held April 5 through 9, 2005.
[2] Formerly with the Inventory and Monitoring Institute, USDA Forest Service, Fort Collins, Colorado.

Climate, Ecoregions, and Fire Regimes

To restore altered systems and to understand how and why they are distributed, we must understand the processes of how they form. Climate largely determines ecosystem differences. As it varies, the other components vary in response. As a result, ecosystems of different climates differ significantly (Bailey 1996).

The most important climatic factor in determining the distribution of ecosystems is the climatic regime, defined as the daily and seasonal fluxes of energy and moisture. For example, tropical rainforest climates lack seasonal periodicity, whereas mid-latitude steppes have pronounced seasons. As the climatic regime changes, so does the hydrologic cycle, as reflected in the streamflow of rivers located in different climatic regions. For example, no water flows in creeks located in the warm, dry summer region of California during summer and fall, but in winter and early spring, groundwater contributes to streamflow. Climate profoundly affects landforms and erosion cycles. Such effects are evident when we contrast the angularity of arid land topography of the Colorado Plateau with the rounded slopes of the humid Blue Ridge Mountains. Plants and animals have adjusted their life patterns to the basic environmental cycles produced by the climate. Whenever a marked annual variation occurs in temperature and precipitation, a corresponding annual variation occurs in the life cycle of the flora and fauna. Climate also determines the distribution, frequency, and density of natural ignitions.

Controls over the climatic effect change with scale. At the macroscale, ecosystem patterns are controlled by macroclimate (in other words, the climate that lies above the local modifying effects of landform and vegetation). Based on macroclimatic conditions, I subdivided the continents into ecoclimatic zones, also known as ecosystem regions, or *ecoregions*. They were mapped at a scale of 1:30,000,000 (Bailey 1989). Three hierarchical levels of macroclimatic differentiation are shown (in order that reflect finer scale climatic differences: *domain*, *division*, and *province*). The domains and divisions are based largely on the climatic zones of Köppen (1931) as modified by Trewartha (1968). While the Köppen-Trewartha system is based on climate, there is good correspondence between its climatic types and the natural climax vegetation types and soils within them (Bailey 1996)[3]. Because of this, it was considered a logical basis for ecological zoning. A major advantage to using the system is it is based on quantitative definitions and as such, can be applied to any part of the Earth where climatic data are available[4]. The zoning was made hierarchical using Köppen-Trewartha's climatic groups and climatic types as ecoregion domains and divisions, respectively, (table 1). At the second level (division), further differentiation was made according to landform—distinguishing mountains with altitudinal zonation from lowland plains.

The climate is not completely uniform within level 2 (division), so a further subdivision was undertaken. Within the dry climates, for example, there is a wide range of degree of aridity, ranging from very dry deserts through transitional levels of aridity in the direction of adjacent moist climates. We refer to these as climate subtypes. The subtypes largely correspond to major plant formations (for example, broadleaved forest), which are delimited on the basis of macro features of the vegetation by concentrating on the life-form of the plants. They form the basis for subdividing ecoregion divisions into provinces, and are based on a number of sources, including a world map of landscape types (Milanova and Kushlin 1993). Of course, not all the space is taken up by the formation, because the nature of the topography will allow the differentiation into many habitats. One ignores these local variations in mapping climatic regions (and therefore ecoregions).

[3] This is largely because Köppen derived his climate classes from observations on the distribution of vegetation types on various continents.
[4] Others have followed this precedent for using Köppen-Trewartha in global ecological zoning. One good example was carried out by the United Nation's Foreign Agricultural Organization (FAO), who developed an ecological zone map for the Forest Resource Assessment 2000 (FAO 2001) using a combination of vegetation characteristics and Köppen-Trewartha.

Table 1. Ecoregion global framework.

Ecoregion Level 1 – Domain		Ecoregion Level 2 – Division		
Name	Criteria (equivalent to Köppen-Trewartha[a] climatic groups)	Name	Code	Criteria (approximate equivalent Köppen-Trewartha[a] types, in combination with vegetation physiognomy)
Polar	No more than 3 months over 10 °C	Icecap	11	All months below 0 °C. Perpetual snow and ice.
		Tundra	12	All months below 10 °C. Vegetation physiognomy: tundra.
		Subarctic	13	Up to 3 months over 10 °C. Vegetation physiognomy: dense coniferous forest dominant.
Humid Temperate	Eight months or less over 10 °C	Warm continental	21	Same as 220, warmest below 22 °C.
		Hot continental	22	4-7 months over 10 °C, coldest month below 0 °C, warmest above 22 °C.
		Subtropical	23	Same as 260, no dry season.
		Marine	24	4-7 months above 10 °C, coldest month over 0 °C.
		Prairie[b]	25	Sub-humid. Vegetation physiognomy: grasslands.
		Mediterranean	26	8 months over 10 °C, coldest below 18 °C, dry summer.
Dry	Evaporation > precipitation	Tropical/subtropical steppe	31	Semi-arid: all months above 0 °C.
		Tropical/subtropical desert	32	Arid: ½ precipitation of steppes, all months above 0 °C.
		Temperate steppe	33	Semi-arid: cold month below 0 °C.
		Temperate desert	34	Arid: All months dry, cold month below 0 °C.
Humid Tropical	All months without frost: in marine areas over 18 °C	Savanna	41	Same as 420, with 2 months dry[c] in winter
		Rainforest	42	Wet: no dry season

[a] Köppen (1931), as modified by Trewartha (1968)

[b] Köppen (1931) did not recognize the prairie as distinct climatic type. Geographer's recognition of the prairie climate (Borchert 1950) has been incorporated into the system presented here. The ecoregion classification system represents it at the arid side of the 210, 220, and 230 types.

[c] A dry month is defined as the month in which the total precipitation, P, expressed in millimeters, is equal to less than twice the mean temperature (°C).

I also used this approach to construct a 1:7,500,000-scale map of ecoregions for the United States (Bailey 1976, revised 1995). A simplified, reduced-scale map of the second highest of three hierarchical levels (division) appears in (fig. 1). They were delineated using both macroclimate data and a map of existing climax or potential vegetation by Küchler (1967). More information on the rationale I used for identifying ecoregion boundaries on maps of the United States and the world's continents is presented elsewhere (Bailey 2005). In 1993, as part of the National Hierarchical Framework of Ecological Units (Cleland and others 1997), the Forest Service adopted the ecoregion classification system for use in ecosystem management.

It seems reasonable that regions that differ substantially in background climate should have different fire regimes. In fact, fires burn with more or less regular rhythms. The simplest means to reveal a fire regime is to consider the distribution of water within an ecosystem. If they are too wet, they won't burn. The ecosystem's moisture changes with the daily and seasonal fluxes of the moisture of air masses as they move through the region. Long-term fire records around the Pacific Ocean trace nicely the pulses of the Pacific Decadal Oscillation. In this oscillation, the Pacific alternates from warm to cool phases and causes wet and dry periods on the adjacent North American continent. These wet-dry rhythms set the ecological cadence for fire regimes.

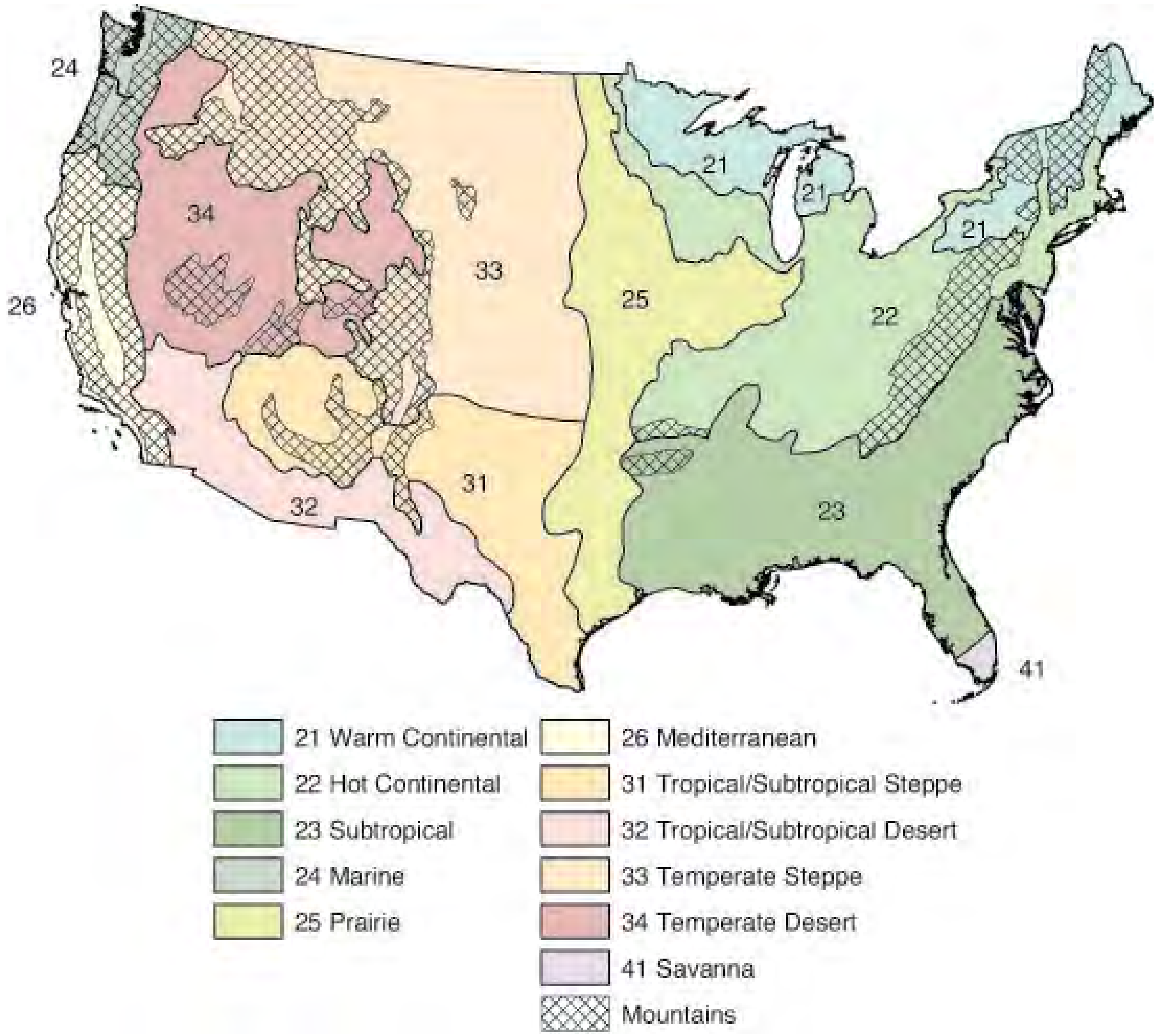

Figure 1. Approximate boundaries of ecoregion divisions, conterminous United States from Bailey 1996.

Different Ecoregions, Different Fire Regimes

Different ecoregions produce different fire regimes. There are several studies that have looked at variation in fire regimes at the ecoregion scale. We will examine three of them.

Pre-Colonial Fire Regimes (Vale)

Pre-colonial fire regimes for different vegetation types in North America have been determined by analyzing fire scars. In areas lacking trees, the development of vegetation after recent fires and early journal accounts and diaries have been used to make inferences about fire regimes. Vale (1982) synthesized this information in his book, *Plants and People*.

Vale analyzed "natural" vegetation types based on ecoregions. He characterized fire regimes from 45 published studies of fire regimes or from his estimates of the fire regimes based on the fire ecology of the plant species in the areas mapped. He found that fire regimes varied by ecoregion (fig. 2). In the northern coniferous forest and woodland (boreal forest), for example, infrequent large-magnitude fires carried the flames in the

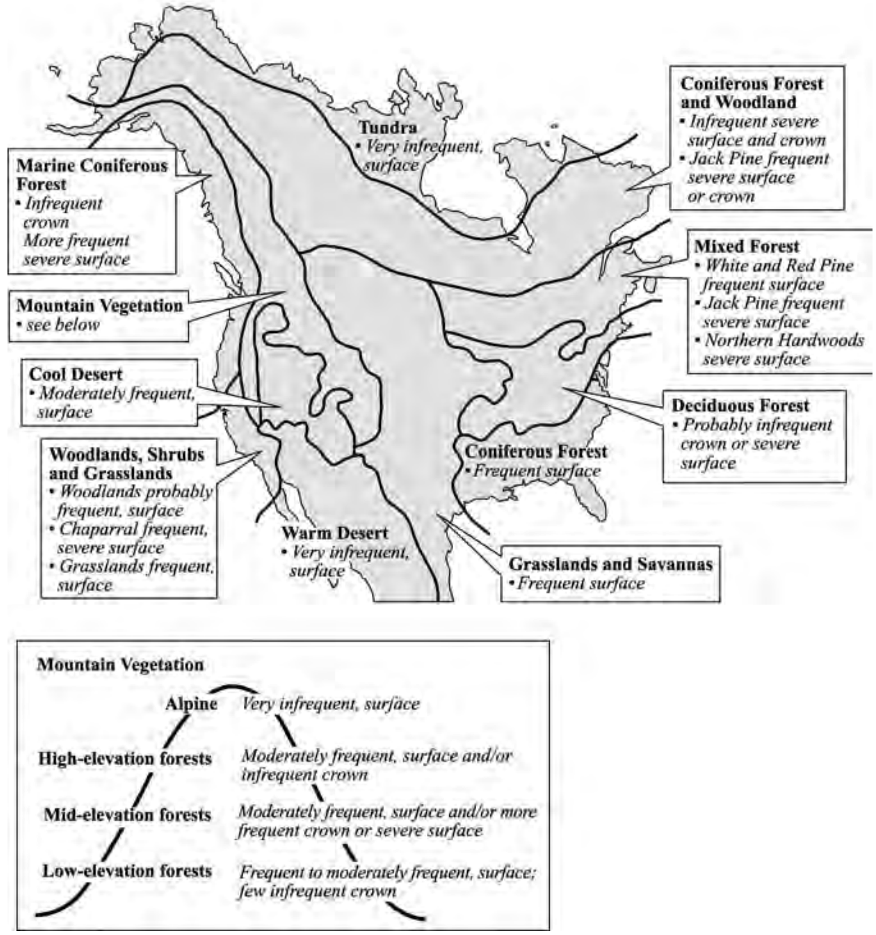

Figure 2. Pre-colonial fire regimes of broad vegetation types (based on ecoregions) in North America. Only major divisions of the ecoregion map are shown (from Vale (1982); reprinted with permission of the Association of American Geographers).

canopy of the vegetation, killing most of the forest. Such fires are called “crown fires” because they burn in the upper foliage or crown of the trees.

Other environments, such as the deciduous forests of the east, probably had infrequent crown or severe surface fires. These areas are typically cool or wet and consist of vegetation that inhibits the start or spread of fire.

In mountainous regions, fire frequency is related to altitude, or elevation. The lower-elevation forests in the western United States had a regime of frequent, small-magnitude, surface fires. Here, the burning was restricted to the forest floor and most mature trees survived. Ponderosa pine (*Pinus ponderosa*) forests are good examples of this kind of forest.

Fire Regime Types (The Nature Conservancy)

The Nature Conservancy (2004), working in cooperation with the World Wildlife Fund and the International Union for Conservation of Nature, has recently completed a global assessment of fire regime alteration on an ecoregional basis. The assessment identified three broad fire regime types (fig. 3). The report reveals that, among globally important ecoregions for conservation, 84 percent of the area is at risk from altered fire regimes. Almost half of priority conservation ecoregions can be classified as “fire-dependent” (shown in reddish brown).

In *fire-dependent systems*, fires are fundamental to sustaining native plants and animals. Many of the world’s ecosystems, from taiga forest to chaparral shrublands to the savanna, have evolved with fires. What characterizes all of these ecosystems is resilience and recovery following exposure to fires. In the case of chaparral, fire does not kill most of the shrub layer, they sprout back from root crowns.

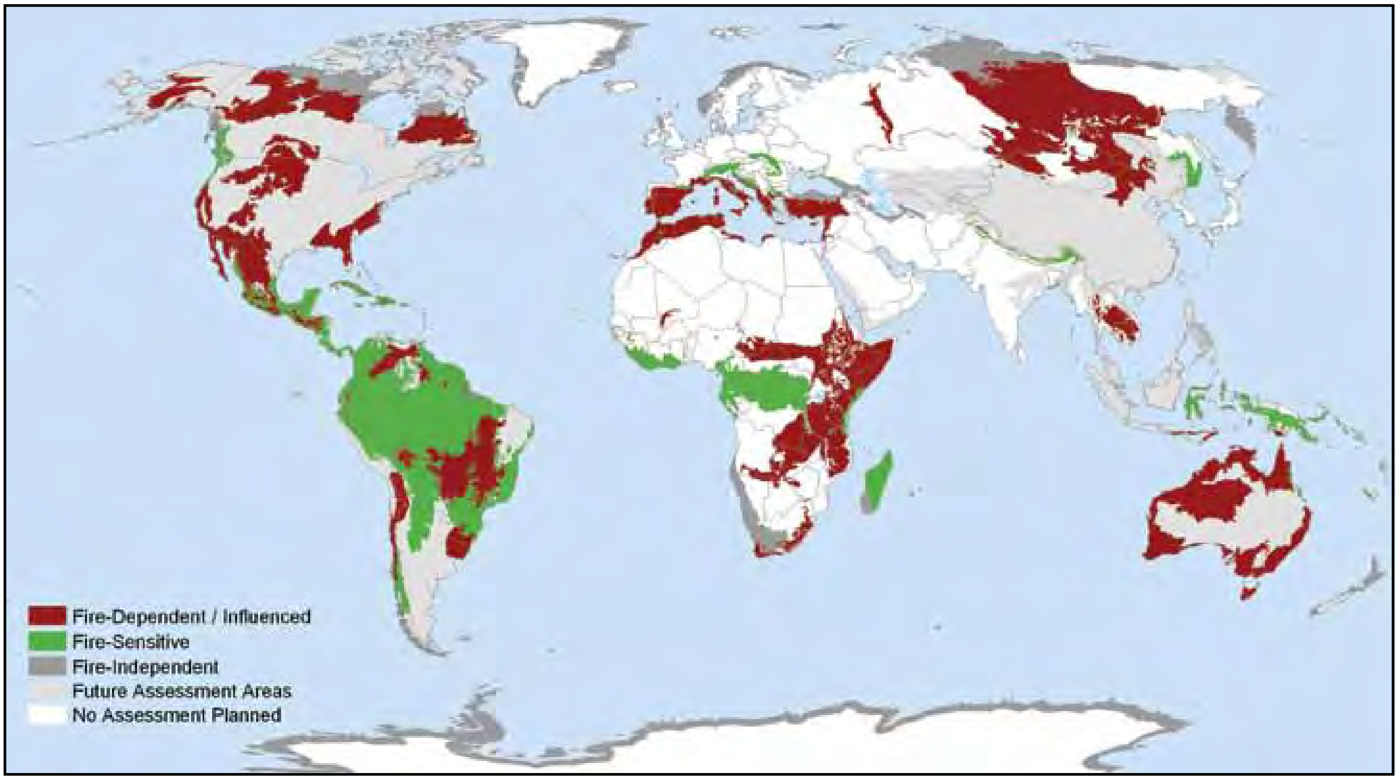

Figure 3. Dominant fire regimes in priority ecoregions for biodiversity conservation. Reddish brown areas are fire dependent; green areas are fire sensitive; and gray areas are fire independent (Copyright, The Nature Conservancy; reproduced with permission).

Thirty-six percent of these important ecoregions are *fire-sensitive*. In these regions, frequent, large and intense fires were, until recently, rare events. In these systems, plants lack adaptations to allow them to rapidly rebound from fire. These areas are typically cool or wet and consist of vegetation that inhibits the start or spread of fire. Examples include the tropical moist broadleaf forest and temperate rainforests.

Eighteen percent are classified as *fire-independent ecosystems*. Here, fires are largely absent because of a lack of vegetation or ignition sources, such as in Africa's Namibian Desert or in tundra ecosystems in the arctic.

According to The Nature Conservancy (2004), fire regimes are degraded in over 80 percent of globally important ecoregions. The majority of North American forests and grasslands are adapted to fire of varying frequencies and intensities. Fire-excluded systems are prone to changes in composition and density, and are susceptible to catastrophic fire and invasion by non-native species. Fire-loving invasive alien plants can drastically change the fire regimes of both fire-dependent and fire-sensitive ecosystems (fig. 4).

Characterizing U.S.A. wildfire regimes (Malamud, Millington, and Perry)

Researcher Bruce Malamud and colleagues report that the spread of wildfires and their severity patterns show distinct regional styles across the United States (Malamud and others 2005). Using high-resolution Forest Service wildfire statistics, this study was based on 31 years (1970-2000) of wildfire data consisting of 88,916 fires ≥1 acre on the National Forest System. To allow spatial analysis with regard to the biophysical factors that drive wildfire regimes, the researchers classified the wildfire data into ecoregion divisions (areas of common climate, vegetation, and elevation). In each ecoregion, they asked: What is the frequency-area distribution of wildfires? The study compared area

Figure 4. Non-native cheatgrass (*Bromus tectorum*) is invading sagebrush steppe in the western United States (photograph copyright, The Nature Conservancy; reproduced with permission).

burned, number of fires, and the wildfire recurrence interval. These parameters were calculated at the ecoregion division level (fig. 5). The study created maps to display wildfire patterns and risk for the entire continental United States.

The authors found that the ratio of large to small wildfires decreases from east to west (fig. 6B). There is a relatively higher proportion of large fires in the west compared to the east. This may be due to greater population density and increased forest fragmentation. Alternatively, the observed gradient may be due to natural drivers, with climate, vegetation, and topography producing conditions more conducive to large wildfires.

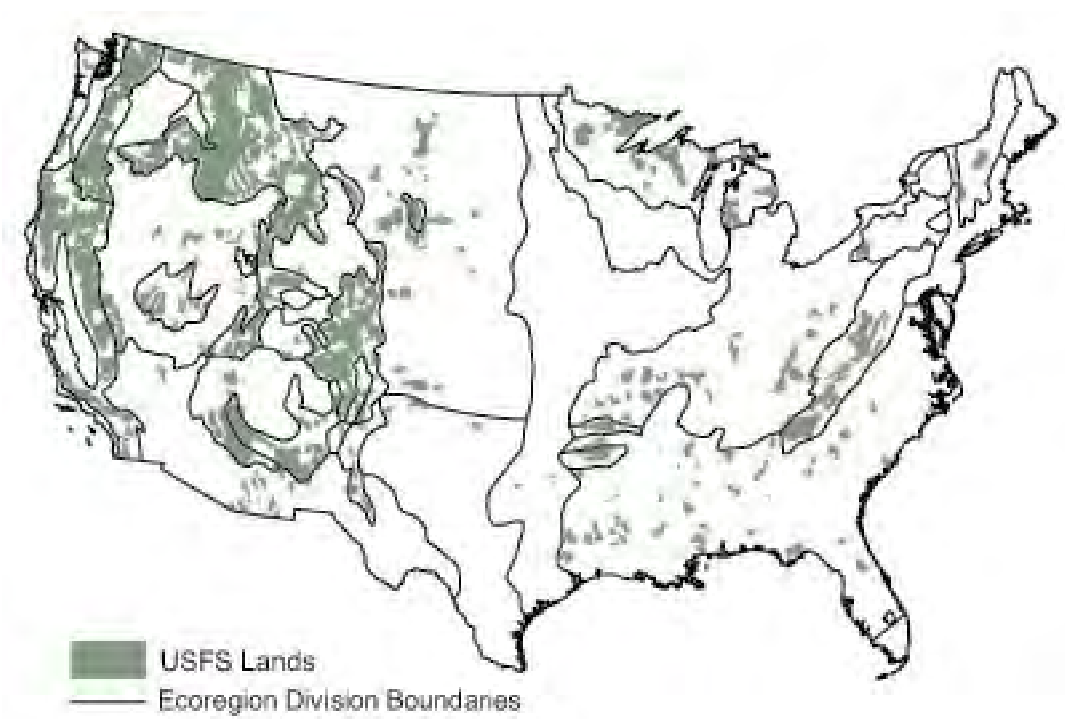

Figure 5. Ecoregion divisions and U.S. Forest Service lands.

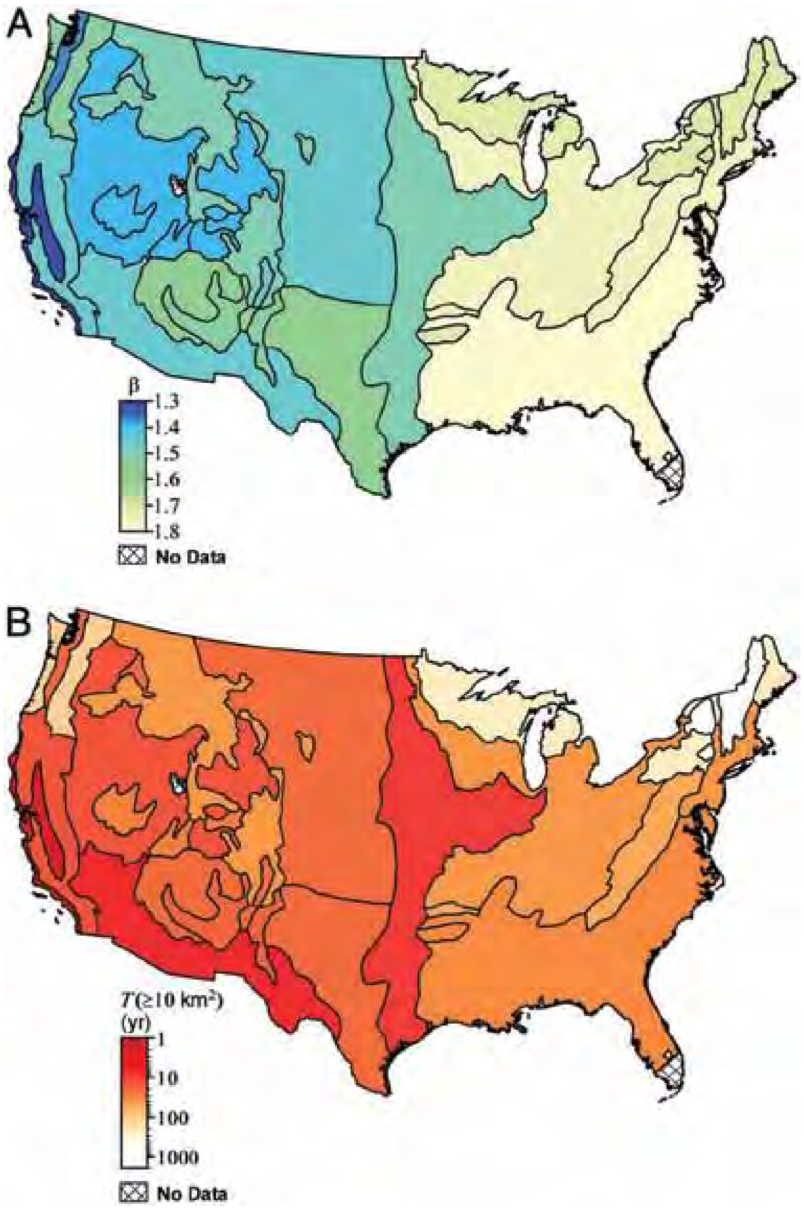

Figure 6. Maps of wildfire patterns across the conterminous United States for years 1970 to 2000 for U.S. Forest Service wildfires classified by ecoregion division. (A) Ratio of large to small wildfires. The darker the color, the greater the number of large fires. (B) Fire recurrence interval. The legend goes from dark red to white, representing "high" to "low" hazard (from Malamud and others 2005).

The fire recurrence interval differs markedly between ecoregions. For example, the fire cycle values ranged from 13 years for the Mediterranean Mountains Ecoregion to 203 years for the Warm Continental Ecoregion (fig. 6B). Note the term "fire cycle" does not mean that a fire will occur "every" 13 years, or "every" 203 years. It is a probabilistic hazard. For example, a recurrence interval of 100 years would mean that in ANY year, we have a 1 in 100 chance of a fire of a given size.

Other Studies

In other studies, gradients similar to those observed by Malamud and others (2005) have been described and related to climate and vegetation. Turner and Romme (1994) describe wildfire occurrence gradients as a function of altitude and latitude. They attribute these gradients to broad climatic variation and note western and central regions tend to have frequent fires with forest stand structures dominated by younger trees, whereas the eastern region experiences longer inter-fire intervals and older stand structures. A statistical forecast methodology developed by Westerling and others (2002) exploits these gradients to predict area burned by western United States wildfires, by ecoregion, a season in advance.

Littell et al. (2009) found that climate drivers of synchronous fire differ regionally. They identified four distinct geographic patterns of ecoregion provinces (ecoprovinces) across the West, each associated with a unique set of climate drivers of annual area burned by wildfire. For example, in northern mountain ecoprovinces, dry, warm conditions in the seasons leading up to and including the fire season are associated with increased area burned, suggesting that fuel condition that is dry vs. wet, was the key determinant of regionally synchronous fires. In contrast, in the southwestern dry ecoprovinces, moist conditions the seasons prior to the fire season are more important than warmer temperatures or drought conditions in the year of the fire, suggesting that fuel abundance determined large fire years.

Use of Fire Regime at the Ecoregion Scale

The results of these studies can be used to assess burn probabilities across the nation to identify areas with high risk. This helps government agencies to better plan for wildfire hazards. They can also be used as a baseline from which to assess natural fire regimes, which can be used to abate the threat of fire exclusion and restore fire-adapted ecosystems. In fact, these baseline reference conditions are currently being developed as part of the LANDFIRE project (http://www.landfire.gov) by the United States Forest Service (Missoula Fire Sciences Laboratory), the U.S. Geological Survey (EROS Data Center), and The Nature Conservancy for all biophysical systems across the United States. In addition, an understanding of fire regimes at the ecoregion scale can provide valuable insights important for designing fuel treatments by helping to identify high from low hazard situations.

Finally, what can be done to reduce the risk of fire? Savage (2003) and Allen and others (2002) suggest several principles to guide the implementation of ecologically justifiable restoration projects. Two of the most important principles are:

1. Restoration of natural fire regimes (for example, in southwestern ponderosa pine forests to reduce the widespread risk of crown fires by return to low-intensity surface fire);
2. Pay attention to both structure and process (for example, thinning young trees to reduce the fuel load may not work unless low-intensity surface fires are also reintroduced).

Recent data from the Forest Service reflects the scale of the challenge. Schmidt and others (2002) mapped fire regime condition class (FRCC), which is an ecological metric used by federal agencies, The Nature Conservancy (2004), and others to determine the degree to which the vegetation and fire regimes of a given area have changed compared to reference conditions. As shown on the Schmidt and others (2002) map (fig. 7), fire management has significantly changed the fuel levels of many forests, and concurrently, the frequency and intensity of fire. About 30 percent of all ownerships (except those related to agricultural, barren, and urban land) are in high risk categories (shown in yellow and red). In many ecoregions, this percentage is much higher. For example, in the mountains of the southwest, as much as 83 percent is moderately to severely altered[5].

[5] Finer resolution and more accurate FRCC maps are being produced by the LANDFIRE project. For more information, see http://www.landfire.gov.

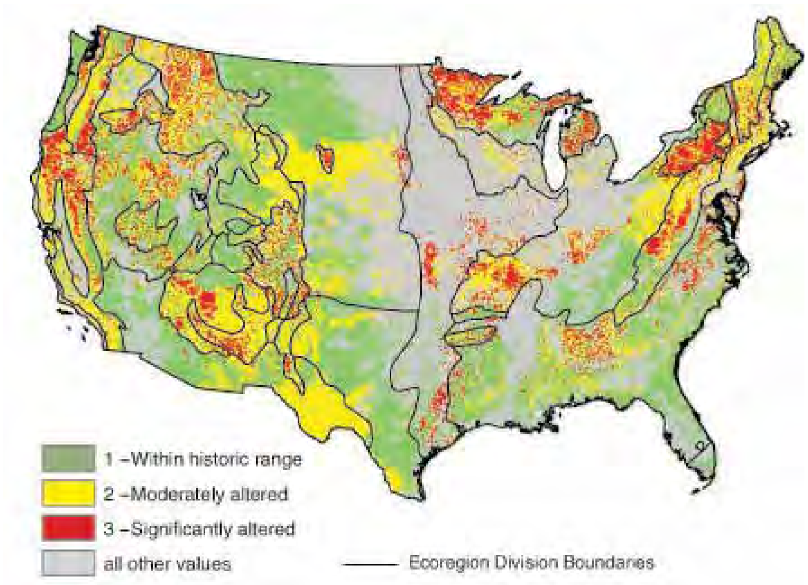

Figure 7. Fire regime condition class (as mapped by Schmidt and others 2002) with ecoregion division boundaries (thick black lines). Green areas (condition class 1) are largely intact and functioning; yellow areas (condition class 2) are moderately altered; red areas (condition class 3) are significantly altered; and gray areas are non-vegetated, agricultural, or urban.

Why Ecoregions Are Needed

The same forest type can occur in different ecoregion divisions. For example, ponderosa pine forests occur in the northern Rockies and the Southwest. This does not imply that the climate, topography, soil, and fire regime are necessarily the same. In the Southwest, the historical fire regime is of frequent, low-intensity surface fires that tend to maintain open, multi-age forests. Farther to the north in the Rockies, cooler conditions mean moister forests in which fires burn less readily. This distinction is important because fire management strategies and restoration protocols are often applicable only to the region in which they were developed. Therefore, management strategies planned to address the fire and fuel issue, such as those documented in the interagency National Fire Plan, should take into consideration ecoregional variation in fire regimes. This 10-year comprehensive strategy can be viewed online at: http://www.fireplan.gov.

Use of Ecosystem Patterns Within Ecoregions

Macroclimate accounts for the largest share of systematic environmental variation at the macroscale or ecoregion level. At the mesoscale level, physiography (geology and landform) modifies the macroclimate and exerts the major control over ecosystem patterns and processes within climatic zones. With this in mind, Bailey and others (1994) used physiographic factors to subdivide the ecoregion provinces of the United States into subregional areas, or sections that have different landform characteristics[6]. These differences are important because the character of the landform with different geology will vary in the climatic zone. In the same climatic zone, different geologies, such as granitic mountains or volcanic plateaus, will weather and erode differently forming different landform relief. Where this occurs, the spread of a disturbance such as wildfire may differ among landforms. Swanson and others (1990) hypothesized that in forested, steep-mountain landforms along the northwest coast of the United States, where landform relief does not exceed several tree heights (for example, Coast Ranges), disturbance agents such as fire and wind can readily move through the forest with little regard for topography. Landforms may have a greater effect on the spread of disturbance and mosaic

[6] Cleland and others (2005) developed another approximation of section boundaries as shown on a revised version of the ecological subregions map.

structure where relief substantially exceeds tree height (for example, Cascade Range). The classification and mapping of physiography, as was done to delineate ecological subregions at the section level, should provide an important means of discriminating broad areas with differing fire regimes within a particular ecoregion.

At finer scales, one finds considerable variation in fire regimes in response to local topography, vegetation, and microclimate (Cleland and others 2004). As we have seen, local ecosystems occur in predictable patterns within a particular ecoregion. Similar fire regimes occur on similar sites within an ecoregion. Knowledge about fire regimes on similar sites allows ecological restoration so as to incorporate the natural variability of fire regimes across the ecoregion.

Future Range of Variation

The range of variation concept is a useful starting point, but it is limited for a number of reasons. First, many systems have been fragmented because of human disturbance. Because of this, fires will not carry the way they did historically. Second, the introduction of non-native species (for example, cheatgrass) has made permanent changes in fire frequencies. Third, fire size and intensity of the past are clearly not acceptable in developed areas. And fourth, system boundaries and fire regimes will change as the climate changes (McKenzie and others 1996). Therefore, only where possible, we need to restore the natural range of variation. We must also determine our feasible alternatives for the "Future Range of Variation."

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Chapter 3.

Fuel Management in Forests of the Inland West

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Introduction

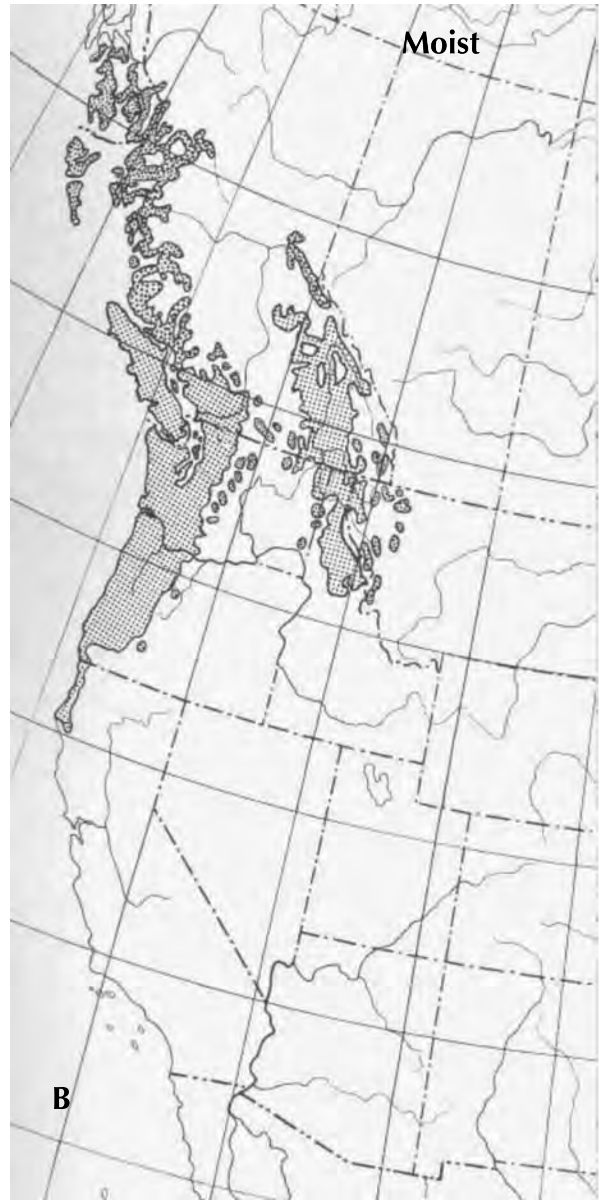
Recent estimates indicate that nearly 40.5 million ha (100 million ac) of forest lands that were historically burned by frequent surface fires in the western United States may benefit from the restoration of surface fire. An additional 4.5 million ha (11 million ac) of forests need to be treated to protect communities from wildfire (Aplet and Wilmer 2003). Rummer and others (2003) estimate that over 26.7 million ha (66 million ac) of forestlands could benefit from fuel reduction. Even with uncertainties in these estimates and arguments as to their precision and accuracy, they clearly illustrate the staggering number of hectares (acres) that need fuel treatments in order to modify fire behavior and burn severity. Access and operability issues further limit the options available on a large portion of western forests. Costs and lack of industrial infrastructure to use small diameter material are other critical factors influencing treatment possibilities. We, the authors, recognize that theoretically, all forests of the western United States could be treated in one way or another to modify wildfire behavior and burn severity. Many of the principles and concepts we discuss are relevant for fuel treatments within other forests and locales; however, we will emphasize forest treatments applicable for use in the cold, dry, and moist forests of the inland western United States. We will discuss forest treatments that influence watershed processes, defined as those that occur when water transports sediment, woody debris, chemicals, heat, flora, or fauna away from a site and deposits it on another site. We define a cumulative effect as one that results from the incremental effects of an event when added to other past, present, and reasonable projected future effects regardless of the triggering action or event (Reid 1988).

Forests of the Inland Western United States

Major river drainages dissect the Rocky, Bitterroot, Salmon, and other mountain ranges along with the Colorado Plateau within the inland western United States. Maritime, continental, and Gulf of Mexico air masses converge and intermingle across this rugged topography with its wide diversity of geologies, soils, and climates that give rise to a plethora of biophysical settings. As the names of the forest classifications infer, what differentiates these forests is their respective differences in climate and biophysical settings and the resulting suite of forest vegetation. Moreover, the biophysical setting, combined with the vegetation composition and structure, result in different disturbances with varying intensities and severities that lead to a variety of interactions and subsequent effects. Fire, grazing, insects, diseases, weather, and timber harvesting, along with vegetation establishment, growth, and succession, interact to create fine (less than 0.1 ha, 0.25 ac) to large (greater than 500 ha, 1,235 ac) mosaics distributed across landscapes. These dynamics occur on some of the most majestic and rugged topography in the world, ranging from less than 300 m (984 ft) above sea level on settings along the Clearwater and Snake Rivers in Idaho to over 3,600 m (11,811 ft) in the mountains of Colorado. There are approximately 147 million ha (363 million ac) of forests within the western United States. About 101 million ha (258 million ac) of these lands are administrated by federal, state, or other public agencies (USDA Forest Service 2001). The forests occupying these lands have a variety of dominant vegetation, ranging from the woodland communities such as chaparral (for example chamise, *Adenostoma fasciculatum*; manzanita, *Arctostaphylos* spp.; *Ceanothus* spp.) in California and pinyon-juniper (*Pinus edulis/Juniperus* spp.) in Arizona, New Mexico, and Utah to forests that can be classified as dry, cold, or moist (Hann and others 1997). Dry forests are typically dominated by ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*), moist forests are dominated by grand fir (*Abies grandis*), western redcedar (*Thuja plicata*), or western hemlock (*Tsuga heterophylla*), and cold forests are dominated by Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), or lodgepole pine (*Pinus contorta*). To refine this broad characterization, potential vegetation type (PVT) is often used. Potential vegetation type is a classification system based on the abundance and presence of the potential vegetation (or sometimes called indicator species) that may grow in a particular area in the absence of disturbance. Since not all vegetation can grow in all places because of limitations in water, soils, etc., this vegetation presence and abundance reflects the physical and biological environment (Daubenmire and Daubenmire 1968; Hann and others 1997; Smith and Arno 1999). Classifications usually offer insight into the type of seral or climax vegetation (ground level and canopy) and the different seral stages that can develop within a particular PVT (Cooper and others 1991; Pfister and others1977). In addition, these classifications provide insight into their expected response to disturbance, how they interact with fire, and the distribution of different species.

Moist Forests

Moist forests of the inland western United States occur in the eastern Cascade Mountains (east of the Cascade Crest in Washington and Oregon) and the northern Rocky Mountains (northeastern Washington and Oregon, northern Idaho, and the western portion of Montana) (fig. 1). They grow at elevations ranging from sea level to 2,300 m (7,550 ft) (Foiles and others 1990; Graham 1990; Hann and others 1997; Packee 1990; Schmidt and Shearer 1990). The topography is usually steep and broken with V-shaped and round-bottomed valleys. In the northern Rocky Mountains and eastern Cascades, total precipitation averages from 500 to 1,520 mm (28 to 60 in) and is influenced by a maritime climate that tends to favor wet winters and dry summers. Most precipitation occurs during November through May with amounts ranging from 508 mm to 2,280 mm (20 to 90 in) (Foiles and others 1990; Graham 1990; Packee 1990; Schmidt and Shearer 1990). Precipitation comes as snow in the Inland West, often accompanied by cloudiness, fog, and high humidity. Rain-on-snow events are common January through March in the northern Rocky Mountains along with a distinct warm and sunny drought period occurring in July and August with rainfall in some places averaging less than 25 mm (1.0 in) per month. Throughout these forests, the soils are quite diverse and can include sedimentary, metamorphic, and igneous parent materials. Soil orders include, but are not limited to, Spodosols, Ultisols, Entisols, Histosols, Inceptisols, and Alfisols. A defining characteristic of the northern Rocky Mountains is the layer of fine-textured decomposed ash (up to 64 cm, 25 in thick) that caps the residual soils. The combination of climate,



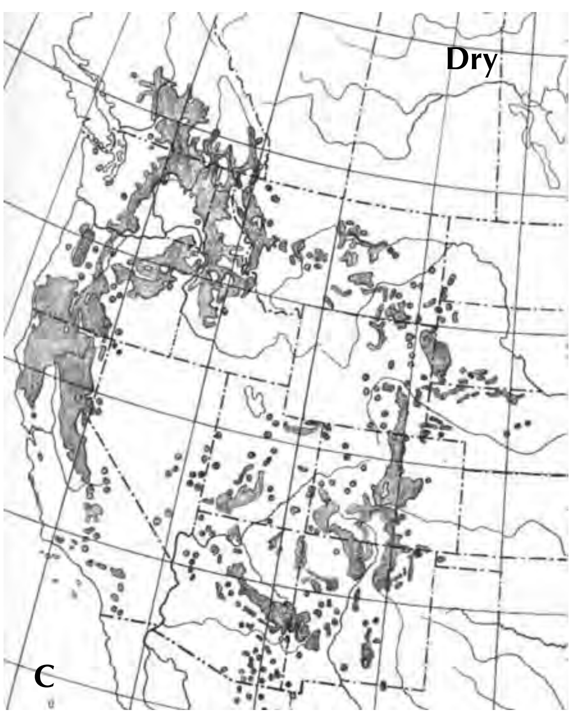


Figure 1. (A) The extent of the cold forests is represented by the range of subalpine fir, (B) the extent of the moist forests is represented by the range of western redcedar, and (C) the extent of the dry forests is represented by the range of ponderosa pine.

topography, parent material, soils, weathering, and ash depth creates the most productive of all forests occurring within the inland western United States.

Vegetation

The vegetation complexes of the moist forests range from early- to late-seral and occur within landscape mosaics possessing all possible combinations of species, seral stages, and structural stages (Cooper and others 1991; Oliver and Larsen 1990). The PVTs in the moist forests of the northern Rocky Mountains include western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), and grand fir (*Abies grandis*). Western white pine (*Pinus monticola*), western larch (*Larix occidentalis*), lodgepole pine (*Pinus contorta*), Douglas-fir, and ponderosa pine occur as the early- and mid-seral species (Daubenmire and Daubenmire 1968; Hann and others 1997). The eastern Cascades and Pacific coast PVTs include western redcedar, western hemlock, grand fir, white fir (*Abies concolor*), Pacific silver fir (*Abies amabilis*), Port-Oxford-cedar (*Chamaecyparis lawsoniana*), incense cedar (*Libocedrus decurrens*), and noble fir (*Abies procera*). On these PVTs, the early- and mid-seral species include lodgepole pine, Douglas-fir, Sitka spruce (*Picea sitchensis*), and ponderosa pine. While less abundant than in the northern Rocky Mountain moist forests, western white pine and western larch do occur as early- and mid-seral species (Franklin and Dyrness 1973; Lillybridge and others 1995). Lush, ground-level vegetation is the norm in the moist forests. The vegetation complexes are similar to those occurring on the west side of the Cascade Mountains and in some Pacific coastal areas. Tall shrubs include vine maple, (*Acer circinatum*), Rocky Mountain maple (*Acer glabrum*), Sitka alder (*Alnus sinuata*), devils club (*Oplopanax horridum*), rose (*Rosa* spp.), gooseberry (*Ribes* spp.), huckleberry (*Vaccinium* spp.), and willow (*Salix* spp.). Forbs include baneberry (*Actaea rubra*), pathfinder (*Adenocaulon bicolor*), wild ginger (*Asarum caudatum*), queencub beadlilly (*Clintonia uniflora*), bunchberry dogwood (*Cornus Canadensis*), and golden thread (*Coptis occidentalis*).

Soil Surface Characteristics and Coarse Woody Debris

Moist forests tend to accumulate large amounts of coarse woody debris (CWD) (fig. 2). Depending on forest age, surface organic layers can contain deep pockets of rotten wood, sometimes representing up to 60 percent of the surface organic horizons (Graham and others 1994; Reinhardt and others 1991). Old forests (greater than 200 years) can have deep (30 cm, 12 in) layers of surface organics and large amounts of CWD ranging between 35 to 72 Mg/ha (15 to 32 tons/ac) (Brown and See 1981; Graham and others 1994).

Cold Forests

Within inland western North America, cold forests generally occur at high elevations (relative to surrounding landscapes) and extend throughout the western United States and into Alberta and British Columbia, Canada (fig. 1). At their northern extent, they occasionally occur at or near sea level and, in southern forests, their range extends to elevations exceeding 3,658 m (12,000 ft) (Lotan and Critchfield 1990). Within the Inland Northwest, they occur primarily in northern Idaho, central Idaho, and the northern Cascades Mountains of Washington (fig. 1). Growing seasons in cold forests are short, ranging from approximately 90 days at low elevations to just a few weeks at the high elevations with frosts occurring any time during the year. These forests are limited by poorly developed soils and, in some areas, by moisture. Nearly all (99 percent) of the cold forests within the inland northwestern United States occur over 1,220 m (4,000 ft), but cold air drainage allows some of them to extend below this elevation (Hann and others 1997; Steele and others 1981). On subalpine fir PVTs, mean annual temperatures range from 3.8 to 4.4 ^{o}C (25 to 40 ^{o}F) with the majority of the precipitation falling as snow and sleet, coming early in the season and staying late. Annual snow fall easily exceeds 1,300 cm (512 in) in the cold forests of the Cascade Mountains with lesser amounts falling where lodgepole pine persists (central Oregon and central Idaho) (Alexander and others 1990). Cold forests have a wide precipitation range with generally less precipitation in southern

Figure 2. A typical mid-aged and mid-seral moist forest containing abundant layers of vegetation and a robust covering on the forest floor consisting of vegetation and coarse woody debris.

cold forests, such as those occurring in New Mexico compared to those in Washington and Idaho. For example, precipitation in the cold forests of the Cascades of Washington ranges from 610 to 2,540 mm (24 to 100 in). Precipitation within the northern Rocky Mountains ranges from 610 to 1,520 mm (24 to 60 in). The central Rocky Mountains receive an average of 610 to 1,400 mm (24 to 55 in) of precipitation and the southern Rocky Mountains receive from 61 to 1,020 mm (24 to 40 in) of precipitation (Alexander and others 1990; Steele and others 1981). Cold forests occurring in southwestern Oregon and California are cool and moist or cold and moist with summer temperatures rarely exceeding 29 °C (85 °F) and winter temperatures rarely dipping below -29 °C (20 °F). Unique to this area is a 4- to 5-month dry spell between April or May through October where precipitation from thunderstorms is rare. Most precipitation occurs during other months, primarily in the form of snow. The snow pack can exceed 4 m (13 ft) in the cold forests of the Sierra Nevada Mountains of California and in the cold forests of southwestern Oregon and northwestern California, up to 2 m (7 ft) of snow often accumulates. In general, the total precipitation of these California and Oregon cold forests ranges from 750 to 1,500 mm (30 to 60 in). The soils supporting the cold forest are relatively young, often shallow, and poorly developed. Mountain glaciers extensively covered these forests during the Pleistocene and have generally been free of ice less than 12,000 years. Most soil parent material is alluvium or glacial tills, but soil surfaces range from very weakly weathered (rocky with no organic layers) to thick soils composed primarily of organic materials. The soil orders common to the cold forests include the Entisols, Inceptisols, Alfisols, and Spodosols (Laacke 1990).

Vegetation

The potential vegetation types dominating the cold forests include subalpine fir (with and without Engelmann spruce) and mountain hemlock (*Tsuga mertensiana*). These

PVTs occur at the highest elevations within the Rocky and Cascade Mountains. Western larch and lodgepole pine are early-seral species in the subalpine fir/Engelmann spruce PVT, Douglas-fir and western white pine are mid-seral species, and western redcedar, grand fir, Engelmann spruce, and subalpine fir are late-seral. The species composition that occurs in the subalpine fir/Engelmann PVT is highly dependent on elevation, associated climate, and disturbance frequency and type. In California, the PVT's dominating the cold forests include red fir, white fir (*Abies concolor*), and noble fir (*Abies procera*), which are also the dominant late-seral species. Lodgepole pine, western white pine, incense-cedar, Brewer spruce (*Picea breweriana*), and Jeffery pine (*Pinus jefferei*) are common early-seral species (Laacke 1990).

Like the moist forests, lush ground-level vegetation is the norm for most settings in the cold forests. Tall shrubs include false huckleberry (*Menziesia ferruginea*) and Sitka alder, and the dominant medium and low shrubs are often huckleberries. Pinegrass (*Calamagrostis rubescens*), bluejoint reedgrass (*Calamagrostis canadensis*), and elk sedge (*Carex geyeri*) typify the graminoids occurring in the cold forests. Some of the most commonly occurring forbs include beargrass (*Xerophyllum tenax*), round-leaved violet (*Viola orbiculata*), and queencup beadlilly (*Clintonia uniflora*) (Cooper and others 1991).

Medium sized shrubs include species such as gooseberry (*Ribes* spp.), pinemat manzanita (*Arctostaphylos nevadensis*), mahala mat (*Ceonothus prstratus*), mountain pride (*Penstemon newberryi*), and mountain whitethorn (*Ceonothus cordulatus*). Areas with deep soils often contain bush chinquapin (*Chrysolepis sempervirens*) or greenleaf manzanita (*Arctosstaphylos partula*). Small upland meadows are common in these forests and provide habitats for a wide variety of sedges, grasses, and forbs (Laacke 1990).

Soil Surface Characteristics and Coarse Woody Debris

In cold forests, such as along the Continental Divide in central Montana, slow decomposition causes needles and other surface litter to accumulate and create dense organic mats from 2 to 8 cm (0.75 to 3.0 in) thick intermixed with large boulders in some areas (fig. 3). Coarse woody debris amounts vary depending on forest age. In mature subalpine forests in Idaho and Montana, CWD can range from 22 to 55 Mg/ha (13 to 25 tons/ac) (Brown and See 1981; Graham and others 1994). In young forests where lodgepole pine dominates the site as an early-seral species, up to 40 percent of the soil surface can be composed of CWD and rotten wood (Brown and See 1981; Harmon and others 1986) (fig. 3).

Dry Forests

Dry forests are typically dominated by ponderosa pine and/or Douglas-fir and can occur throughout the western United States, southern Canada, and northern Mexico (fig. 1) (Little 1971). Their greatest extent is in the inland northwestern United States and in northern California. These forests also occur in the Black Hills of South Dakota and Wyoming, along the Front Range of the Rocky Mountains in Colorado, and along the Mogollon Rim in Arizona, the rugged escarpment that forms the southern limit of the Colorado Plateau. Elevations range from sea level to 3,281 m (10,000 ft) depending on latitude (Oliver and Ryker 1990).

The topography of the dry forests is highly variable and dependent on the region where they occur. East of the Continental divide in central Montana, South Dakota, eastern Wyoming, and central Nebraska, the dry forests tend to occur within discontinuous mountainous regions, plateaus, and canyons intermixed with the plains. Large expanses of dry forests dominated by ponderosa pine occur in the Black Hills of South Dakota and Wyoming, which is largely an isolated mountain range (fig. 4). In the southwestern Rocky Mountains (Colorado, New Mexico, Arizona, Utah), dry forests can occur on flat plateaus to steep mountain slopes and often populate all slope aspects depending on location (Boldt and Van Deusen 1974; Oliver and Ryker 1990).

Figure 3. (A) Mature lodgepole pine growing on a subalpine potential vegetation type typifying much of the cold forests. (B) The inset shows a large accumulation of brown rotten wood that is often present in many cold forests.

Figure 4. The Black Hills of western South Dakota are dominated by late-seral ponderosa pine forests.

Average annual temperatures in dry forests range between 5 and 10 °C (41 and 50 °F), and July and August temperatures average between 17 and 21 °C (62 and 70 °F). The number of frost-free days ranges from 90 to 154 in eastern Montana and South Dakota and to more than 200 days in central California (Oliver and Ryker 1990). Soil moisture influences growth and development of these forests and is highly variable depending on location. In the southwestern United States, Colorado Rockies, Black Hills, and Utah forests, summer rains often occur although little precipitation occurs in May and June. In eastern Oregon and Washington, July through September is usually dry with precipitation amounts in Montana and east of the Continental divide averaging from 280 to 430 mm (11 to 17 in), with approximately half of it occurring from May to August. In the dry forests of eastern Oregon, Washington, and Idaho, precipitation in general averages from 355 to 760 mm (14 to 30 in) with much of it falling in the form of snow. In these forests, July to September are generally dry with precipitation averaging less than 25 mm (1 in). The dry forests growing on the west slope of the Sierra Nevada Mountains in northern California receive on the average 1,750 mm (69 in) of precipitation with very little moisture falling in July and August.

Parent material supporting the dry forests comes from a variety of substrates including igneous, metamorphic, and sedimentary rocks, and they often include quartzite, argillite, schist, shale, basalt, andesite, granite, cinders, pumice, limestone, and sandstone. The soils derived from these materials are most often included in the orders Entisols, Inceptisols, Mollisols, Alfisols, and Ultisols. The vegetation distribution within the dry forests is a function of available moisture, with soil texture and depth influencing the availability.

Vegetation

Ponderosa pine is the primary conifer that defines the dry forests throughout much of the western United States, southern Canada, and northern Mexico (fig.1) (Little 1971). It is the principle species on over 27 million acres and, for every 2.8 ha (7 ac) that it dominates, it is present on an additional 1.4 ha (3.5 ac) (fig. 5). Within the western United States, California alone contains the greatest concentrations of ponderosa pine (2.07 million ha, 5 million ac) closely followed by Oregon (1.9 million ha, 4.7 million ac). When combined, Arizona and New Mexico contain an additional 2.5 million ha (6 million ac) of ponderosa pine (Van Hooser and Keegan 1988). The species occupies sites with elevations ranging from sea level to 3,050 m (10,000 ft) depending on latitude (Oliver and Ryker 1990). In terms of area occupied, it is second only to Douglas-fir.

In the southern and extreme eastern portion of the range, ponderosa pine grows primarily on ponderosa pine PVTs. On these settings, quaking aspen (*Populus tremuloides*) is the most frequent early-seral tree species (Huffman and Alexander 1987; Youngblood and Mauk 1985). Ground-level vegetation includes grasses (for example *Festuca* spp. *Agropyron* spp.) and shrubs such as snowberry (*Symphorcarpus* spp.), spirea (*Spirea* spp.), and russet buffaloberry (*Shepherdia canadensis*) (fig. 5).

With increasing moisture, ponderosa pine occurs as a mid-seral species and Douglas-fir becomes the late-seral species. Quaking aspen and lodgepole pine are early-seral associates of ponderosa pine on these Douglas-fir PVTs (Mauk and Henderson 1984). These ponderosa pine forests occur in the Rocky Mountains along the Front Range of Colorado, in Utah, and in southern Idaho. They also occur along the western slopes of the Sierra Nevada Mountains in California and the eastern slopes of the Cascades in Oregon (Franklin and Fites-Kaufman 1996; Hann and others 1997). On these settings, ground-level vegetation includes ninebark, elk sedge, and pine grass. These species, in particular, exemplify aggressive survivors after disturbance (for example, fire, mechanical site preparation) and can colonize a site quickly after disturbance (Baumgartner and others 1986).

In several locales, dry grand fir and/or white fir PVTs represent the dry forests (Hann and others 1997). On such settings, ponderosa pine and Douglas-fir occur but are succeeded by late-seral grand fir and/or white fir in the absence of disturbance (Bradley and others 1992a) (fig. 6). Additional trees that can occur in such forests include juniper,

Figure 5. Ponderosa pine growing in central Oregon. (A) Note the open understories and high crown base heights of these mature trees and the forest floor covered by a layer of needles. This needle layer does not readily decompose and provides a highly flammable fuel. (B) The inset shows lush ground-level vegetation developing after a ponderosa pine timber harvest on a Douglas-fir potential vegetation type in central Idaho.

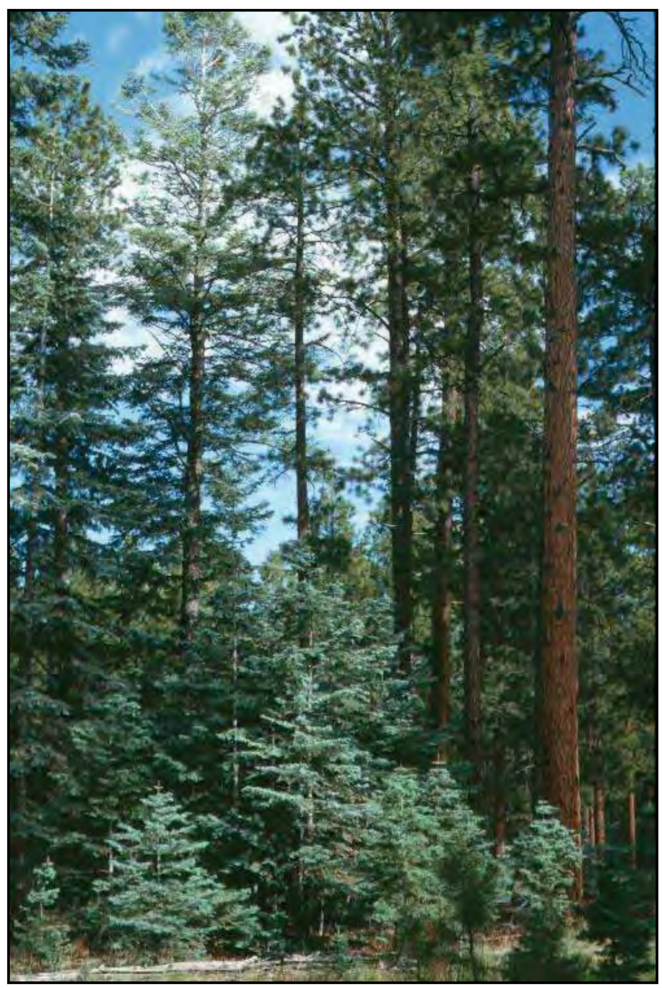


Figure 6. On the Douglas fir and grand fir potential vegetation types, ponderosa pine is readily succeeded by these species in the absence of disturbance (for example fire, harvesting). This succession results in abundant fuel layers and nutrient rich crowns that extend to the soil surface making them highly susceptible to loss during fires.

pinyon pine, sugar pine (*Pinus lambertiana*), incense-cedar (*Libocedrus decurrens*), western larch, Jeffrey pine, and lodgepole pine. Pine grass and ninebark (*Physocarpus malvaceus*) are frequent associates, but tall shrubs such as Rocky Mountain maple often occur.

Soil Surface Characteristics and Coarse Woody Debris

In the dry forests, needle shed from long needle pines is a major component of the surface litter (fig. 5). Without fire, these layers can quickly accumulate, often minimizing the amount of grasses, forbs, and shrubs that were more prevalent on these sites when fire maintained smaller amounts of these litter layers. Coarse woody debris (CWD) is relatively scarce when compared to moist and cold forests (figs. 2 and 5). However, the more productive dry forests tend to accumulate relatively high amounts of CWD, especially in the absence of fire. For example, on grand fir and Douglas-fir PVTs in Montana and Idaho, CWD ranges from 10 to 30 Mg/ha (5 to 20 tons/ac). In contrast, on ponderosa pine PVTs in Arizona, CWD can range from 10 to 20 Mg/ha (5 to 10 tons/ac) but can quickly accumulate where white fir are the late-seral species, as exemplified by the 10 to 30 Mg/ha (5 to 15 tons/acre) reported by Graham and others (1994) for these PVTs.

Forest Change

Most forest conditions of the western United States have changed from those that occurred historically (pre-1900). In particular, many of these changes relate to both the decrease in fire occurrence and the increase in the size and severity of how the wildfires burned. The changes in how fires burn, often because of successful fire exclusion, combined with timber harvesting and climate cycles resulted in changes in the abundance, composition, and distribution of tree, ground level vegetation, and forest floor components within the three-forest types (Graham and others 2004; Jain and Graham 2005). These changes include shifts in species composition, volume, and density, thus creating situations such as altering vegetative chemistry and the biological and chemical composition of the forest floor (Harvey and others 1999a) (fig. 6).

For the most part, western forests of the late 1800s and the first half of the 20th century produced timber crops for expanding local and national economies. Thus, the forests were aggressively protected from damaging animals, insects, diseases, and fire (Graham and Jain 2004). The species harvested included western white pine, western redcedar, western larch, ponderosa pine, and Douglas-fir that produced diverse products such as toothpicks, matches, ship masts, railroad ties, fuel, and building material. At this time, many tree and shrub species were considered weeds (for example, western hemlock and grand fir) and were removed (slashed and burned) to provide growing space for product producing species. In many locales throughout the western United States, this product centric forest management continued into the 1950s and early 1980s (Davis 1942; Graham and others 2005; Steen1976).

Two large fire events, the Peshtigo fires that burned in northern Wisconsin in October of 1871 and the wildfires that burned in northern Idaho, eastern Washington, and western Montana in August of 1910, burned millions of forested hectares (acres) and towns and caused loss of life (Gess and Lutz 2002; Pyne 2001) (fig. 7). These destructive events were the catalyst that cemented the mission of the newly formed Forest Service to protect the valuable forestlands of the United States from damaging agents (Steen 1976). The result was a significant decrease in the number of fires burning and the amount of area burned in western forests. The effectiveness of fire exclusion increased dramatically especially with the use of airplanes and smoke jumpers to aerially combat fires and the development of access roads (Graham and others 2004; Graham and Jain 2004). Although fire exclusion and timber harvesting influenced the moist and cold forests, the dry forests had the most noticeable changes (Covington and Moore 1994; Hann and others 1997).

Figure 7. The wildfires that burned in the northern Rocky Mountains in 1910 consumed over 1. 2 million ha (3 million acres) and burned communities, reinforcing the need to protect forests from disturbances and fire in particular.

Since the dry years of the 1930s, the climate of the western United States has been relatively moist, facilitating the regeneration and development of large amounts of forest vegetation (Haig and others 1941; Pearson 1950). Historically, native insects (for example, pine beetle [*Dendroctonus* spp.]), diseases (for example, root rots [*Armillaria* spp.]), and mistletoe (*Arceuthobium* spp.) infected and killed the very old or stressed individuals, which tended to diversify vegetation communities (Hessburg and others 1994). However, in present forests, changes in vegetation have facilitated development of unprecedented epidemic levels of these insects and diseases in many locales (fig. 8). These disturbance agents were often encouraged by weather events such as ice storms, windstorms, and periodic droughts. In addition, the movement of Europeans into the

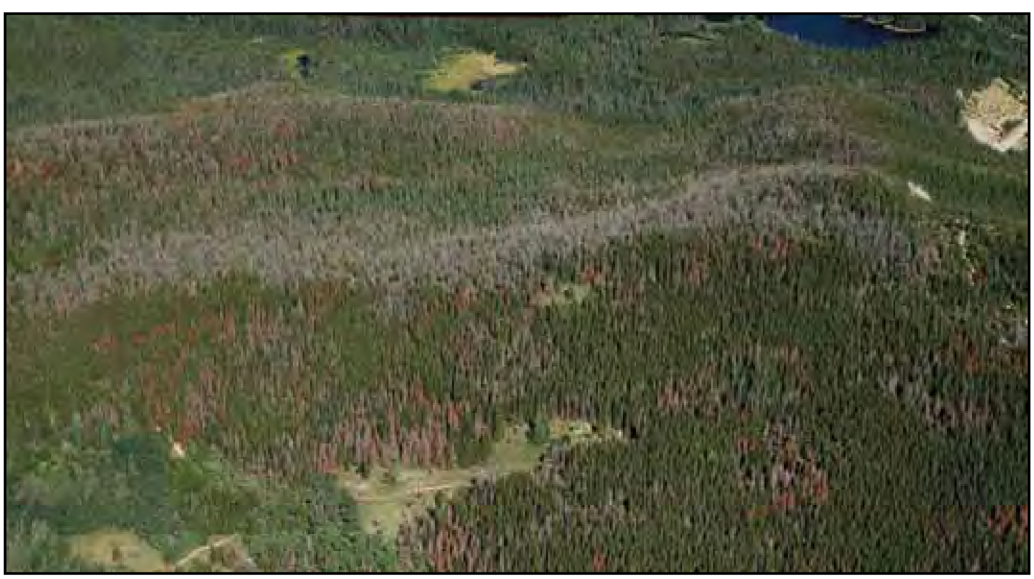


Figure 8. Lodgepole pine killed by mountain pine beetle in Colorado (photo by William M. Ciesla, Forest Health Management International, Bugwood.org).

western United States resulted in the intentional and inadvertent introduction of both exotic plant and animal species that often displaced native species. Non-native insects and diseases often found no natural checks or balances, enabling them to invade western forest ecosystems unimpeded. In concert, these and allied events have facilitated the development of abundant amounts of forest vegetation and, in some settings, a different suite of vegetation, insects, and diseases than those that occurred prior to 1900.

Fire Regimes

Fires, in concert with other disturbances, create mosaics of forest composition and structure within and among stands and across landscapes. These mosaics can occur over relatively fine spatial scales (in other words, less than 0.5 ha, 1.2 ac) to rather large mosaics exceeding hundreds of hectares (acres). Wildfires historically burned the cold, moist, and dry forests at various intensities and frequencies giving rise to a wide variety of burn severities and producing a variety of vegetative successional pathways and a diversity of vegetation and forest floor conditions. A fire regime is a generalized description of the role fire plays in a forest and is related to a fire's frequency, severity, and intensity (Agee 1993). While multiple fire regime classifications are available, we find that for describing fires occurring in the dry, cold, and moist forests, a system containing three classes—stand replacing fires (lethal), mixed fires, and low intensity surface fires (non-lethal)—is most useful (Hann and others 1997; Schmidt and others 2002).

The most extreme fire regime, the stand replacing (lethal) regime, kills all canopy layers across stands or relatively large areas (in other words, areas larger than ~2 ha, 5 ac). In general, these fires kill the standing vegetation and could be classified as intense (all black stems), but some moderately severe (brown needles present) burning may exist in numerous areas. Within forests containing burned and blackened trees, the level of soil burn severity can be highly variable, ranging from light to highly severe depending on the state of the ground-level vegetation, surface fuels, forest floor, and burning conditions during the fire.

Low intensity surface fire (non-lethal) regimes clean the forest floor of vegetation and accumulated woody debris but yet leave the majority of the high forest cover alive. These non-lethal fires rarely kill dominant trees, but soil burn severity can range from nothing to highly severe depending on the amount, condition, and extent of the surface fuels and the heat and residence time of the fire.

Mixed severity fires combine lethal and non-lethal fires, killing small to medium sized groups of dominant trees, burning surface fuels in other areas while leaving the dominant trees alive. Mixed fire spatial extents can range from small patches of vegetation to rather large landscapes that burned for days to weeks. By their nature, fires described by this regime leave compositionally and structurally fine scale mosaics of forest vegetation and simple to complex burn patterns. With the advent of successful fire exclusion, these mixed severity fires were frequently extinguished preventing them from modifying many forested landscapes. Historically, all three fire regimes occurred in all three forest types allowing fire suppression activities, to some degree, to impact all three forest types.

Moist Forest Changes

Fire exclusion, insects, diseases, and weather interacted with climate change (climate cycles) to alter the moist forests from one dominated by early-and mid-seral species to one dominated by mid-to late-seral species. Historically, wildfires burning the moist forests were highly variable. Estimates suggest that nonlethal surface fires occurred at relatively frequent intervals (15 to 25 years) within 25 percent of the moist forests. Lethal crown fires burned about 25 percent of the moist forests every 20 to 150 years but occasionally extended to 300-year intervals. The mixed fire regime burned across about 50 percent of the moist forests at 20- to 150-year intervals with some lethal events occurring at 300-year intervals. Fires typically started burning in July and were usually out by early September with the change in weather. Most fires were small, but occasionally,

Figure 9. Blister rust, an introduced Eurasian disease, significantly reduced the abundance of western white pine from much of its historical range.

large fires did occur with 74 percent of fires killing a portion of the canopy (Hann and others 1997; Pyne 2001) (fig. 7). Because of fire exclusion, surface fires now burn 10 percent of the moist forests, mixed fires burn 30 percent, and crown fires burn 60 percent of the moist forests. Although fire exclusion played a role in altering the moist forests in the western United States, introduction of a European stem rust, white pine blister rust (*Cronartium ribicola*), was a greater factor in causing change in these forests (fig. 9) (Jain and Graham 2005). The disease attacked and killed western white pine, a primary commercial species. Upon introduction of blister rust, a massive effort was initiated to control the disease by removing currant (*Ribes* spp.) bushes (the alternate host). During the Civilian Conservation Corp days of the 1930s, workers pulled, sprayed, and grubbed *Ribes* bushes throughout the northern Rocky Mountains. Hutchison and Winters (1942) described the effort "like bailing the ocean with a teacup" and efforts to control the disease proved futile. By 1968, the western white pine blister rust program and management of the species was "realigned," resulting in the accelerated removal of naturally occurring western white pine before they supposedly succumbed to the rust (Ketcham and others 1968).

Harvesting of ponderosa pine and western larch, other high value timber species, exacerbated the forest changes resulting from the introduction of blister rust. The partial and intermittent canopy removal and minimal soil surface disturbance caused by both tree harvest and western white pine mortality were ideal situations for grand fir and western hemlock to establish and aggressively encroach. In the eastern Cascades, the effect of blister rust on forest composition and structure was less severe since western white pine was not as dominant, thus fire exclusion and harvesting were more important change agents in these forests. Grand fir and Douglas-fir readily filled the niches western white pine, ponderosa pine, and western larch once held.

Since 1991 native insect and pathogen activity in the moist forests far exceed those of the past. The Douglas-fir beetle (*Dendroctonus pseudotsugae*), mountain pine beetle (*Dendroctonus ponderosae*), spruce budworm (*Choristoneura occidentalis*), and tussock moth (*Orgyia pseudotsugata*) that were historically endemic are now often epidemic. Similarly, the root diseases *Armillaria* spp. and *Phellinus weirii* that were historically endemic are now common in the current fir-dominated forests (Hann and others 1997; Hessburg and others 1994).

Weather, another formidable disturbance, in the form of snow or wind, often creates a variety of canopy openings ranging from gaps to large openings (~16 ha, 40 ac). In early-seral dominated species (ponderosa pine, western larch, western white pine) forests, snow will often slip from the trees, minimizing breakage, while other species in the intermediate crown classes (grand fir, Douglas-fir) will break, creating gaps and openings, decreasing forest densities, and altering species composition. Today, species that dominate many sites, such as grand fir and Douglas-fir, have dense crowns that hold snow and tend to break more readily compared to western larch or western white pine, which shed the snow (Graham and Jain 2005; Jain and Graham 2005).

Disturbances in the moist forests, singularly or in combination, or the lack thereof, have culminated in the current distribution of successional-stage, forest structure, species composition, and disturbance regimes that differ from historical conditions (1850 to 1900) (Hann and others 1997). In some settings, the mixed-fire regime maintained closed canopy conditions that allowed the mid-seral stage to develop into late-seral, multi-story stages (Hann and others 1997). The late-seral, multi-story structure, which typically developed in cool, moist bottoms and basins, has decreased by about half in the last century. The early-seral, single-story stands that once occupied an estimated 25 to 30 percent of the area now occupy only 9 to 10 percent, except in the northern Cascades (Washington) where they increased in abundance. The mid-seral stages have generally increased in abundance in the northern Rocky Mountains and, to a lesser degree, in the eastern Cascades.

Because of fire, weather, disease, and insect interactions, species composition has shifted in the moist forests of the northern Rocky Mountains (Fins and others 2001; Hann and others 1997; Neuenschwander and others 1999). For example, before 1900, western white pine (early-to mid-seral species) dominated many settings, often representing 15 to 80 percent of the trees within stands (Fins and others 2001). Western larch and ponderosa pine also occurred in the early-and mid-seral structures but declined along with western white pine and were succeeded by grand fir, Douglas-fir, and western hemlock (Atkins and others 1999; Hann and others 1997). The eastern Cascades had limited amounts of western white pine and western larch; therefore, ponderosa pine, lodgepole pine, and Douglas-fir played more of a role in occupying the early-to mid-seral successional stages.

Western white pine, ponderosa pine, and western larch-dominated forests are generally tall and self-pruning, even in moderately dense stands. They have large branches high in the crowns and the base of the crowns is well above surface fuels. In general, this crown architecture protects the nutrients stored in the canopy from surface fires. In contrast, young- to mid-aged (less than 150 years) western hemlock and grand fir/white fir generally do not self prune. Forests dominated by grand fir tend to concentrate both nitrogen and potassium in their foliage, which often extends to the soil surface. In general, low canopy structure, combined with nutrient and microbial activities concentrated in or near the soil surface, makes these two critical ecological resources susceptible to mechanical and fire destruction (Harvey and others 1999a; Mika and Moore 1990; Minore 1979; Moore and others 1991).

A positive change in the moist forests is the introduction of western white pines that are resistant to blister rust (Bingham 1983). Tree improvement programs that began in earnest in the 1950s started yielding rust-resistant seedlings for use in reforestation. By 2005, millions of rust resistant seedlings have been planted throughout the moist forests. In addition, some native resistance to the disease remained in non-harvested residual western white pines. As a result, western white pine has many opportunities to play both significant ecological and commercial roles in the moist forests (Graham and Jain 2005).

Cold Forest Changes

Depending on the physical setting, estimates suggest that the cold forests of the inland northwestern United States historically (1850 to 1900) burned in the range of 25 to 100 year intervals. Non-lethal surface fires burned approximately 10 percent of these forests, every 30 to 100 years. Lethal crown fires burned 25 to 30 percent of the cold forests every 30 to 100 years with the longer intervals occurring in moist areas. During the short fire season (~60 days), a mixed fire regime burned about 60 percent of the cold forests at 25- to 100-year intervals, with occasional large fires occurring every 100 years (Hann and others 1997). It is estimated that the proportions of the cold forests burned by non-lethal (10 percent), mixed fires (60 percent) and lethal fires (30 percent) is similar to the proportions historically burned by the different fire regimes. Therefore, fire exclusion has had minimal influence on these forests. However, local impacts on forest composition and structure are very likely the result of fire exclusion, as are the locations where the different fire regimes likely burned. In turn, the vegetative mosaic and its texture (size, shape, and location of vegetation patches) most likely differ from the historical characteristics of these forests (Bradley and others 1992a).

In the subalpine fir PVT, fire maintained 23 percent of the cold forest in early-seral vegetation primarily dominated by lodgepole pine. In these areas, lethal fires created ideal conditions (for example, ~10 acre openings and larger, burned-over and mineral soil surfaces) for the regeneration of lodgepole pine. Extremely dense stands of lodgepole pine would develop and, without subsequent disturbance, dominate settings for decades. Similarly, in some of the drier and colder portions of the cold forests, a single canopy of lodgepole pine could persist for over a hundred years.

Historically, mid-seral structures occupied about 53 percent of the cold forests (Hann and others 1997). A mixed fire regime in these forests, along with periodic wind, floods, snow, and other small-scale disturbances, allowed uneven-aged and patchy stands to develop. Engelmann spruce, western redcedar, grand fir, mountain hemlock, and subalpine fir would readily regenerate in a variety of canopy conditions (gaps), giving rise to dense stands with many canopy layers. In the absence of lethal fires, the warmer forest settings containing lodgepole pine were readily succeeded by Engelmann spruce and subalpine fir.

Prior to 1900, late seral, multi-storied forests occupied 15 percent of the cold forests (Hann and others 1997), and subalpine fir and Engelmann spruce dominated these forests with lodgepole pine and Douglas-fir as intermittent associates. Forest densities were often high (exceeding 29,000 trees/ha, 11,733 trees/ac) and dominated by subalpine fir and Engelmann spruce but, occasionally, a few large, fire resistant Douglas-firs would be interspersed throughout these multi-storied forests (Franklin and Mitchell 1967). Non-lethal surface fires would encourage single canopies of subalpine fir, Engelmann spruce, or lodgepole pine to develop over approximately 8 percent of the cold forests.

Due mainly to harvesting, and to a lesser degree lethal fires, the greatest vegetative changes occurring in cold forests over the last 100 years is the increase in the extent of early-seral structures consisting primarily of lodgepole pine. Large (100s to 1,000s of ha) expanses of lodgepole pine forests, especially those with large (greater than 20-cm, 8-in) diameters, are often killed by mountain pine beetle (fig. 8). Also Douglas-fir beetle, spruce budworm, tussock moth, and spruce bark beetle (*Dendroctonus rufipennes*) appear to be defoliating and killing trees more readily in the cold forests than they did historically. Singly, and in combination, the defoliation and mortality caused by these insects can provide large amounts of dead fuel for future fire events (Hann and others 1997) (fig. 8).

Dry Forest Changes

In the western United States, domestic livestock grazing and harvesting of ponderosa pine forests was occurring by the mid 1800s (Barrett 1979; Cooper 1960; Pearson 1950; Rasmussen 1941;Van Hooser and Keegan 1988). In mesic forests, grand fir and/or white fir and Douglas-fir rapidly colonized these sites when ponderosa pine was harvested. Grass cover tended to decrease ponderosa pine seedling establishment and survival, especially on the ponderosa pine PVT (Brawn and Balda 1988). However, in the early 1900s in the southwestern United States when heavy livestock grazing ceased (thus eliminating seedling damage), abundant ponderosa pine seedlings became established. Because of fire exclusion, climate changes, and other factors, these trees readily developed into dense stands (Covington and Moore 1994; Pearson 1950; Stein 1988).

Before successful fire exclusion, temperature and precipitation patterns combined with natural and human ignitions that allowed fires to burn the dry forests at relatively frequent (for example, less than 40 years) intervals. Cultural burning by Native Americans augmented and even dominated burning in several locations (Barrett and Arno 1982; Stewart 1951). In the northern Rocky Mountains of Idaho and western Montana, dry settings (ponderosa pine and/or Douglas-fir PVTs) were historically burned by non-lethal wildfires at 15- to 23-year mean return intervals. These fires could be quite large often burning for weeks to months (Weaver 1943). Mixed fires frequently burned mesic forests containing ponderosa pine (grand fir and/or Douglas-fir PVTs) at mean return intervals extending to over 60 years. Non-lethal fires dominated the central and southern Rockies (ponderosa pine and/or Douglas-fir PVTs), although mixed severity fires also occurred, especially along the Front Range of the Rocky Mountains in Colorado (Bradley and others 1992a; 1992b; Fulé and others 1997; Kaufmann and others 2001). Fires tended to be few on the driest settings (ponderosa pine and/or woodlands) because of discontinuous surface fuels (Bradley and others 1992a). In contrast to other locales dominated by late-seral ponderosa pine, the forests of the Black Hills of Wyoming and South Dakota possibly experienced greater extents of lethal fires because of the abundant ponderosa pine regeneration that normally occurred (Shepperd and Battaglia 2002; Shinnen and Baker 1997). Nevertheless, historical wildfires likely burned through most ponderosa pine forests leaving in their wake a wide variety of species compositions and vegetative structures arranged in a variety of mosaics.

Within the dry forests, dense conditions often develop and are exacerbated by fire exclusion, increasing the abundance of insect and disease epidemics, which significantly altered the composition, and structure of these forests (Harvey and others 2000). Historically, western pine beetle (*Dendroctonus brevicomis*), pine engraver (*Ips* spp.), fir engraver (*Scolytus ventralis*), and Douglas-fir tussock moth were insects associated with regularly burned areas (Hessburg and others 1994). In most years, bark beetles occurred at endemic levels in ponderosa pine, Douglas-fir, and grand fir, killing large weakened trees that were struck by lightning, infected by root disease (*Armellaria* spp.), or too old to resist attack (Williams and others 1986; Wu and others 1996). Pine engraver and fir engraver beetles attacked young, densely stocked ponderosa pine, killing trees scorched by low-intensity surface fires and severely infected trees containing root rot or dwarf mistletoe.

Because of forest change, these same insects have reached epidemic levels in many forests (Gardner and others 1997; Hedden and others 1981). Today, ponderosa pine continues to be susceptible to the western pine beetle, and the mountain pine beetle is prevalent on Douglas-fir and grand fir PVTs. The pine engraver beetle is more abundant and destructive now with some of the severest outbreaks occurring on low-elevation ponderosa pine PVTs (Hessburg and others 1994). Pandora moth (*Coloradia pondora*) defoliates ponderosa pine and scattered outbreaks have occurred in Arizona, California, Colorado, and Oregon during the 20th century (Speer and others 2001). Dense stands of Douglas-fir and grand fir that developed on many grand fir and Douglas-fir PVTs are very susceptible to both defoliators and root diseases.

Harvesting western larch and ponderosa pine precipitated the regeneration and growth of grand fir/white fir and Douglas-fir in the dry forests, which subsequently facilitated the accumulation of both above-and below-ground biomass and associate nutrients close to the soil surface (Harvey and others 1986) (fig. 6). As a result, even low-intensity surface fires often consume the surface organic layers, killing tree cambiums and/or fine roots, volatilizing nutrients, killing trees, and increasing soil erosion potential (Debano 1991; Hungerford and others 1991; Robichaud and others 2000; Ryan and Amman 1996). In addition, the abundant fir in the understory creates nutrient-rich ladder fuels that facilitate crown-fire initiation, increasing the likelihood of nutrient loss (Harvey and others 1999a; Minore 1979; Van Wagner 1977). The risk of nutrient loss is greater on infertile sites because dense stands of late-seral species are more demanding of nutrients and water than the historical stands dominated by widely spaced early-seral species (Harvey and others 1999a; Minore 1979) (see Chapter 9).

With the advent of fire exclusion, animal grazing, timber harvest, and climate cycles, on the moist potential vegetation types (for example grand/white fir), ponderosa pine is being succeeded by Douglas-fir, grand fir, and/or white fir (fig. 6) (Arno and others 1997; Graham and others 2004; Gruell and others 1982). Fire intolerant vegetation, dense forest canopies, and homogenous and continuous horizontal and vertical structures are developing, thus creating forests favoring crown fires rather than low intensity surface fires that historically occurred in these forests (Arno and Brown 1991; Dodge 1972; Peterson and others 2005; Van Wagner 1977). Within the Inland Northwest, the extent of mid-seral (for example, Douglas-fir) vegetation has increased by nearly 3.2 million ha (8 million ac) and the extent of single storied mature vegetation (for example, ponderosa pine) has decreased by over 1.6 million ha (4 million ac) (Hann and others 1997). Another way to view these changes is that the successional processes (movement from one successional stage to another) in some locations have been shortened by a factor of at least 10. For example, ponderosa pine may or may not be succeeded by Douglas-fir in 300 to 400 years within forests historically burned by frequent fires, but in many locations Douglas-fir has succeeded ponderosa pine in less than 50 years (fig. 6) (Hann and others 1997; Harvey and others 1999a; Smith and Arno 1999).

Forest Floor Changes

The shift in species composition from western white pine, western larch, and/or ponderosa pine to Douglas-fir, grand fir/white fir, and/or western hemlock dominated forests (including the shrub and forb components) has changed litter (soil surface) type and quantity from that which occurred historically. In addition, the accumulation of both above- and below-ground biomass from roots, needles, and boles in fir forests is accelerating activities of decomposers by increasing and changing the basic substrate they use (Harvey 1994). Associated with these changes in litter type and quantity is a likely change in ectomycorrhizal relationships and soil surface chemistry, including allelopathic substances, with the potential to alter a variety of microbial activities (Rose and others 1983). In addition, fire exclusion, timber harvesting, and animal grazing have exacerbated these forest floor alterations in many locales singly, or in combination, by soil compaction and displacement. For example, decomposed true firs create white rotten wood, which rapidly disperses into the soil and is quickly consumed by decomposers. In contrast, decomposed ponderosa pine, western white pine, and western larch create brown rotten wood, which can persist in soil for centuries and in this condition, can retain nutrients and hold water (Harvey and others 1988; Larsen and others 1980). Western larch and ponderosa pine tend to be deep-rooted, in contrast to the relatively shallow-rooted western hemlock and grand fir, which have abundant feeder roots and ectomycorrhizae in the shallow soil organic layers (Harvey and others 1987; Minore 1979). The soil microbial activities in fir-dominated forests compared to pine-dominated forests may diminish the post-fire acquisition and cycling of nutrients (Neary and others 1999). Moreover, these changes in soil microbial activities may increase the likelihood of uncoupling any continuity between current and preceding vegetative communities (Amarnathus and Perry 1994).

In the dry forests, biological decomposition is more limited than biological production. When fire return intervals reflected historical fire frequencies, the accumulation of thick organic layers was minimized and nutrient storage and nutrients were dispersed in the mineral soils (Harvey and others 1999b). In the absence of fire, bark slough, needles, twigs, and small branches can accumulate on the forest floor and when these layers are continuously moist, ectomycorrhizae and fine roots of all species tend to concentrate in the surface mineral soil and thick organic layers, making them vulnerable to disturbances (Harvey and others 1994). In addition, historical ponderosa pine forests were likely well matched to soil resources, relatively resistant to detrimental fire effects, well adapted to wide ranges of site and short-term climate variation, subject to modest (largely beneficial) insect and pathogen mortality, and could be considered long-lived and relatively stable. In contrast, forests that were dominated by ponderosa pine and are now dominated by Douglas-fir, grand fir, or white fir are probably not well matched to soil resources and are also not likely resistant to the wide range of site and climate variations found within the dry forests (fig. 6). In turn, they are often subject to high insect and pathogen mortality and are no longer considered either long-lived or stable (Harvey and others 1999a).

Forests as Fuel

Fire behavior and burn severity depend on the properties of the various fuel (live and dead vegetation and detritus) strata and the continuity of those fuel strata, horizontally as well as vertically. The fire hazard for any particular forest stand or landscape relates to its potential for the fuels to cause specific types of fire behavior and effects. Understanding the structure of fuelbeds and their role in the initiation and propagation of fire is the key to developing effective fuel management strategies. Fuelbeds are classified in six strata:

1. tree canopy,
2. shrubs/small trees,
3. low vegetation,
4. woody fuels,
5. moss, lichens, and litter, and
6. ground fuels (in other words, humus, fermentation layer, surface and partially buried rotted wood, etc. (Sandberg and others 2001) (fig. 10).

Modification of any fuel stratum has implications for fire behavior, suppression, and burn severity.

Ground Fuels

Ground fuels consist of duff (organic soil horizons), roots, and buried woody material (fig.10) (Sandberg and others 2001). Often, needle fall and bark slough will accumulate at the base of trees and eventually create deep organic layers in which fine roots and ectomycorrhizae of trees and ground level vegetation may accumulate (Graham and others 2000). Ground fuels typically burn by smoldering and may burn for many hours, days, or even weeks if initial moisture contents are high (Frandsen 1991; Hungerford and others 1991) (fig. 11). This long duration smoldering can often lead to soil damage, tree mortality (high severity), and smoke (Ryan and Noste 1983; Ryan and Reinhardt 1988; Wells and others 1979). Rotten material on the ground surface is particularly ignitable by firebrands (small twig segments or bark flakes supporting glowing combustion) falling ahead of an advancing fire front and increases the success of spotting.

Surface Fuels and Ladder Fuels

Surface fuels consist of grasses, shrubs, litter, and woody material lying on, or in contact with, the ground surface (fig.10) (Sandberg and others 2001). The bulk density (weight within a given volume) of surface fuels and size class distribution of fine fuels (sticks less than 7.62 cm (3.0 in) are critical to frontal surface fire behavior (spread rate and intensity) compared to fuel loading (weight per unit area) alone. Other characteristics of surface fuels that determine surface fire behavior are fuel depth, continuity, and chemistry. Surface fires burn in both flaming and postfrontal (smoldering or glowing)

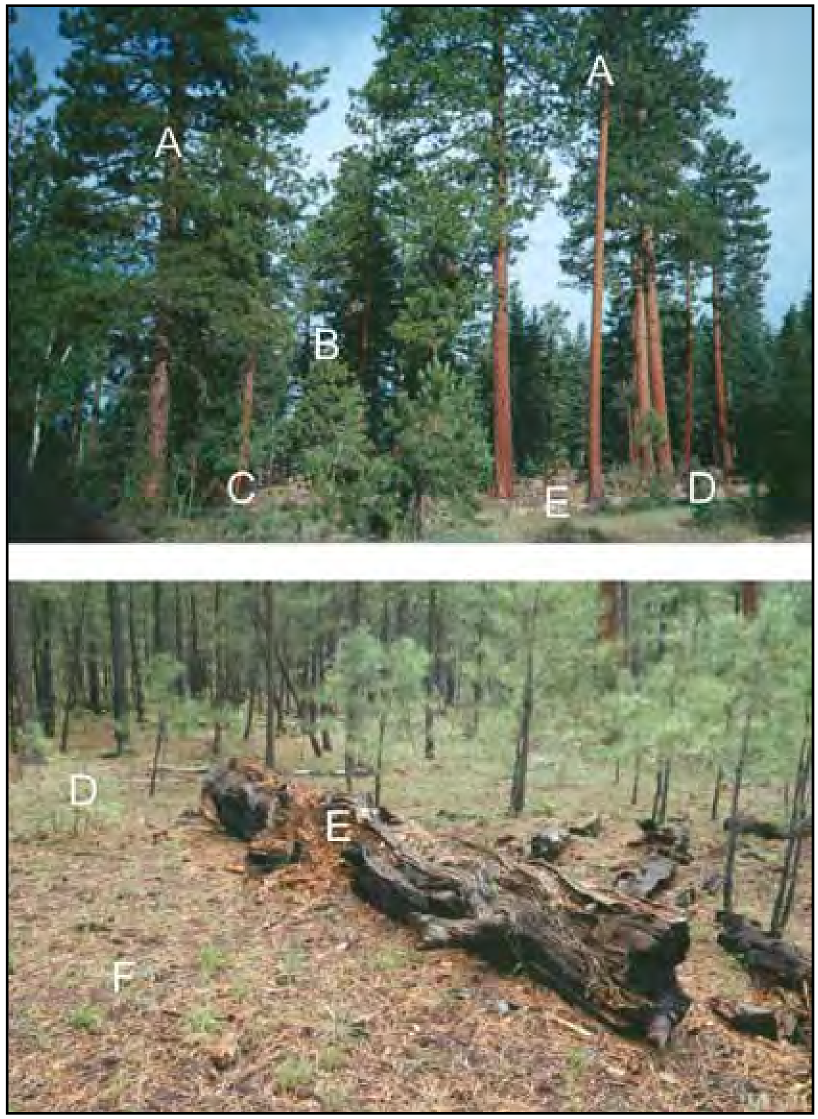


Figure 10. Fuelbed strata have different implications for combustion environment, fire propagation and spread, and fire effects. (A) The canopy, (B) ladder fuels and (C) shrub layers contribute to crown fires. (D) Low vegetation, (E) woody fuel, and (F) ground fuel contribute to surface fires. (E) Woody fuel and (F) ground fuels are most often associated with smoldering fires and residual combustion that can transfer large amounts of heat deep into the soil (Graham and others 2004; Sandburg and others 2001).

Figure 11. Large amounts of smoke can be produced from smoldering ground fuels after flames have subsided and large amounts of heat can be transferred to the soil.

Figure 12. Large amounts of fine fuels can be created when reducing canopy fuels. Without treatment, these fuels can increase the risk of unwanted wildfires and increase burn severity to both the high forest canopy and forest floor during prescribed fires and wildfires. Mechanical fuel treatments (for example piling and mastication) are often used to treat these fuel conditions.

phases (fig. 11). High-energy release rates occur during the relatively short flaming phase when fine fuels are consumed. Low energy release rates occur over longer periods by smoldering and glowing phases that consume larger (greater than 7.62 cm, 3 in diameter) fuels. Surface fuel complexes with high loadings of large material, such as slash left after timber harvesting or precommercial thinning operations (fig. 12), have long flaming residence times compared to fine fuels such as needles, shrubs, or grasses (fig. 5). High surface fire intensity usually increases the likelihood for igniting overstory canopy fuels, but surface fires with long residence times can contribute to drying aerial fuels that can lead to torching (when a tree or group of trees' foliage ignites and flares up, usually from bottom to top) (Alexander 1988).

Even in the dry forests, shrub and small tree regeneration can be abundant and frequent creating dense and robust layers of vegetation covering the forest floor (Pearson 1950) (fig. 5). In the moist and cold forests, tolerant tree and shrub regeneration is common even in forests with continuous canopy cover (Cooper and others 1991) (figs. 2 and 3). How these ground vegetative layers develop into mid-canopy layers depends on disturbance and how species differentiate as they develop based on their competitive and successional abilities (Oliver and Larson 1990). Stem differentiation can occur in even-aged, single species forests as stands self-thin because of inter-tree competition often in association with disturbances such as fires, diseases, and insects. Most often, these mid-canopy layers constitute the majority of the ladder fuels (Sandberg and others 2001).

Canopy Fuels

Crown fuels (also referred to as canopy fuels or aerial fuels) are those suspended above the ground in trees or vegetation (for example, vines, mosses, needles, branches etc.) (fig. 10). These fuels tend to consist mostly of live and fine materials less than 0.635 cm (0.25 in) in diameter. Crown fuels are the biomass available for a crown fire, which can ignite from a surface fire via understory shrubs and trees (ladder fuels) or from crown to crown. The shrub/small tree stratum is also involved in facilitating crown fires by increasing surface fireline intensity and serving as ladder fuels that provide continuity from the surface fuels to canopy fuels. These essentially bridge the vertical gap between surface and crown strata. The size of this gap is critical to ignition of crown fire from a surface fire below (Van Wagner 1977). Van Wagner (1977) identified two thresholds of crown fire activity: (1) crowns are ignited after the surface fire reaches critical fireline intensity relative to the height of the base of the aerial fuels in the crown; (2) crown ignition can become an "active" crown fire if its spread rate is high enough to surpass

Figure 13. Fires can progress from torching individual and groups of trees (A) until entire hillsides are a blaze (B) (K. Watenmaker photograph).

the second threshold based on the crown density (often referred as canopy bulk density, canopy weight for a given volume). Aerial fuels separated from surface fuels by large gaps are more difficult to ignite because of the distance above the surface fire, thus requiring higher intensity surface fires, surface fires of longer duration that dry the canopy before ignition, or mass ignition from spotting over a wide area (Byram 1966). However, once ignited, high-density canopy fuels are more likely to result in a spreading crown fire (active crown fire) than are low-density canopies (fig. 13).

The upper canopy is composed of leaves, branches, and boles of trees. Again, depending on the forest, its setting, and inherent disturbances, these layers may be simple and uniform such as those that occur in young (30 to 50 years) to mid-aged (80 to 120 years) early-seral species such as lodgepole pine and western larch. In contrast, dense and highly complex upper canopy layers often occur in late-seral, moist forests in which over five conifer species may occur (Haig and others 1941).

Wildfire

Several terms and concepts describe a wildfire, including fire intensity, fire behavior, fire severity, fire line intensity, burn severity, and fire effects (Scott and Reinhardt 2001; Peterson and others 2005). However, several of these terms are often used interchangeably and mean different things to different individuals and disciplines. Therefore, to further the understanding of wildfires and their effects, we suggest the circumstances concerning a wildfire be described as a continuum, beginning with the pre-fire environment, then fire environment, and finishing with the post-fire environment (Jain and others 2004).

Pre-Fire Environment

The pre-fire environment describes the condition of the forest before a fire occurs. It includes, but is not limited to, such descriptors as physical setting (for example, location, geology, soils, topography, landform, etc.), current vegetation, seral stage of vegetation, structure of vegetation, and fuel moisture content. Other information, such as the time since fuel treatment and time of year may also be included (Peterson and others 2005; Jain and Graham 2007) (figs. 4 through 6).

Fire Environment

The fire environment includes the state of the fuel, physical setting, weather (short- and long-term), and their interactions (Graham and others 2004; Jain and others 2004; Peterson and others 2005; Rothermel 1983). Weather characteristics, such as wind speed and direction, relative humidity, temperature, and atmospheric stability interact with physical attributes such as aspect, slope angle, topographic position, and landscape orientation. In turn these characteristics interact with the presence or absence of fuel (in other words, canopy, surface, or even buildings), their moisture contents, composition, and structure (for example, multiple canopies of conifers and deciduous trees and shake-roofed homes). As such, how a fire ultimately burns, and its effects are predicated on weather, and how the fuels are arranged within the physical setting (Finney 2001; Graham and others 2004; Peterson and others 2005; Scott and Reinhardt 2001) (fig. 10).

Fire intensity (fire line intensity) describes how a fire burns and the amount of energy it produces. Flame length, rate of spread, amount and location of torching, and spotting distance are frequently used to describe fire intensity (Jain and others 2004). The direct effects of a fire are often referred to as fire severity and/or as first-order fire effects. Fire severity is a function of how much heat a fire produces and the resulting consumption of plants and forest floor, heating of soils (which can volatize nutrients, create water repellent soils, distill organic matter, etc.), smoke production, and in some circumstances, the burning of homes (Cohen and Stratton 2003; Jain and others 2004; Ryan and Noste 1983; Turner and others 1999; Ulery and Graham 1993). For example, tree fire severity describes the amount of pre-fire tree crowns consumed, charred, or scorched, and the proportion of the tree cambium killed (Ryan and Reinhardt 1988). Similarly, how a community responds (for example, supportive, angry, frustrated) to fire suppression activities or smoke could also be a direct fire effect (Kent and others 2003).

Post-Fire Environment

The post-fire environment describes conditions when all the flames, smoldering, and heat are gone, or in simpler terms, what is left after the fire is out (fig. 14) (Jain and others 2004). Descriptors of the post-fire environment include, but are not limited to, the amount and condition of live and dead vegetation, condition of the forest floor and soils, and the state of the homes and buildings impacted by the fire (Cohen and Stratton 2003; Jain and others 2004; Peterson and Arbaugh 1986; Ryan and Noste 1983; Ryan and Reinhardt 1988; Skinner and Weatherspoon 1996). However, the post-fire environment does not describe what was consumed, nor does it describe the influence of the fire alone (Jain and others 2004). Instead it assumes that the pre-fire environment, fire environment, and combustion processes all contributed to creating the post-fire environment or burn severity (figs. 14 and 15).

Figure 14. A ponderosa pine stand burned by the highly intense Rodeo-Chediski Fire in Arizona that resulted in high tree burn severity (level 5) (Jain and Graham 2007).

Figure 15. Ground fires (especially those with long residence times) can severely burn soils, removing organic matter, volatilizing nutrients, killing tree roots, and creating water impermeable layers (soil burn severity level 6) (Jain and Graham 2007).

Burn Severity

How an ecosystem responds in the post-fire environment is often referred to as indirect or second-order fire effects (Reinhardt and others 2001). These effects include characteristics such as soil erosion and sedimentation of streams, the opening of serotinus lodgepole pine cones, colonization of a burned forest by woodpeckers, and the introduction of exotic plants. Burn severity may also include homeowners' response to burned houses or community reaction to post-fire rehabilitation activities (Kent and others 2003). These biological, physical, social, and economic responses to a fire vary over time and space and are interdependent. Therefore, we suggest the term burn severity best describes the post-fire environment because for a given fire intensity (behavior), depending on the pre-fire environment, a fire can create a variety of post-fire characteristics (Cohen and Stratton 2003; Graham and others 2000; Hungerford and others 1991; Reinhardt and others 1991; Robichaud and others 2003). For example, an intense canopy consuming wildfire can severely burn tree crowns but minimally affect the forest floor and surface soils. This wildfire outcome frequently occurs when the surface organic layers are too moist to burn, resulting in low soil burn severity. Similarly, a low intensity surface fire with long residence times can produce large amounts of heat that can alter mineral soil, kill vegetation, and predispose trees to insect attacks (figs. 11 and 15). As such, a low intensity fire can result in high burn severity. To this end, we have found that the relation between soil burn severity and tree burn severity is inconsistent. For example, tree death can be a function of soil burn severity, tree burn severity, or both severities combined. In addition, it appears that forest structure and composition can differentially influence both soil burn severity and tree burn severity.

Soil Burn Severity

We use six levels to classify soil burn severity that link fire intensity to the physical and biological responses to the fire (Jain and Graham 2007). The factors we chose to define the levels include: proportion of surface organic layers, mineral soil, and exposed rock present after a fire and the dominant char class defined as unburned and black char specific to both litter and mineral soil, and grey and orange char specific to mineral soil.

Level 1 soil burn severity occurs when surface organic horizons (litter, humus, rotten wood) cover approximately ≥85 percent of the forest floor after a fire, either in an unburned state or showing some evidence of black char (Debano and others 1998; Ryan and Noste 1983; Ulery and Graham 1993). This ≥85 percent threshold for organic horizons followed the soil quality standards used in Forest Service Regions 1, 2, 4, and 6 that define a 15 percent loss of soil organic horizons as detrimental to forest productivity (Page-Dumroese and others 2000). Not only do unburned surface organic layers maintain productivity they also provide refugia for microbes and other organisms (Hungerford and others 1991; Neary and others 1999).

Level 2 soil burn severity occurs when surface organic horizon cover is ≥40 and <85 percent of the forest floor, indicating minimal consumption occurred (Wells and others 1979). However, much of the mineral soil could be covered by black charcoal, and small and isolated gray to orange colored mineral soils may be present. Soils exhibiting this burn severity would contain nutrients in the surface horizons and microbes would be living in the humus and/or mineral soil layers (Hungerford and others 1991).

Level 3 soil burn severity occurs when forest floor conditions exhibit less than 40 percent surface organic horizons and black char dominates the site. Little or no areas are unburned, but a considerable amount of black, charred litter exists (Hungerford and others 1991; Neary and others 1999). Although some volatilization of soil nutrients would occur when large logs burned, many nutrients remain in the upper mineral soil and residual surface organic materials (organic cover tends toward 40 percent rather than 5 percent). If the pre-fire environment favors soil erosion (for example, slope angle, soil texture, parent material, etc.), this level of soil burn severity would indicate the potential for rill erosion; however, soil heating and duration may be insufficient to create hydrophobic soils (Debano 2000; Johansen and others 2001; Wondzell and King 2003).

Level 4 soil burn severity occurs when organic layers comprise less than 40 percent of the forest floor and gray or orange colored char dominates the mineral soil. The organic layers covering the site would be dispersed, and isolated amounts of surface organics and nitrogen volatilization would most likely occur. Refugia for organisms would be absent except in isolated areas, water repellent soil could exist, all areas would show some evidence of fire, and temperatures that create these types of conditions would have ranged from 20 to 400 ^{o}C (Debano 2000).

Level 5 soil burn severity occurs when no litter is present and black soil dominates the area. Depending on the site, potential soil erosion could be high and nitrogen in the surface organic horizons would likely be volatized, but nutrients in the mineral soil would likely remain (Hungerford and others 1991).

Level 6 soil burn severity describes a forest floor condition where no surface organics are present and gray to orange char dominates the mineral soil appearance. This soil burn severity indicates that minimal or if any nutrients or microorganisms are present in the soil surface (Debano 2000). Soil erosion and water repellent soils are possible and the abundance and presence of exposed rock may influence overland water flow (fig. 15).

Tree Burn Severity

Tree burn severity can be characterized two ways (Jain and Graham 2007). The first describes the vertical distribution of post-fire tree condition and is typically used to estimate post-fire tree mortality (Peterson 1985; Peterson and Arbough 1986; Wyant and others 1986). For example, ponderosa pines with 30 percent green crown have a 0.38 probability of dying after a fire and grand firs with similar crowns have a 0.87 probability of dying (Ryan and Reinhardt 1988). Secondly, we define horizontal tree burn severity as the proportion of trees and their condition remaining in a patch, stand, or landscape after a fire (Broncano and Retana 2004; Turner and others 1999) (fig. 14).

We identified five levels of tree burn severity. Because in most wildfires there are areas that do not burn, we defined a level 0 in our tree burn severity, as did several other authors (for example, Hutto 1995; Pollet and Omi 2002; Weatherspoon and Skinner 1995). Level 1 tree burn severity is defined as trees having greater than 60 percent of the residual crown remaining green. Defined horizontally, level 1 burn severity would have all trees within an aerial extent containing trees with greater than 60 percent green crowns. Mixed green burn severity (level 2) is highly variable with trees containing <30 percent green crowns to those having over 60 percent residual green crowns. Completely brown (scorched) trees can be present within level 2, but no trees with completely black crowns can occur. Typically, the site is dominated by trees containing some green crowns (Hutto 1995; Pausas and others 2003; Skinner and Weatherspoon 1996). Mixed brown crown severity (level 3) also contains green trees; however, trees with black crowns (needles consumed by the fire) occur as well as some trees with partially consumed crowns (brown needles and no needles present). In this situation, the abundance of green trees tends to be much less when compared to the abundance of completely dead trees (black or brown trees).

Level 4 tree burn severity occurs when the entire spatial extent contains trees without green needles and the majority are brown. However, within a given patch, some trees could be black (all needles consumed). When brown needles fall, they can decrease inter-rill and rill soil erosion, and because of the organic matter input, soil productivity can be enhanced (Harvey and others 1987; Pannkuk and Robichaud 2003). Level 5 tree burn severity occurs when an intense crown fire consumes all foliage (needles), leaving black branches, stems, and boles (figs. 13 and 14).

The tree and soil burn severity levels we defined are a series of classes that partition a continuum from everything remaining after a wildfire (as in the pre-fire state), to where all of the foliage and forest floor are consumed leaving blackened boles and bare soil. As such, the soil and tree burn severity we present is highly flexible allowing for the severity levels to be combined or kept separate depending on the data resolution required to meet a particular purpose or need.

Fuel Treatments

Crown fires are generally considered the primary threat to forests and human values; however; low intensity surface fires can degrade forest soils, kill trees, and burn domestic structures (Cohen and Stratton 2003; Graham 2003; Graham and others 2004) (fig. 11). In the moist, cold, and dry forests, crown fires are also the primary challenge for fire management. Our current understanding of fire behavior in most forests indicate that a crown fire begins with a transition from a surface fire to the ignition of the canopy. Therefore, crown fire development depends on the sequence of available fuels beginning with surface fuels, followed, in order, by woody fuel, low vegetation, shrubs, ladder fuels, and canopy fuels (figs. 5, 6 and 10).

There is a wide range of treatments that can be used to modify forest fuels (vegetation) that in turn influence both fire intensity (behavior) and burn severity (what is left). As a result, fuel treatments not only influence how a fire burns, but they can also influence how a fire affects water quality, timber quantities, wildlife habitat, scenery, and other forest elements that society values. Treating forests to produce commodities and values that society favors is not a new endeavor. Forest treatments that produced desired conditions over time existed by the mid-1600s (Evelyn 1664). These treatments have been improved over the years, especially the last 100 years, with continued research and experience in the practice and art of silviculture (Nyland 2002; Smith and others 1997).

In general, silviculture is the art and science of controlling the establishment, growth, competition, health, and quality of forests and woodlands to meet the diverse needs and values of landowners and society on a sustainable basis (Helms 1998). The plan and execution of forest and fuel treatments over time fall under the umbrella of a silvicultural system. There are not two or three systems, but rather an infinite number that can be developed and implemented over time and space integrating biology, management, and economic knowledge to treat forest fuels (Nyland 2002; Schlich 1906; Smith and others 1997). Even though silvicultural methods historically stressed the production of wood crops, treatments can be designed to create and maintain a variety of forest compositions and structures relevant and effective for modifying fire intensity and burn severity. The challenge is to develop systems to manage forests as fuel yet create variable and complex forest structures and compositions that address other values such as maintaining the sense of place or maintaining wildlife habitat (Graham and Jain 2004).

The most effective strategy for reducing crown fire occurrence and tree and soil burn severity is to:

1. reduce surface fuels,
2. increase height to live crown,
3. reduce continuity of the forest canopy, and
4. reduce canopy bulk density (Cruz and others 2002; Graham and others 1999a; Scott and Reinhardt 2001; Van Wagner 1977) (fig. 10).

The documentation of treatments in a fuel treatment strategy is written in a silvicultural prescription that describes the treatments and the resulting composition and structure of dead and live vegetation through time.

Forest Floor Treatments

Forest floor treatments, possibly more than any other, influence how forest vegetation establishes and develops and are major determinants of both fire intensity and burn severity. No matter what method is used to treat the forest floor, a key to treatment success is ensuring that highly diverse (for example, diverse amounts of CWD, bare soil, litter depths and compositions etc.) forest floor conditions remain after a treatment to facilitate forest recovery after a wildfire. By doing so, a fire burning these fuels would burn heterogeneously and result in soil burn severity dominated by levels 1 through 4.

Mechanical Treatments

Machines through their proper use can create and maintain desired forest floor conditions in many forest settings. However, if not used properly they can compact and/or displace mineral soils and reduce soil organic content (Harvey and others 1996; Page-Dumroese and others 1997). As such, they are usually limited to operating on gentle slopes (approximately <40 percent). Machines capable of separating slash of different sizes, often called grapple machines, displace less soil than rakes attached to the front of tractors, which in turn conserve and protect the soil surface layers (fig. 16). After piling the slash, although usually burned, piles can also be chipped, used for bioenergy, or used for domestic firewood.

Machines can rearrange, compact, chip, masticate, or otherwise change the fire hazard without reducing the fuel loads (fig. 17). The effects such mastication and similar machines have depends on the size, composition, and location of the residual fuels they leave (Graham and others 2000). For example, thin layers of wood chips spread on the forest floor tend to dry and rewet readily, and deep layers of chips and chip piles may have insufficient air circulation creating poor decomposition conditions. In addition, these layers of small woody material on the forest floor can insulate the soils and when decomposition does occur, the decomposing organisms utilize large amounts of nitrogen, which reduces its availability to plants. Therefore, use of any of these crushing, chipping, or mulching treatments needs to consider the impact on the decomposition processes and the potential contribution to smoldering fires.

Figure 16. Grapple machines have the ability to separate coarse from fine fuels more readily than tractors and tend to displace and compact soils less than tractors.

Figure 17. Masticator used to treat ladder and surface fuels in the moist forests near Lake Tahoe, California.

Prescribed Fire

Prescribed fire is commonly used to treat and manage forest floor conditions throughout the western United States. Fire can reduce fine fuels, reduce ground-level vegetation, preserve surface organic layers, and maintain appropriate amounts of coarse woody debris (fig. 18). However, unless fuel and weather conditions at the time of ignition are appropriate, fire can also create conditions adverse to vegetative development and impair soil productivity (Debano 1991; Hungerford and others 1991). The amount of forest floor consumed by a fire is dependent on its moisture content, particularly in the lower humus and fermentation layers. Within the Inland West, when the moisture content of these lower layers exceeds 100 percent when a fire occurs, the majority of these layers are generally conserved (Ryan 1982). When burning occurs under these conditions, nutrients (P, N, K) in the litter and fine fuels (≤7.6 cm, 3 in) have the opportunity to condense in the lower layers (for example humus) and, therefore, are not lost from the site (Harvey and others 1989) (soil burn severity less than or equal to level 3).

Figure 18. (A) Fire being used to decrease the amount of organic material that developed, most likely because of fire exclusion, at the base of this large ponderosa pine located in southern Idaho. Fire was applied early in the spring when the temperature of the lower organic layers was below 4.4 °C (40 °F) (when fine root activity is minimal) and when their moisture contents exceeded 100 percent. (B) The application of a low intensity prescribed fire treating the entire free selection area after three spring “snow well” treatments used to reduce the organic layers at the base of the trees.

Some ground-level vegetation responds vigorously to heat, such as *Ceanothus* spp., which has seeds buried in the forest floor. In addition, many ground-level species sprout aggressively in response to fire (fig. 19). The amount, kind, season, and type of fire used to treat the forest floor can create a particular soil burn severity. Therefore, the relation between the desired soil burn severity and the expected response of vegetation for a given forest, PVT, and biophysical setting can be highly variable (Baumgartner and others 1986, 1989; Bradley and others 1992a, 1992b).

Prescribed fire can reduce horizontal fuel continuity (shrub, low vegetation, woody fuel strata), which in turn disrupts growth of surface fires, limits their intensity, and reduces the potential of spot fire ignition. In addition, by reducing fine fuels, duff, large woody fuels, and rotten material, their continuity changes the fuel energy stored on the site and potentially reduces both fire intensity and burn severity (figs. 13, 14 and 15). Also, prescribed fire can directly consume low ladder fuels (shrubs, dead trees, needle drape, and small trees) and scorch and kill the lower branches of overstory trees, effectively raising the live crown above the ground surface.

Prescribed fire is an excellent tool for treating the forest floor; however, there are also considerable risks and uncertainties with its use (Biswell and others 1973; Cooper 1960; Fernandes and Botelho 2003; Weaver 1955, 1957). For example, fire cannot readily create precise stand structures and compositions compared to controlled mechanical treatments. Climatic and fuel moisture conditions can severely restrict when fire can be used, especially in forests with large amounts of fuel or located in areas sensitive to smoke production. Also, fires may escape and cause unintended resource and economic damage. Even with these short-comings, prescribed fire, by influencing multiple fuels, can effectively modify both fire behavior and burn severity.

Ladder Fuel Treatments

Precommercial thinning and other intermediate forest treatments can be designed to target specific fuel strata and disrupt, (1) the vertical progression of fire from surface

Figure 19. Ponderosa pine saplings growing among *ceanothus*, a ground-level shrub that responds aggressively in response to fire. The clearcut is located on a grand fir potential vegetation type in eastern Oregon.

fuels to ladder fuels to canopy fuels and (2) the horizontal progression of fire through individual fuel strata. These treatments can be designed and applied to reduce both the continuity and density of shrubs and trees and disrupt fire spread, and prevent the fire from producing sufficient heat to detrimentally affect the surface soils (Biswell 1960; Biswell and others 1973; Cooper 1960; Fisher 1988; Martin and others 1989; Pollet and Omi 2002; Scott 1998a; 1998b; Scott and Reinhardt 2001; Weaver 1955). Depending on the forest and its condition, these tending activities can occur at a variety of time intervals and intensities, thereby creating an infinite number of stand structures and compositions (Graham and others 1999a) (figs. 17 and 20).

Treatments emphasizing the smaller trees and shrubs (ladder fuels) can effectively reduce vertical fuel continuity that fosters crown fire initiation (fig. 21). During this period in the life of a forest, the structure and composition are the most plastic (in other words, responsive to treatments) and future stand dynamics are largely determined (Graham 1988; Haig and others 1941; Pearson 1950). In addition, thinning small material and pruning branches are more precise methods than prescribed fire for targeting ladder fuels and specific fuel components in the ladder-fuel stratum. The net effect of removing ladder fuels is that surface fires burning through treated stands are less likely to produce enough energy to ignite the overstory fuels.

The issue in many forests is excessive regeneration, which makes weeding and cleaning young stands essential to develop fire resilient forests (Helms 1998; Smith and others 1997). For example, in the ponderosa pine forests of the Black Hills (northeastern Wyoming and western South Dakota), ponderosa pine regeneration is often prolific, creating a fire hazard and compromising stand development (Shepperd and Battaglia 2002) (fig. 22). Similarly, in both the cold and moist forests, abundant regeneration is often the norm; again requiring tending operations to develop stand characteristics that reduce fire intensity and burn severity (Deitschman and Pfister 1973; Graham 1988; Haig and others 1941; Johnstone 1985; Johnstone and Cole 1988) (fig. 23). Such ladder fuel treatments can be most often accomplished mechanically and, in rare situations, fire can be used (Saveland and Neuenschwander 1988).

Canopy Treatments

Classically, the term thinning was applied to stand treatments aimed at redistributing growth on remaining stems, but often, any kind of partial cutting (tending) such as

Figure 20. A thinned mixed conifer stand located in northern Idaho.

Figure 21. The first entry of a free selection system being accomplished using a masticator. Depending on stand conditions, the majority of the ladder fuels and small trees (less than 20 cm, 8 in) were chunked, leaving a clumpy high forest canopy. The objective of the system is to create and maintain a wildfire resilient ponderosa pine forest located in southern Idaho, similar to those that occurred historically.

Figure 22. Ponderosa pines in the Black Hills of western South Dakota tend to regenerate prolifically using shelterwood regeneration methods.

Figure 23. In moist forests, western hemlock (a late-seral species) readily regenerates in relatively closed canopy conditions to create multiple fuel layers.

liberation, preparatory, improvement, sanitation, and selection cuttings could be termed thinning (Graham and others 1999a). Thinnings can be designed to affect canopy bulk density, which in turn determines whether a crown fire could be sustained.

Low thinning occurs when trees are removed from the lower canopy, leaving large trees to occupy the site. This method mimics mortality caused by inter-tree competition or surface fires, and primarily removes small and suppressed trees (ladder fuels). Crown and selection thinnings can be used to reduce canopy density and continuity within the main forest canopy and alter forest composition (Nyland 2002) (fig. 20). Usually, different tree species have characteristic development rates that result in individual species dominating specific canopy layers. For example, in many dry forests, ponderosa pine primarily occupies the dominant canopy layers, whereas shade-tolerant grand fir, white fir, or Douglas-fir occupy the intermediate and suppressed layers (fig. 6). In this situation, low thinning favors the development of the dominant and codominant ponderosa pine, often a desirable forest fuel condition. Thinnings can remove few to many trees and need not create regularly spaced forests, but the number, clumping, and juxtaposition of residual trees can be varied (Long and Smith 2000; Nyland 2002; Reynolds and others 1992). As such, thinnings can precisely create targeted stand structures and compositions that will influence both fire intensity and burn severity (Agee and others 2000; Miller and Urban 2000; Stephens 1998; van Wagtendonk 1996; Weatherspoon and Skinner 1996).

A clearcut may be an appropriate fuel treatment, especially for fuel breaks or in other areas in which the spatial continuity of fuels needs to be drastically decreased (Agee and others 2000; Graham and others 2004). Disposal of hazard fuels (in other words, activity fuels, tree limbs, and boles less than 7.6 cm, 3 in diameter etc.) can occur by using prescribed fire and/or mechanical means. This amount of canopy and fuel removal will have the greatest impact on watershed, wildlife, and wildfire responses. Clearcuts can be modified by not removing all trees but retaining a few to many. Traditionally for timber production, seed-tree and shelterwood regeneration methods created such conditions. This condition might be similar to those created by a mixed severity fire (Graham and Jain 2005; Graham and others 2005). The retained trees will modify a site's environment, influencing fuel moistures and how fires would burn. The number, location, juxtaposition, disposition (longevity), and species of the residual trees can be specified. By doing so, a variety of forest structures and compositions and landscapes can be developed (Long and Smith 2000; Reynolds and others 1992).

Selection systems are another set of forest treatments that have applicability for treating all forest (fuel) layers (Graham and Jain 2005; Graham and others 1999b; Graham and Smith 1983; Long and Smith 2000; Marquis 1978; Nyland 2002; Reynolds and others 1992; Smith and others 1997). Depending on the design, they can maintain high forest cover, which is an important component to many current management objectives. Individual tree, group, and free (irregular) selection systems have been described. As traditionally applied, they maintain uneven-aged (diameter distributions) forest structures by planning for and executing frequent (in other words, 10- to 20-year) entries (treatments). However, by varying the opening sizes, canopy gaps, tree clumping, species preferences, and their juxtapositions, highly heterogeneous forest structures and compositions can be created and maintained (Jain and others 2008). An integral part of all selection systems is the tending of all canopy (fuel) layers over time and space to ensure the desired conditions are created and maintained (Graham 1990; Graham and others 1999b). This is of particular importance in treating the ladder fuels and multiple canopies using selection systems. Canopy gaps, highly variable tree arrangements, and forest floor treatments can be used in concert, as to ensure the lethal fire risk is not exacerbated (Graham and Jain 2005) (fig. 18).

Fuel treatments can increase the probability of modifying fire behavior and burn severity during most weather conditions. However, extreme weather conditions (low fuel moisture contents, low humidity, high winds) can create fire behavior that can burn through or breach most fuel treatments (Finney and others 2003) (fig. 24). A realistic objective of fuel treatments is to reduce the likelihood of crown fire and severely burned soils that would lead to a loss in value or undesirable future conditions and not necessarily guarantee crown fire elimination.

Figure 24. The intense Rodeo-Chediski Fire that burned in Arizona in 2002. This ponderosa pine forest was historically burned by low intensity surface fires, but singly and in combination, fire exclusion, timber harvest, climate change, and livestock grazing contributed to forest changes that facilitated this uncharacteristically intense fire. Often, fuel treatments will locally impact such fires but most treatments are readily breached.

Canopy Treatments Combined With Forest Floor Treatments

The most effective and appropriate sequence of fuel treatments depends on the amount of surface fuel present; the density of understory and mid-canopy trees (Fitzgerald 2002); long-term potential effects of fuel treatments on vegetation, soils, and wildlife; and short-term potential effects on smoke production (Huff and others 1995). In forests that have not experienced fire for many decades, multiple fuel treatments are often required to achieve the desired fuel conditions. Canopy treatments, followed by prescribed burning, reduce canopy, ladder, and surface fuels, thereby providing maximum protection from intense fires in the future (Peterson and others 2005). Potential fire intensity and/or burn severity in treated stands is significantly reduced only if canopy treatments are accompanied by reducing the surface fuels (woody fuel stratum) created from the thinning operations (Alexander and Yancik 1977; Graham and others 1999a) (fig. 18). Given current accumulations of fuels in some stands, multiple prescribed fires—as the sole treatment or in combination with thinning—may initially be needed, followed by long-term maintenance burning or other fuel reduction (for example, mowing, mastication, scarification, etc.) to reduce crown fire hazard and the likelihood of high burn severity (Jain and Graham 2004; Peterson and others 2005). (see Chapter 4).

The most appropriate fuel treatment strategy is often thinning (removing ladder fuels and decreasing tree crown density), followed by prescribed fire, piling and burning, mastication of fuels, or other mechanical treatments that reduce surface fuel amounts and often promote decomposition (figs. 18 and 21). This approach reduces canopy, ladder, and surface fuels, thereby reducing both the intensity and burn severity of potential wildfires. Restoring forests to a condition in which fire alone can maintain the desired conditions will take time (fig. 18). Wildland fire use (that is, allowing certain wildfires to burn under certain conditions and locations) offers some hope once homes, communities, and key resources are protected through thinning, prescribed fires, or other treatments.

Post Treatment Environment

Thinning and prescribed fires can modify understory microclimate that was previously buffered by overstory vegetation (Scott and Reinhardt 2001; Pollet and Omi 2002; Weatherspoon 1996). Thinned stands (open tree canopies) allow incoming solar radiation to penetrate to the forest floor increasing surface temperatures, decreasing fine fuel moisture, and decreasing relative humidity compared to unthinned stands—conditions that facilitate intense fires (Countryman 1955; Pollet and Omi 2002). An increase in surface fire intensity may increase the probability of a fire exceeding the critical threshold needed to initiate a crown fire (Van Wagner 1977). Therefore, it is important that the gap between the surface and crown fuels is increased by either prescribed fire or pruning (fig. 18) so if a fire should occur, the potential for crown fire initiation is minimized. Rothermel (1983) found significant differences in fire behavior between a closed stand (no harvest) and an open stand (thinned). Changing stand structure, while ignoring surface fuels, will only affect the likelihood of active crown fires—it will not necessarily reduce the likelihood of surface fires intense enough to damage soils or cause significant overstory mortality. For example, figure 25 illustrates the build-up of organic layers as a function of fire exclusion. If only ladder and crown fuels are treated ignoring the ground fuels, in this situation, a fire would most likely lead to high soil burn severity and killing of trees through cambium kill. Moreover, surface fire intensity may be greater in thinned stands compared to untreated stands depending on whether the thinning activity adds to the surface fuels (Alexander and Yancik 1977; van Wagtendonk 1996; Stephens 1998). Therefore, it cannot be emphasized enough that all fuel layers need to be managed (over time and space) to minimize the unwanted consequences of wildfires (fig. 10).

Treatment Longevity

Very few specific experiments have evaluated the longevity of treatments and their effectiveness in altering fire behavior and burn severity. However, there is considerable information on forest growth and development, and this information is useful in providing estimates on the longevity of potential treatments. There are good models available (Reinhardt and Crookston 2003; Wykoff and others 1982) that predict vegetation development over time and provide estimates of treatment longevity.

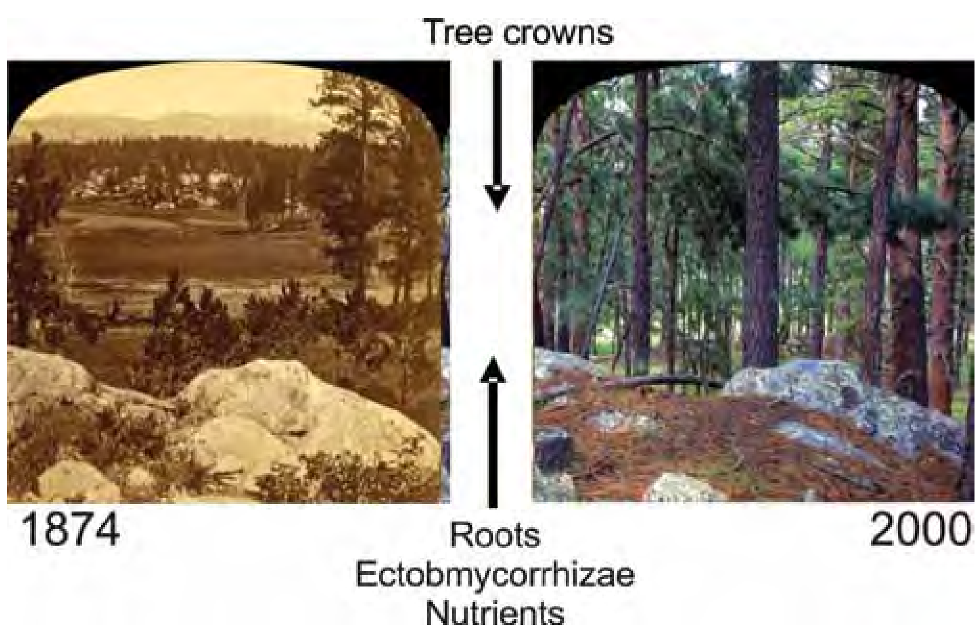


Figure 25. When fires are excluded from ponderosa pine forests, organic layers tend to accumulate and tree roots, ectomycorrhizae, and nutrients also tend to concentrate in these layers. Note the contrast between the amount of organic material on the forest floor when General Custer came through the Back Hills in 1874 and the amount that has accumulated around the rocks in the photo in 2000. Photograph courtesy of Paul Horsted/custertrail.com (Grafe and Horsted 2002).

There are limited amounts of information on the relation between canopy structure and ground-level vegetation, or the relation between vegetation development and fuel moisture. For short time periods (months) after treatment, fuel changes can produce dramatic differences in fire behavior. Biswell and others (1973) showed that the effectiveness of prescribed fire treatments in maintaining desired fuel conditions decreased significantly over 2 decades in a ponderosa pine forest. Van Wagtendonk and Sydoriak (1987) directly examined fuel accumulation following prescribed burning and found that fuel amounts reached 67 percent of their pre-burn loading after 7 years. Many of the prescribed fires they used were the first fuel treatments that occurred in these stands in decades and would potentially kill many small trees that would contribute to the woody fuel load. Repeated burns were not studied, but the elimination of small trees using a series of burns would be expected to retard fuel accumulation compared to the amounts they reported. Van Wagtendonk and Sydoriak (1987) concluded that prescribed burning would be required at least every 11 years to maintain fuel loads below their preburn condition. Van Wagtendonk (1995) also reported reductions in fire spread and intensity of fires up to 14 years after previous burns within the mosaic of large fires in the mixed-conifer forests of Yosemite National Park.

The duration of treatment effectiveness will vary with climate, PVT, soils, and other factors that influence productivity and the nature of the fuel treatments (Keyes and O'Hara 2002). For example, the longevity of thinning slash is greater on drier sites, particularly for finer woody material compared to fine fuels occurring in wetter forests (Christiansen and Pickford 1991). Treatment effects will likely last longer in areas in which vegetation development is slower than in highly productive areas where vegetation development is more rapid and lush (Weatherspoon and Skinner 1996). Few data exist, but inferences from fire history research and modeling show that the length of treatment effectiveness will vary with forest type (general fuel characteristics) and fire regime (Heyerdahl and others 2001; Miller and Urban 1999; Taylor and Skinner 1998; Taylor and Skinner 2003).

Fuel Treatment on Landscapes

The spatial patterns of fuel treatments in landscapes will most likely determine their effectiveness in modifying wildfire behavior (Hessburg and others 2000), because multiple stands and fuel conditions are involved in large fires (Finney 2001). Fire behavior under extreme fire weather may involve large areas of fuels, multiple fires, and spotting, so a "firesafe" landscape needs to populate hundreds to thousands of hectares (acres) with strategically located fuel treatments (Finney 2003). Treating small or isolated stands without assessing the broader landscape will most likely be ineffective in reducing wildfire extent and severity.

There are limited examples of how fuel treatments have altered subsequent fire behavior and burn severity (Helms 1979; Martin and others 1989; Pollet and Omi 2002). However, despite small-scale modification of fire behavior, none of these studies demonstrated that spread or behavior of a large fire was significantly altered, probably because the units were relatively small and were surrounded by areas containing vegetation favoring continued fire growth. In the mixed-conifer forests of northern California, fire intensity varied with dominance of short-needle or long-needle conifers in the same fire regime (frequent, low-moderately intense surface fires). Under similar burning conditions in a retrospective study of the widespread fires of 1987, stands dominated by Douglas-fir sustained significantly less damage than did stands dominated by ponderosa pine (Weatherspoon and Skinner 1995). Given current fuel accumulations across the Interior West, small areas (unknown threshold) favoring low intensity fires will probably be irrelevant to fire behavior (Dunn 1989; Salazar and Gonzalez-Caban 1987). Therefore, treatments that alter vegetation to favor low intensity fires must consider spatial arrangement of fuel structures to alter wildfire behavior.

Large-scale, frequent mosaic burning may maintain many portions of some landscapes in a treated condition and disrupt growth of the inevitable wildfire (Brackebusch 1973). Evidence that mosaic patterns reduce fire spread comes from natural fire patterns that have fragmented fuels across landscapes. This spatial pattern produces self-limiting fire growth and behavior by management of natural ignitions, as shown in Yosemite National Park and Sequoia National Park (van Wagtendonk 1995, 1996). The spatial arrangement of vegetation influences the growth of large fires (Brackebusch 1973; Finney 2001). Patches of vegetation that burn relatively slower or less intensely than surrounding patches may force the fire to move around them by flanking (at a lower intensity), which locally delays the forward progress of the fire. Such strategically placed treatments create landscape fuel patterns that collectively slow fire growth and modify behavior while minimizing the amount of treated area required (Finney 2001) (fig. 26). The importance of spatial pattern is emphasized by findings that random fuel treatment arrangements (Finney 2003) are extremely inefficient in changing fire behavior (fig. 26)—requiring perhaps 50 to 60 percent of the area to be treated compared to 20 percent in a strategic fashion (Finney 2001). If fuel treatments are to be effective at changing the growth of large fires, then strategic placement of treatment areas must incorporate land ownership, endangered species, riparian buffers, and other concerns. The costs and maintenance levels needed to maintain this forest pattern would vary depending on forest type, PVT, access, and public acceptance.

An alternative to a landscape approach to altering fuels is to create fuel breaks, which modify easily accessible portions of the vegetation in strategic locations across a landscape (Agee and others 2000; Weatherspoon and Skinner 1996). The purpose of fuel breaks, which are typically placed in defensible locations like a ridge, is to aid suppression efforts of firefighters to stop fire spread (Green 1977). The benefits of a fuel break are successful only if the fire suppression activities anchored to the fuel break limit the size or perimeter of the fire. If fire suppression does not occur, a fire can continue to burn through the fuel break with little or no effect on fire size. Moreover, fuel breaks (clearcuts) most often require long-term maintenance and repeated treatments.

Treatment Efficacy

Fire behavior and burn severity (see section on burn severity, page 42) are strongly influenced by stand structure as it relates to live and dead fuel loadings and ladder fuels. The type and abundance of surface fuels have an effect on the abundance of falling embers, which can ignite distant fuels and spread, thus influencing fire behavior. Reducing both ladder fuels and surface fuels is essential to effectively change fire behavior and burn severity (Graham and others 1999a; Omi and Kalabokidis 1991; Pollet and Omi 2002). Examples from the Hayman Fire in Colorado illustrate these interactions (Finney and others 2003). The Polhemus prescribed burn in November 2001 removed most surface fuels and pruned lower live branches from trees in a ponderosa pine forest but did

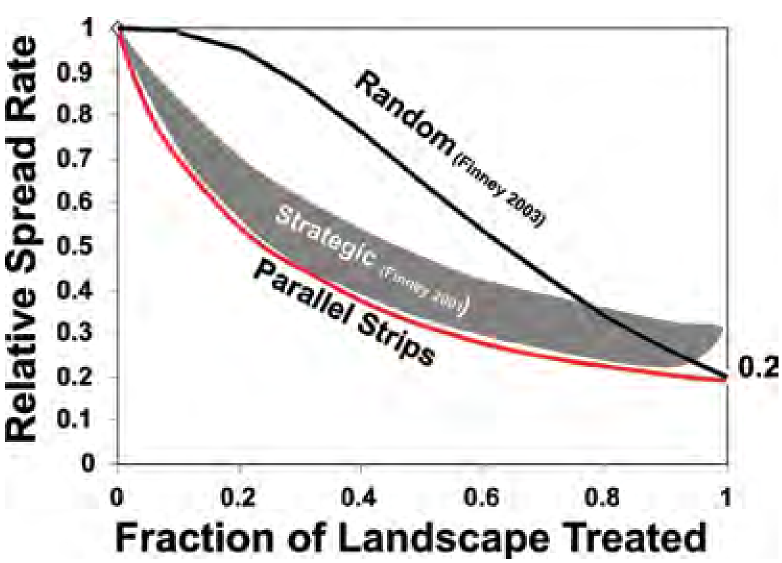


Figure 26. Comparison of large fire growth rate among different spatial fuel patterns (Finney 2003).

not significantly reduce overstory density (tree burn severity less than or equal to level 2). These changes were sufficient to stop the Hayman Fire when it burned into the area in June 2002 even though intense fire behavior was present, facilitated by high winds (48 kmph, 30 mph and greater) and low relative humidities (near or below 10 percent). This treatment was applied within a few months of the fire, thus decreasing the surface fuels substantially (soil burn severity less than or equal to level 3). In this case, the time since treatment, plus the treatment contributed to the change in fire behavior and subsequent burn severity. On the Manitou Experimental Forest (Hayman Fire), mechanical harvesting (selection silvicultural system) reduced the density of all sizes of trees in a pure ponderosa pine forest and concentrated logging slash in large piles. These actions resulted in an easily suppressed surface fire when the Hayman Fire burned into the area. On the other hand, all trees were killed in the Sheepnose Fuels Reduction Project (shelterwood) within the Hayman Fire. Although the stand was heavily thinned from below, heavy surface fuels from non-merchantable logging slash allowed the fire to burn intensely through this stand, potentially damaging the soils and scorching and killing the trees left after the treatment (Finney and others 2003) (fig. 27).

Another example of fuel treatment effectiveness is the Cone Fire (September 2002) in northern California that burned into the Blacks Mountain Experimental Forest. The fire burned approximately 809 ha (2,000 ac) of a study area that was designed to evaluate the effect of varying forest structures (thinnings) on wildlife. When the fire encountered forest structures in which the surface fuels had been burned and the canopy density was reduced, the fire dropped from a crown to a surface fire within the first few yards of entering the treatment units. In areas where the surface fuels were not treated, the fire continued through the unit as a surface fire with variable intensity. There was considerable crown scorch and bark charring in these treatment units with areas of up to 1 ha (2.47 ac) where all trees were dead (Graham and others 2004). These examples show that variability in weather, physical setting, and forest fuels (composition, structure,

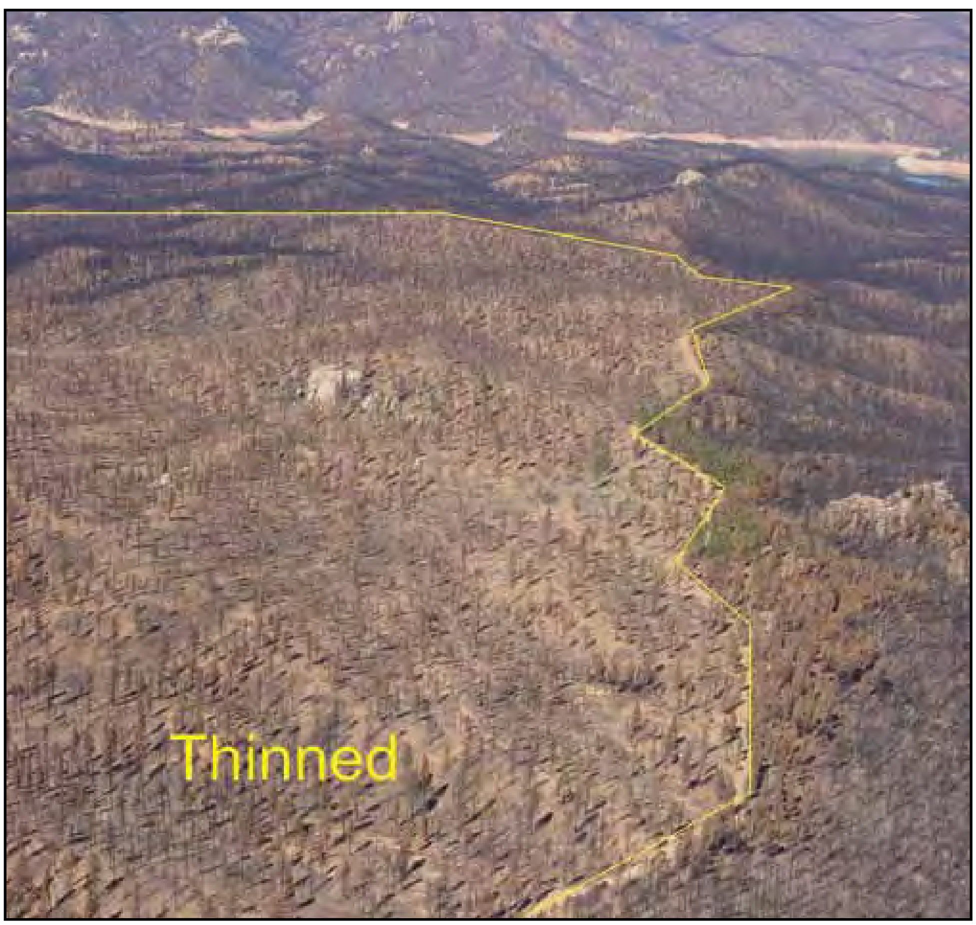


Figure 27. A thinned ponderosa pine stand in which the surface fuels remained after harvest, which resulted in an intense surface fire (tree burn severity level 4) and high soil burn severity (level 5 to 6) (Jain and Graham 2007) when the Hayman Fire burned near Colorado Springs, CO, in 2003.

etc.), influences fire behavior and effects, making it difficult to generalize the effects of treating forests to alter fire behavior and burn severity. However, a key point from these examples is that in many cases, particularly if combined with fire suppression efforts, reduced surface fuels and thinning can significantly limit fire spread and influence burn severity to both vegetation and soils.

Fuel Treatments and Cumulative Watershed Effects

The cumulative effects (either positive or negative) of a fuel treatment are the environmental consequences of the activity when added to the existing landscape condition and any reasonably foreseeable future actions or disturbances. An environmental consequence has both spatial and temporal dimensions and short-term and long-term effects (Reid 1988). Fuel treatments should cumulatively influence vegetation (forest composition and structure function) and be considered as wildfire fuel (Finney 2003; Graham and others 1999a; Graham and others 2004). The cumulative impacts that fuel treatments have on burn severity and fire behavior can range from local to the landscape and watershed levels. As Finney (2003) suggests, fuel treatments can be strategically located and designed for disrupting a fire's progression as it burns through watersheds and landscapes. The cumulative impact of such treatments on potential fire behavior and burn severity would depend on how much of a watershed or landscape was treated, the treatment locations, the timing of the treatments, the kind and intensity of the treatment, and the length of time before both dead and live vegetation would return to pretreatment conditions (for example, the longevity of the fuel treatments and their impact on the fuels complex). In addition, the physical character of the watershed (in other words, slopes, aspect, geology, soils, orientation, and elevation), forest type, PVT, structural stages, seral stages, and patch sizes and juxtapositions would interact with current, planned, and continued maintenance fuel treatments to influence wildfire intensity and burn severity.

Similar to targeting the vertical distribution of fuels to modify fire outcomes, treating different fuel strata would most likely differentially affect an array of wildlife species that are dependent on the vertical forest strata for habitat or habitats of their prey (Reynolds and others 1992; Theobald and others1997; Thomas 1979). Fuel treatments that affect the accumulation and disposition of snags and coarse woody debris, and the retention, disposition, juxtaposition, size, and amount of canopy cover, seral stages, and structural stages occurring on a site, would most likely impact wildlife (Reynolds and others 1992; Thomas and others 1979). These wildlife implications would be important to current planning and the execution of fuel treatments, but future and maintenance treatments would need to include their potential impact on wildlife. Also, depending on the wildlife species and the extent, number, and location of fuel treatment, their effects would most likely be cumulative and could encompass watersheds and large landscapes (Thera and Wildman 2001). Canopy cover also affects water relations within and across watersheds similarly to the potential impacts it has on wildlife. Through evapotransporation, shading, snow interception, snow retention, and other watershed impacts, canopy cover can affect watershed stability (Ziemer and Lisle 1998).

Canopy cover can influence watershed processes (for example, sedimentation and peak flow) through altering rainfall intensity on established snow packs and raindrop intensity on the forest floor; however, these effects must be placed with the context of the soil type, geology, and other biophysical characteristics to understand the cumulative impacts to water quality and quantity. Even though they are not part of fuel treatments, present and future roads are integral to forest management actions. Their location and use for managing fuels inherently affects watersheds and the cumulative effects of fuel treatments on watersheds (Berg 1989; Elliot 2000) (see Chapter 5).

The effect fuel treatments have on visual quality is often an important component of a cumulative effects analysis. Fuel treatment appearance and juxtaposition within a landscape are integral to their visual effects (Bergen and others 1995). The visual quality of a watershed is highly intertwined with the road network along with the character (in other words, prescribed fire, piling slash, timber harvest, etc.) of the treatments and their visual quality. Canopy closure—its shape, location, size, and other attributes—has significant impact on visual quality, both in the short- and long-term, and the cumulative visual impact of such treatments (Brown and Daniel 1986). In addition, the visual quality attributes of a fuel treatment are predicated by whether the treatments are viewed from the foreground, middle ground, or the background.

Fuel treatments should treat forests using silvicultural systems. The silvicultural system documented in a silviculture prescription can disclose the cumulative effects fuel treatments have on vegetation regeneration and development. In particular, silvicultural systems designed for treating fuels should integrate the cumulative and interactive role that insects, diseases, and wildfires play in forest development. Moreover, by designing a silvicultural system and documenting it in a silvicultural prescription, the prescription could display the dynamics of a forest and all of its components over time, which can facilitate the understanding of the cumulative effects of fuel treatments among a wide array of disciplines and stakeholders. By disclosing the effects in a prescription, the effects of fuel treatments on the sustainability of a forest, along with their risks and uncertainties, are identified and documented. In addition, silvicultural systems and their documentation provide a framework for understanding cumulative watershed effects and can be developed into visualizations and other communication tools applicable to a wide range of disciplines and stakeholders.

Conclusion

The cold, moist, and dry forests are all inherently different (figs. 1 through 5). Each forest has a unique suite of forest vegetation, seral and structural stages, and compositions. With this uniqueness comes distinctiveness in where wildfires burn, what they burn, and the effects they have as well as the effect and latitude of forest treatments that modify fire behavior and burn severity. The fuels that wildfires burn range from high canopy fuels to those located on and below the soil surface (fig. 10). Fires can be rather short-lived (minutes), producing little heat to long-lived (months), producing large amounts of heat. The spatial extent of fires can be small (a few m^2, ft^2) to large (100s km^2, mi^2). The burn severity within these extents can range from homogeneous to extremely heterogeneous, leaving a wide variety of vegetation and soil conditions in the after fire (figs. 11, 14, and 15).

Lethal fires (stand replacing), non-lethal (low intensity and severity) surface fires, mixed fires, or a combination, burn through the moist, cold, and dry forests at various intervals and intensities resulting in a variety of burn severities. Fire exclusion has affected all forests, but in general, it has had the largest impact on the dry ponderosa pine/Douglas-fir forests by changing species compositions, structures, and forest floor constituents in many locales. Fire regimes have minimally changed in the moist and cold forests because of fire exclusion but they have greatly altered dry forests.

White pine blister rust (an exotic stem disease) has altered the moist forests by killing western white pine and is progressing to severely alter the cold forests by killing white bark pine (*Pinus albicalus*) (fig. 9). In addition, because of the uniformity of age and size of lodgepole pine and Engelmann spruce in many cold forests, bark beetles are killing millions of trees (fig. 8).

Forests that were once dominated by vegetative structures and compositions relatively resilient to native insects, diseases, and fire regimes are now (2009) more prone to epidemics of insects and diseases and uncharacteristically large and severe wildfires. The changes in forests occurred through a variety of components, ranging from ground level vegetation to high forest canopies and the forest floor and mineral soil that support the vegetation. In addition, all forest vegetative components—live, dead, and in various stages of decay—are part of the fuel matrix, and their composition, development, structure, and juxtaposition at both fine and broad temporal and spatial scales influence how fires burn. These forest characteristics, in concert with physical setting and weather (observable both at fine and broad spatial and temporal scales), ultimately determine fire behavior and burn severity.

The final item influencing the scope and impact of wildfires is the efficiency of suppression activities that, in concert with locale, fuels, and weather, determine the extent and severity of wildfires. Fuel treatments are the only management activity that can influence fire behavior and burn severity. In addition, depending on forest development and values at risk, the frequency of treatments to maintain desired fuel conditions could occur often. It is recommended that surface fuels be treated first, followed by ladder and crown base height treatments and canopy treatments.

Fuel treatments that affect canopy openings most likely impact the amount of water produced within a watershed, and treatments that disturb the forest floor subsequently influence water sedimentation. Also, the location, number, size, age, intensity, and vegetative and forest floor recovery (development) will determine the cumulative effects of fuel treatments within and among watersheds. Because they tend to occupy locales at the higher elevations relative to the surrounding landscapes, cold forests would contain more headwater stream locations. Both the dry and moist forests occupy the lower elevations and often the dry forests border grass and/or shrublands, especially in the central and southern Rocky Mountains. Therefore, the affects that fuel treatments would cumulatively have on watersheds is highly variable among forest types and highly variable depending on the location and juxtaposition of the forests and treatments within and among watersheds.

Forests die, regenerate, and develop in response to disturbances or the lack thereof. Fuel treatments are yet another disturbance that influences how wildfires behave and the subsequent outcomes they produce (figs. 17, 18 and 21). Along with influencing wildfires, such treatments influence both the physical (soils, water, and air) and vegetative properties of a setting. Cumulatively fuel treatments affect many forest attributes, including wildlife habitat, sense of place, invasive species, cultural resources, sensitive plants, and ecosystem services (air and water quality, water quantity, and soil productivity). Moreover, forests are dynamic and, therefore, fuel treatments should change over time and space. We suggest that it is very useful to display fuel treatments and associated outcomes spatially and temporally in addition to identifying the associated risks and uncertainties. At a minimum, these silvicultural systems should demonstrate their cumulative impacts over time on both wildfires and the soil and water resources within and among watersheds. This is the essence of silvicultural systems and forest planning.

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CHAPTER 4.

Tools for Fuel Management

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Introduction

Fuels management is an active term. It is an intentional, planned activity defined by consideration of fire behavior, silvicultural principles, ecological constraints, and the economic and technical limitations of the tools selected to implement the treatment. A forest operation is a tool used to manipulate vegetation or site condition in order to achieve some desired management objectives. Given the wide range of forest operations that can be employed to treat forest fuel, it is imperative to employ a tool that is well-matched to both operational needs and treatment constraints. Selecting a poorly suited tool increases costs and reduces the effectiveness of the operation in achieving the desired outcomes. The selection of a forest operation also plays a critical role in determining the amount and type of cumulative effects associated with the treatment. A tool that is not matched to the terrain or job requirements will likely produce more undesirable impacts.

The purpose of this chapter is to give a basic overview of forest operations for fuel treatments along with information to guide selection of appropriate technologies. Terminology is also important in this discussion. In the biological sciences we have learned that it is important to use scientific names of organisms, rather than common names, to avoid confusion. Unfortunately, many forest operations acquire common names that are contradictory, regionally limited, or non-specific. When someone speaks of a "hydro-ax" treatment, for example, they could mean a vertical-shaft brushcutter, horizontal-shaft masticator, shear feller-buncher, or sawhead feller-buncher. These possible meanings represent very different costs, capabilities, and fuel treatment outcomes. The reference listing at the end of this chapter provides some standard definitions.

Forest Operations for Fuel Treatment

The objective of fuel treatment is to alter fire behavior and severity by modifying properties of various fuel strata in a stand (Graham and others 2004). Treating one strata may improve fire behavior in one respect, but aggravate it in another. For example, activity fuel resulting from a thinning may reduce crown fuel but increase surface fuel loading. A clear fuel treatment prescription should consider effects on the total fire behavior response and clearly specify acceptable treatment outcomes. The primary challenge of selecting appropriate operations, then, is to match the task requirements specified by the fuel prescription to equipment capabilities within constraints of terrain and cost.

Because it should deal with all fuel strata, not just merchantable trees, a fuel treatment operation may involve activities, such as mastication or raking, that have different types of disturbance than conventional forest harvesting. The disturbance effects can be direct (scraping soil to reduce fine surface fuel, for example) or indirect (dozer piling resulting in high temperature burning with resulting hydrophobic soils and poor herbaceous regeneration). The selection of an operation will affect the spatial pattern and total extent of disturbance. The interaction between the type of operation and the sensitivity of the site affects the severity of disturbance and thus the temporal pattern of recovery or effect.

Forest operations for fuel treatment can be broadly divided into two types—in-situ treatment where no biomass or product removal occurs and removal treatments that extract some amount of fuel loading for utilization or disposal outside of the stand. In-situ treatments are selected when there are no economically viable markets for biomass material and it is technically feasible to meet the fuel reduction goals with the material left in place. Removal treatments are selected when it is possible to recover additional value from the treated material or when it is not feasible to treat the fuel in the stand. Resource managers in the western United States have often faced a lack of biomass markets resulting in extensive in-situ piling and burning treatments. More recently however, growing restrictions on burning have motivated efforts to find economically viable removal treatments.

In-Situ Treatments

Fuels treatment can be accomplished within the stand by performing two basic functional tasks: (1) killing selected vegetation and (2) reducing the resulting activity fuel loading to acceptable fire behavior conditions. The selection of an in-situ treatment is probably limited more by the second function than any other factor. Simply rearranging high fuel loading in the stand may not be sufficient to lower fire risk. In fact, shifting fuel loading from ladder fuel or crown strata to surface fuel can significantly aggravate some aspects of fire behavior. Thus most in-situ treatments combine an initial vegetation cutting treatment with a follow-on burn to reduce the volume of activity fuel in piles or scattered slash under controlled burning conditions.

Generally the least expensive in-situ treatment is prescribed fire. Cleaves and others (2000) found that average prescribed burning costs ranged from \$22.80 to \$121.00/acre (1994 dollars, excluding Region 5). Slash burning was generally about twice as expensive as management burning. Prescribed fire mimics many of the ecological functions of natural wildfire. However, the use of this tool has significant limitations. The pattern of vegetative mortality is difficult to control, air quality is adversely impacted, there is risk of escape, and acceptable burning conditions may only occur in limited windows of opportunity. Perhaps the largest limitation to the use of prescribed fire is fuel loading. Many forest areas in the western United States have such high fuel loading that fire is not acceptable without some initial pre-treatment (definition of Condition Class 3).

Chopping, or drum chopping, is a pre-treatment to knock down brush and small trees before broadcast burning. A large steel drum with cutting knives mounted on the face of the drum is rolled across a site (fig. 1). The drums can range in size from 8 to 12-ft wide and can be loaded with water for additional weight. The drum can be towed behind a wheeled or tracked tractor or it can be pulled on a winch cable. As the drum rolls over vegetation, the knives break limbs and stems into shorter pieces. Some trees may even be uprooted in the process. Chopping increases surface roughness by incorporating organic material into the soil; however, there is little soil displacement associated with the treatment. When the drum is towed by a winch line, this treatment can be used on steep slopes with little soil impact.

After several months of drying, the chopped material can be burned. Chopping lowers the fuel bed depth, which reduces flame height. It also increases surface fuel density and continuity, which can make it easier to carry prescribed fire across a site. While this treatment is most often used for residue treatment after clearcut harvesting, it has also been

Figure 1. Drum chopper pulled by a bulldozer.

used effectively for fuel treatment in brush fields, understory control in open pine stands, and as treatment for wildlife habitat improvement.

Chaining is similar to chopping, although it is strictly a clearcut or open field brush fuel treatment. A long, heavy chain, often anchor chain, is connected between two tractors. As the tractors drive forward, the chain knocks over or uproots the brush and trees between the machines. Soil disturbance results from uprooting and the movement of debris with the chain. However, Farmer and others (1999) showed that chaining for pinyon-juniper restoration actually reduced runoff and erosion when compared to untreated areas. A variation of chaining uses a single tractor towing a heavy steel ball connected to the end of the chain. Operating cross-slope on hilly land, the heavy ball pulls the chain downhill and serves as the second anchor. Depending on the fuel loading, chained sites can be burned or left to decompose over time.

Grubbing also kills vegetation by uprooting and breaking plant vegetation to reduce growth. It is principally applied to hard-to-control species that will resprout from cut stumps (for example, salt cedar [*Tamarix* sp.] or alligator juniper [*Juniperus deppeana*]). Grubbing attachments vary from subsoil cutting blades to specially designed grasping attachments for excavators. Extracted plants are piled for disposal or removal. A grubbing treatment creates more severe soil disruption in the areas where plants have been removed, but this soil disturbance is discontinuous compared to a chaining treatment. Grubbing is often the alternative to herbicide treatment.

Manual lopping is another pre-treatment for in-situ fuel management. Chainsaws, brush saws or manual loppers can be used to fell small trees and brush. Lopping may require slashing to reduce piece sizes to specified length or height. Depending on fuel loading, lopping can be combined with scattering (spreading activity fuel across the stand) or handpiling. Generally, lighter fuel loads would be treated by scattering, while heavier loading would necessitate concentrating the slash into piles for burning. Manual lopping results in minimal site impact and can be used on steep slopes. The primary disadvantages of this operation are safety concerns associated with chainsaws and the significant labor requirements to achieve modest production rates. Manual operations are also limited by piece size and stems per acre.

An alternative to manual lopping is to use a swing machine with a brushcutter or sawhead attachment. The approach is to cut small stems quickly and leave them scattered on the site. Feller-bunchers have been used in such applications, but the head is generally not designed to cut or grasp small stems effectively. Mechanical lopping has very little impact on the site. The machine cuts material to the front and drives on the felled mat of slash. This treatment can be applied on a wide range of slopes depending on the capabilities of the base machine. Self-leveling feller-bunchers, for example, are able to operate on 50 percent slopes. Non-leveling swing machines should be limited to gentler slopes. Site disturbance is further reduced because a swing machine can access a 60-ft wide swath from one position.

Lopped material can also be mechanically piled using either a brush rake or a grapple. Brush rakes mount on the front of a wheeled or tracked machine to facilitate pushing debris. The rake teeth on the lower edge of the blade catch residues while minimizing the amount of soil displacement that occurs. However, dozer or tractor piling still causes significant soil disturbance from debris movement. Fuel loading and pile size constraints will determine the number of piles per acre and the required amount of trafficking. Grapple piling is an alternative method that uses a swing machine, either a knuckleboom log loader or a modified hydraulic excavator, to grasp and pile residues. Because grapple piling lifts the material rather than pushing it, soil disturbance is negligible. The resulting piles have very little soil and rock and can be built higher than tractor or hand piles.

Chopping, lopping, and piling are all pre-treatment activities that require subsequent burning to reduce fuel loading. If burning is not possible, however, there are still two options for in-situ fuel treatment—chipping and mastication. Both of these mechanical treatments convert existing fuel into smaller size classes with the objective of removing forest fuel through decomposition. Chipped or masticated material is spread on the forest floor and, as a result of more direct soil contact, has significantly different fuel moisture and burning characteristics than typical forest fuel. It may be possible to use chipping or mastication as a tool to reduce fuel loading prior to a prescribed burn, but more commonly these techniques are used in lieu of burning.

Mobile chippers can be self-propelled or towed machines that reduce trees into chips through slicing. The chips are relatively uniformly sized due to the process and are projected into the stand through a discharge spout. Chippers are fed by a loader and will be most productive if the felled material has been pre-bunched. Towed chippers are typically limited to roadside processing, while self-propelled tracked chippers can operate in the stand. Chipping would be a good alternative to burning if piles had already been constructed.

The direct impacts of chipping include trafficking by the machine and the direct impact of spreading material on the soil surface. Trafficking effects are limited since most of the undercarriage systems produce a ground pressure of less than 7 psi. The effects of the chipped material on soils and water quality are more uncertain. Given the density of wood chips, 20 bone dry tons spread across an acre would be a layer about 1-inch deep. Chips could exclude herbaceous regrowth, alter soil moisture regimes, and change nutrient cycling processes. Chips may also reduce soil exposure to rainfall and thus reduce erosion.

Mastication equipment shreds, rather than chips, standing trees and brush. Unlike mobile chippers, masticators are generally able to fell material. Windell and Bradshaw (2000) provide a thorough review of the range of machines that can be used. There are two basic types of attachments—vertical shaft and horizontal shaft (fig. 2). Either of

Figure 2. Horizontal drum masticator mounted on a tracked tractor.

these can be equipped with pivoting flail-type cutters or rigidly mounted cutting teeth. Masticators can be mounted on nearly every form of base machine including tracked machines, wheeled machines, swing machines, agricultural tractors, or even walking excavators. Johnson (1993) described the use of a walking excavator to masticate material on the Olympic National Forest in areas with slopes exceeding 60 percent. While the shredded material is highly variable given the range of attachments, it is generally coarser and more irregular in shape than chips.

The principle impact of mastication will result from the trafficking of the base machine and the work area defined by the attachment configuration. Direct-mount cutters must traverse nearly the entire stand to implement a treatment. This would approximate the extent of trafficking by a feller-buncher in a clearcut harvest. Boom-mounted cutters, on the other hand, have limited trafficking and soil impact. The type of trafficking disturbance is also a function of the type and size of tire or track that is used. A wheeled machine with wide tires may actually have lower ground pressure than a tracked machine with standard tracks. Careful consideration should be given to the specification of appropriate base equipment for particular soil conditions.

Removal Treatments

If the activity fuel loading from a particular treatment is going to exceed acceptable levels, or if there are marketable products that can be recovered, a removal fuel treatment may be required rather than an in-situ treatment. Like conventional forest harvesting, a removal treatment will involve felling and extraction. However, the type of material removed in a fuel treatment may make the operation radically different in terms of effects and cost than traditional product recovery. For example, skidder load sizes could be smaller and the total number of trips into the stand may be greater when removing small-diameter thinnings. In a fuel treatment, material may be brought out of the stand simply for roadside disposal without the need for product merchandizing that would occur in a sawlog harvest.

Felling for removal can use chainsaws, feller-bunchers, or harvesters. Manual felling is effective for a wide range of tree size and terrain. However, as the number of stems per acre increases, mechanical options become more desirable. Mechanized felling can also move felled material into concentrated bunches for more effective extraction. It is also easier to control the direction of fall and minimize residual stand damage with machines. Like other forest operations, the type of carrier (wheeled or tracked) and the type of attachment mounting (drive-to-tree or swing-to-tree) will determine the primary impacts of felling. Swing machines can operate on steeper slopes and can access a larger area with minimal traffic. Drive-to-tree machines are generally more appropriate for flatter terrain (fig. 3).

Figure 3. Wheeled feller-buncher thinning a ponderosa pine stand.

Felled material can be removed from the stand using skidders, forwarders, cable systems, or helicopters. A basic functional difference among these methods is how the load is moved—skidders drag one end of the load, forwarders carry the load on a wheeled frame, cable systems drag the load but without wheel traffic, and helicopters lift the load completely above the ground. Cost per ton removed increases with increasing extraction distance. This cost-distance curve is a function of load size, operating costs, and travel speed. Skidders will generally be used for distances less than 400 ft; forwarders and cable systems can work effectively at distances of 800 to 1,000 ft; while helicopters can move material several miles.

With any extraction system where repeated cycles are necessary to remove material, the cost per acre is strongly influenced by load size. Collection and removal of slash and brush is particularly challenging because small pieces make it hard to get full payloads. A forwarder load of biomass limbs and tops is about one-third the bulk of a load of logs (fig. 4). If the fuel reduction treatment requires slash removal, the least expensive approach is skidding whole trees. By taking limbs and tops to a roadside attached to the main stem, activity fuel are minimized and the number of trips into the stand to accomplish the treatment is reduced.

Cut-to-length (CTL) systems are considered the lowest impact ground-based harvesting system (fig. 5) and require special consideration. In CTL, trees are felled and processed at the stump using a harvester, and each tree is cut into log lengths that are

Figure 4. Forwarder carrying thinning residues to roadside.

Figure 5. Wheeled harvester performing a fuel reduction thinning.

piled by product. The forwarding function then collects the logs and carries them to roadside. In some CTL operations, trees may be processed in front of the harvester, creating a mat of slash for the machines to travel on. The slash mat, coupled with forwarding, significantly reduces soil disturbance and compaction with CTL. Harvesters also minimize soil impacts by using a boom-mounted attachment to cut and process the trees. In small-diameter treatments, special harvester heads may be needed to effectively handle material.

Material brought to roadside may be separated into product classes in a process called merchandizing. Various log categories can be bucked into specified lengths; pulpwood logs may be debarked and chipped; and fuelwood and residues may be processed through a grinder. Non-merchantable residues can be disposed of at roadside by piling and open burning or with an air curtain incinerator. Roadside merchandizing increases the area of landings and heavy traffic. The more product options involved, the larger the area required for loading, processing, stacking, and transporting. Processing operations also create additional disposal problems—sawdust, bark, butt cuts, and other miscellaneous forms of biomass. Depending on site constraints and the amount of this material, it may be spread on-site or collected for trucking to off-site disposal.

Roadside processing operations can be limited by available area, road access, or the total volume brought to individual landings. If this occurs, trees and biomass can be directly loaded onto a variety of truck types and hauled to a concentration point or woodyard for processing. This "two-stage" hauling can improve operational efficiency by increasing volume and minimizing setup times. Woodyards also reduce in-woods impacts associated with erosion and soil disturbance. If the processed volume is high enough, measures such as gravel surfacing and stormwater management may be warranted.

The final function in removal treatments for fuel management is transportation. Forest roads are recognized as a primary contributor to the water quality impacts associated with forest management. Some type of road access is necessary for all of the operations discussed in this chapter. In-situ treatments are possible with a minimal amount of roading and lower standard roads. Removal treatments impose additional constraints on road spacing and standard. Road spacing affects, or is affected by, the type of extraction system. Skidding requires closer roads, while helicopters can operate at longer distances. The type of product and processing operation determines requirements for road standard. Chipping and grinding produce low-density products that necessitate large transport containers. Right corners or steep grades may exclude this kind of transportation system and thus limit treatment options. The important point to keep in mind is that the road system is part of the forest operation. Transport and access have to match the type of in-woods operation and the impacts of the total system must be considered.

Conclusions

There are many options for forest fuel treatment. Specifications of the prescription, particularly slope requirements and treated material size, may easily exclude some operations from consideration. However, there will generally be a range of feasible alternatives for the resource manager to review. As a project develops, a manager must know:

1. all feasible alternatives that are under consideration (are any options missing),
2. the performance attributes of each option,
3. the tradeoffs among alternatives, and
4. the treatment cost associated with each option.

In general, cost considerations dictate treating fuel as close to the stump as possible. Removal must be justified by fire risk considerations or product values. Forest operations for fuel treatment must satisfy the often conflicting demands of ecological compatibility and economic viability. Minimal impact can be achieved but nearly always at higher cost. Project managers need to balance anticipated impacts of the operation against estimated impacts of the "no treatment" alternative as they select appropriate tools for fuel treatment.

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Terminology

As described in ISO 6814 (ISO 1999), forest machines are defined primarily by the function performed (for example, skidder), then by additional adjectives defining mode of operation (e.g., grapple skidder) and mobility method (that is a tracked grapple skidder). Some of the following terms are from Stokes and others (1989).

Air curtain—a machine that uses forced air to improve combustion of wood in a fire pit or fire box

Bone dry ton—a quantity of wood or biomass weighing 2,000 lbs at zero percent moisture content (also called ovendry ton). This is the typical basis for defining forest fuel loading.

Brush rake—a blade for a skidder or crawler tractor with teeth extending down from the bottom edge

Cable system—an arrangement of winches, rigging, and wire rope used to pull trees or parts of trees from the stand

Chaining—the process of knocking over brush and small trees by dragging a length of heavy chain between two tractors or using a tractor and a heavy weight

Chipping—the process of reducing trees into uniformly dimensioned pieces by slicing

Chopping—the process of knocking down and rolling over brush and small trees with a heavy towed drum that has blades mounted across the face of the drum

Clambunk skidder—a machine that drags trees or parts of trees from the woods to a landing by grasping the load in a large inverted grapple (fig. 6) on the back of the machine (a specialized form of a grapple skidder)

Figure 6. Clambunk skidder.

Clean chips—chips with very low bark content, generally produced by chipping debarked logs, that are marketable for pulp production or high-quality pellet fuel

Cut-to-length—a harvesting system that fells trees, processes in the woods into product lengths, and uses a forwarder rather than a skidder to move wood to roadside

Dirty chips—chips produced by chipping whole trees (also called whole-tree chips)

Dozer piling—the process of pushing residues or felled stems into a pile with a crawler tractor that may be equipped with a straight blade, brush blade or a brush rake

Feller-buncher—a machine that fells trees and accumulates the felled stems into a pile using either a shear head or a sawhead attachment

Forwarder—a machine that carries trees or parts of trees from the woods to a landing

Grapple piling—the process of placing residues or felled stems into a pile with a knuckleboom loader or hydraulic excavator

Green ton—a quantity of wood or biomass weighing 2,000 lbs at field moisture content

Grinder—a machine that coarsely reduces wood or biomass through a shredding action

Grubbing—the process of pushing or pulling to extract most of a plant's root system from the ground

Harvester—a forest machine that fells, delimbs, and bucks trees

Harwarder—a machine that combines the functions of a harvester and forwarder

Hog fuel—coarsely reduced wood material that is intended for direct combustion use

Horizontal grinder—a grinder with a horizontal infeed table

Hotsaw—a high-speed continuous rotation sawhead that is attached to feller-bunchers

Knuckleboom log loader—a swing machine with a hydraulically operated boom and a log grapple attachment to lift and position trees or parts of trees (fig. 7)

Lopping—felling stems to leave them laying on the ground

Mastication—the process of reducing standing trees and brush by shredding or grinding

Merchandizing—the process of separating trees or parts of trees into specified product categories by sizing and sorting

Mobile chipper—a towed machine that reduces trees or parts of trees by chipping

Processor—a machine that takes a felled tree and delimbs and bucks it

Raking—the process of pushing slash or residues into piles, generally windrows, with a brush rake or a towed rake implement

Self-propelled chipper—a tracked chipping machine that can move from place to place

Skidder—a machine that drags trees or parts of trees from the woods to a landing, using either cables or a grapple to grasp the load

Strokeboom delimber—a machine that processes trees into delimbed lengths using delimbing knives and a sliding boom (fig. 8)

Tub grinder—a grinder with a circular rotating top-loaded infeed tub (fig. 9)

Figure 7. Knuckleboom loader (Source: http://www.collectiblereviews.net/).

Figure 8. Strokeboom delimber.

Figure 9. Tub grinder (Source: http://www.collectiblereviews.net/).

Fuel Management and Erosion

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Introduction

As the severity and extent of wildfires increase, efforts to reduce forest densities on public lands by thinning and prescribed burning also are increasing. The intentional suppression of fires in the western United States, beginning in the early 1900s, has altered the extent, frequency, and severity of wildfires (Agee 1993; Baker 1993). Reductions in timber harvest and in grazing, when combined with the suppression of wildfires, have resulted in higher fuel loadings, which increase the risk of high severity wildfires (Norris 1990). Changes in the fire regime may also cause vegetation changes, such as increases in tree stand density, spread of noxious weeds, and tree invasion into grasslands (Arno and Gruell 1986). Any increase in high severity wildfires is of considerable concern because of the potential damage to life and property and the adverse effects on water quality, fish habitat, and other aquatic resources (Keane and others 2002). The goal of most fuel management efforts is to reduce the potential adverse effects due to increased frequency of large, high severity wildfires. Although forest managers attempt to minimize impacts of fuel management activities, the removal of vegetation and the alteration of soil properties due to logging, road building, and prescribed fire may affect site conditions, forest runoff, and water quality (Lindeburgh 1990; Lousier 1990; Rice and Datzman 1981; Tiedemann and others 1979).

Undisturbed forests are an important source of the clean water that is necessary for ecosystem health as well as urban and agricultural uses. Forest vegetation and litter promote high infiltration rates and low levels of overland flow and protect the soil from rainsplash and erosive forces due to overland flow (Baker 1990; Robichaud 2000). This results in high quality runoff, low erosion rates, and low sediment yields (Binkley and Brown 1993; Buckhouse and Gaither 1982; MacDonald and Stednick 2003). Reported sediment yields from undisturbed forests in the western United States are typically around 0.003 t ac^{-1} (0.007 Mg ha^{-1}) (Mg = 10^6 grams or 1 metric tonne), but values up to 11 t ac^{-1} (25 Mg ha^{-1}) have been measured (Stednick 2000). Sediment yields are dependent on several factors such as climate, topography, soil type, vegetation, historic land use, and the dominant erosion processes (Stednick 2000). By altering infiltration rates, evapotranspiration rates, and disturbing the soil, forest management activities—including road construction, timber harvesting, site preparation, and fuel reduction—can greatly increase overland flow rates and sediment yields.

This chapter reviews the effects of roads, thinning operations, and prescribed fire on runoff and erosion, and discusses the current understanding of the cumulative effects of these activities on water yield, stream flow, and sediment production at the watershed scale. Much of the relevant research has been done at the plot, or occasionally, the hill-slope scale; however, simply "scaling up" or summing the measured small-scale effects will not necessarily provide an accurate estimation of cumulative watershed effects. In addition, relatively few studies have specifically measured the effects of fuel treatments. Thus, this review includes the results from selective timber harvest studies and low severity wildfires, as these studies provide the data needed to estimate the likely effects of different fuel treatment activities. The effects of roads, forest thinning, and prescribed fire on runoff and erosion in the western United States are discussed in separate sections, even though many fuel management programs will require more than one of these activities. The final section discusses the cumulative effects of these fuel management activities, as well as the potential cumulative effects at the watershed scale.

Effects of Forest Roads for Fuel Management

Roads are ubiquitous in the forest environment. Forest roads are needed for economical removal of forest products, resource management activities, recreation activities, and public access. From a fuel management perspective, forest roads are needed to conduct prescribed burning, thinning, and timber harvest operations. The majority of forest roads are unpaved. These compacted road surfaces typically have very low infiltration rates and, as a result, generate large amounts of surface runoff (Luce and Cundy 1992; Reid and Dunne 1984; Vincent 1979). Road surfaces are subjected to rainsplash, and the combination of rainsplash with large amounts of surface runoff results in surface erosion rates that are several orders of magnitude higher than the adjacent undisturbed forest (for example, MacDonald and others 2004; Megahan 1978). Research has consistently shown that roads have the greatest effect on erosion of all practices associated with forest management (Megahan and King 2004). Although other forest management activities usually occur on a larger proportion of the landscape, the erosion rates on roads are the dominant source of sediment in most managed forests (Brown and MacDonald 2005).

Forest road effects have been summarized in *Forest Roads: A Synthesis of Scientific Information* (Gucinski and others 2001) and *Roads Analysis: Informing Decisions About Managing the National Forest Transportation System* (Bisson and others 1999). The former compiles current knowledge about the direct physical and ecological effects, indirect and landscape-scale effects, and direct and indirect socio-economic effects of forest roads. *Roads Analysis* is a six-step planning tool designed to evaluate, mostly in qualitative terms, the ecological, social, and economic effects of existing and future forest roads. Thus, only relevant studies addressing runoff and sediment yield from roads used for fuel management are presented.

Effects of Forest Roads on Runoff and Erosion

Effects of roads as structures

Infiltration rates in undisturbed forests are typically at least 1.5 to 3 in h^{-1} (40 to 80 mm h^{-1}) (Robichaud 2000); therefore, few rainstorms or snowmelt events initiate infiltration-excess (Horton) overland flow. In comparison, road components (cut slope, ditch, running surface, and fill slope) have infiltration rates from 0.004 to 0.4 in h^{-1} (0.1 to 10 mm h^{-1}), which frequently results in overland flow.

The flow paths of overland flow depend on road geometry. On insloped roads, water from the cut slope flows to the ditch and reaches the forest floor via a culvert (fig. 1A), and runoff from the fill slope flows onto the forest floor. On outsloped roads, runoff from the cut slope and running surface flows across the road and fill slope to the adjacent forest floor (fig. 1B). The benefits of insloped roads include:

1. the ability to control and direct the concentrated flow;

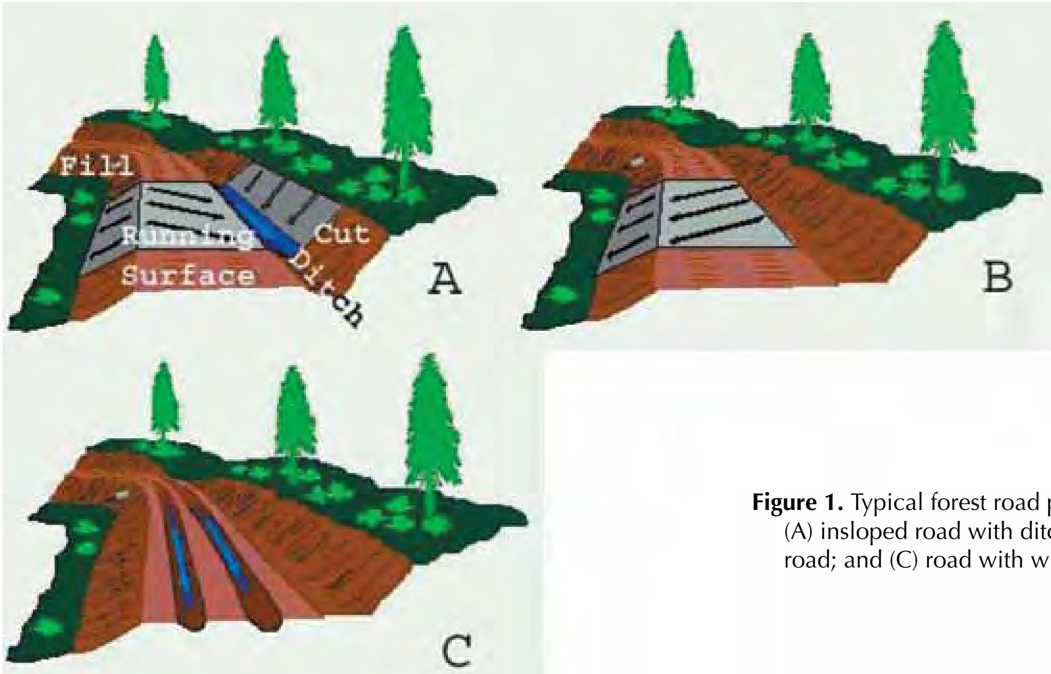

Figure 1. Typical forest road prism structure. (A) insloped road with ditch; (B) outsloped road; and (C) road with wheel ruts.

2. the absence of concentrated flow on the structurally weaker fill slope; and
3. lower risk that a vehicle will slide off the road in wet conditions.

The benefits of outsloped roads include:

1. less concentrated flow because surface runoff immediately drains off the road prism;
2. less undercutting of the hillslope because flow is not concentrated parallel to the hillslope;
3. fewer culverts are used, reducing culvert maintenance and road damage due to culvert failure; and
4. reduced delivery of concentrated flow.

Proper road maintenance is needed to attain the benefits of both insloped and outsloped roads. In particular, if wheel ruts form on the running surface or grading results in a small berm at the edge of the road surface, runoff will be concentrated on the running surface (fig. 1C). On insloped roads, concentrated road surface runoff may result in bypassed relief culverts, while on outsloped roads, road runoff will drain off the road prism as concentrated flow rather than dispersed sheet flow (Foltz 2003).

Effects of road use

Runoff can detach and transport the fine material available on unpaved road surfaces. Without vehicle traffic, the sediment concentration in the road runoff decreases over time. However, vehicle traffic, especially heavy trucks, can crush road surface aggregate material and this generates more fine particles that are available for transport by runoff. In addition, the pressure of vehicular tires on saturated road aggregate can force fine particles from below the surface to move to the surface (Bilby and others 1989; Truebe and Evans 1994). In western Oregon, 20 percent of the material finer than 0.003 in (0.075 mm) diameter was eroded over 3 months from a structurally weak road aggregate that was subjected to 26 in (660 mm) of rainfall and 884 logging truck trips (Foltz and Truebe 1995). The authors concluded that truck traffic generated 11 tons of fines per acre of road surface (24 Mg ha^{-1}).

Road erosion rates generally increase with increased traffic, and heavy vehicles tend to cause more erosion than light vehicles (Megahan 1974; Reid and Dunne 1984). Higher use also is associated with more frequent maintenance operations, and grading increases the amount of available sediment and road erosion rates (Luce and Black 1999). Bilby and others (1989) measured sediment production from two forest roads in southwestern Washington—one mainline road with high traffic and one secondary road with little traffic. Routine maintenance was performed on the mainline road once or twice per week while maintenance was done on the secondary road every 7 to 8 weeks. Sediment production over the 23-week study period was 2.5 times greater for the mainline road (46 t mi^{-1}, 26 Mg km^{-1}) than for the secondary road (18 t mi^{-1}, 10 Mg km^{-1}).

Many techniques used to estimate road sediment production assume factors that influence it (for example, rainfall, traffic, roadway material, etc.) are additive. For example, in the Washington Forest Practices (1995) analysis method, sediment production estimates are independently modified by factors for traffic and surface material. However, a recent study in western Oregon found little difference in sediment production between road plots that were subjected to traffic and those that were recently graded but had no traffic (Luce and Black 2001). They concluded that applying adjustment factors independently overestimated the effect of traffic on new roads or recently maintained roads.

Mitigation of road use effects

The impacts of traffic on sediment production can be mitigated through the use of Best Management Practices (BMP) such as slash filter windrows, rocking the road surface, and ditch armouring (Burroughs and King 1989; Megahan and others 1992). On the Eldorado National Forest, rocking reduced road sediment production by approximately one order of magnitude (MacDonald and others 2004). A comparison of 20 road surface aggregates showed that sediment production was directly proportional to the amount of aggregate finer than 0.24 in (0.60 mm) (Foltz and Truebe 2003). However, some fines are needed to prevent the aggregate from rolling under vehicle tires and being ‘kicked’ off the road by traffic. More recent BMPs include practices such as reducing truck tire inflation pressures, which reduces the contact pressure on the road surface and traffic-induced sediment production. For example, Foltz and Elliot (1996) found that reducing tire pressure from 90 psi (620 kPa) (highway pressure) to 70 psi (480 kPa) reduced sediment production by 45 percent, and a further reduction to 50 psi (350 kPa) reduced sediment production by 80 percent.

Use of low-use, brushed-in roads for fuel management activities

On many miles of low-use forest roads, vegetation has been allowed to grow on the running surface to reduce road-generated sediment. Although no formal assessment has been done, observations of these “brushed-in” roads indicate that sediment production rates are a tenth of the rates for bare roads with traffic. Forest access for fuel management activities will likely require that these brushed-in roads be reopened by scraping the vegetation off the running surface and, to some degree, the cut and fill slopes. In many cases, the increased road availability and use for fuel management activities will also attract additional recreational traffic, including off-highway vehicles (OHV). While OHVs, particularly all-terrain vehicles (ATVs), are lighter than trucks or automobiles, they loosen and move road surface material laterally making it available for subsequent transport (Iverson 1980).

Road Obliteration Effects on Runoff and Erosion

The type and frequency of fuel management activities must be factored into forest transportation plans, as this may affect both road construction and road removal. The removal of forest roads from service, or “decommissioning,” is usually accomplished by blocking the road entrance and restoring the road prism to a more natural state (USDA Forest Service 2000). Road obliteration is the most complete form of decommissioning, and this involves ripping the road surface, removing culverts, re-establishing stream channels, reshaping the roadbed to match the hillside contour, and planting vegetation.

Like road construction, road obliteration typically causes a spike in sediment production that decreases rapidly after the activity ends. Brown (2002) measured the sediment generated during road obliteration at five stream crossings with wooden culverts in central Idaho. Peak suspended sediment concentrations ranged from 2.9 to 68,400 mg L^{-1}, depending on the number of straw bales placed in the stream and the flow diversion channel. Foltz and Yanosek (2005) reported sediment yields of 4.4 to 375 lb (2 to 170 kg) from the removal of each of three corrugated metal pipe culverts in central Idaho. The removal of these culverts did cause the instantaneous turbidity levels to exceed the 50 NTU standard for aquatic habitat (IDEQ 1994) immediately below the culverts, but this standard was not violated 0.5 miles (0.7 km) downstream. The 10-day criteria of 25 NTU (IDEQ 1994) was not exceeded at any of the three crossings, as the peaks in turbidity caused by mechanical activity in the stream typically decreased by an order of magnitude within 2 hours.

Forest Road Effects at the Watershed Scale

The watershed-scale effects of forest roads are much more difficult to detect than the effects at the site or road segment scale for several reasons. First, forest roads rarely exist without some accompanying timber harvest activities. In some watershed studies, the road network was installed 1 or more years prior to logging to identify the effects of the roads on runoff, sediment yields, or water quality (for example, Lewis 1998; Troendle and others 2001). However, only 1 to 3 years of data were collected before timber harvest began, and this short time period made it difficult to detect a distinct impact for small- to moderate-sized rain events (Bunte and MacDonald 1999; Loftis and others 2001).

Three studies, conducted in snowmelt-dominated climates where the roaded area was only 2 to 4 percent of the watershed area, were able to isolate forest roads and measure the impact on watershed runoff rates. Two paired-catchment studies in Colorado have shown no detectable change in runoff due to just the roads (MacDonald and Stednick 2003). Similarly, in the third study, the road system did not alter annual water yield or peak stream flows from a 4,035-ac (1,633-ha) watershed in central Idaho (King 1994).

A second issue is the connectivity of roads to the stream network. Reported road erosion rates for the western United States vary from negligible amounts to 1,400 t mi^{-2} yr^{-1} (500 Mg km^{-2} yr^{-1}) (table 1). However, road erosion rates measured at the plot or road-segment scale cannot be directly extrapolated to the watershed scale (in other words, tens to thousands of acres [hectares]) because not all of the runoff and sediment may be delivered into and through the stream network (MacDonald 2000). For roads immediately adjacent to a stream, much of the road-generated sediment is delivered directly to streams. However, when a sufficient forest buffer is located between the road and the stream, much of the sediment may be deposited on the forest floor (Megahan and Ketcheson 1996). Recent multi-agency management agreements (PACFISH to protect anadromous fish and INFISH to protect inland native fish) require a 300-ft (90-m) forest buffer between roads and fish-bearing streams. This buffer width reflects a “consensus” opinion among scientists and managers, and it is designed to minimize the delivery of runoff and sediment to the stream network.

In addition to road location, road-stream connectivity can be increased because the concentrated runoff from roads can increase the drainage density (Croke and Mockler 2001; Montgomery 1994; Wemple and others 1996). Wemple (1994) reported that nearly 60 percent of the road network was hydrologically connected to the stream network in two adjacent 5th order basins in western Oregon (mean precipitation is 89 in or 2,260 mm per year). On the Olympic peninsula of Washington, the average annual precipitation is 153 in (3,890 mm), and 75 percent of the roads were reported to be connected to the stream network (Reid and Dunne 1984). Bilby and others (1989) found that 34 percent of the roads were connected to the streams in southwestern Washington (precipitation amounts not reported). In western Oregon, the road-stream connectivity was reported to be 23 to 47 percent in areas with a mean annual precipitation of 20 to 100 in (500 to 2,540 mm) (Skaugset and Allen 1998). The study sites in the Wemple, Reid, and Bilby studies were roads that had been constructed between the 1950s and 1970s, while the roads examined in the Skaugset study were constructed in the 1980s

Table 1. Road erosion rates from selected studies in the western United States (after MacDonald and Stednick 2003 and Elliot and Foltz 2001).

Reference	State	Traffic, slope, running surface	Erosion rate reported*	Sediment yield (lb ft^{-2}-yr^{-1})	Sediment yield (kg m^{-2}-yr^{-1})
Road surfaces					
Megahan and Kidd 1972	ID	Variable use, slopes	51 t mi^{-2}-yr^{-1}	1.5	7.3
Megahan 1975	ID		20 Mg ha^{-2}-yr^{-1}	0.41	2.0
Wald 1975	WA	Moderate traffic, 6.4%	44.2 t mi^{-2}-yr^{-1}	1.4	6.6
		Low traffic, 3.0%	3.4 t mi^{-2}-yr^{-1}	0.1	0.5
Bunkhouse and Gaither 1982	OR		0-7 Mg ha^{-2}-yr^{-1}	0.0-1.4	0-7
Reid and Dunne 1984	WA	Heavy traffic, 10%	500 Mg km^{-2}-yr^{-1}	20	100
		Moderate, 10%	42 Mg km^{-2}-yr^{-1}	1.7	8.5
		Light, 10%	3.8 Mg km^{-2}-yr^{-1}	0.16	0.77
		Abandoned, 10%	0.51 Mg km^{-2}-yr^{-1}	0.020	0.1
Bilby 1985	WA	Mostly high use, 1%	0.0052 Mg m^{-2}-yr^{-1}	1.1	5.2
Vincent 1979	ID	Light use, 6.3-13.4%	9.3-31 t ac-yr^{-1}	0.47-1.6	2.3-7.6
Foltz 1996	OR	Heavy, 12%, good aggregate	18 Mg ha^{-1}	**	
		Heavy, 12%, marginal aggregate	132 Mg ha^{-1}	**	
Luce and Black 1999	OR	Aggregate	30-99 Mg ha^{-1}	**	
Luce and Black 2001	OR	Aggregate and ditch maintenance	0.05-4.8 Mg ha^{-1}	**	
Cutslopes					
Wilson 1963	OR	6- to 7-yr old cutslopes	153 Mg ha^{-2}-yr^{-1}	3.1	15
		new cutslopes	370 Mg ha^{-2}-yr^{-1}	7.6	37
Dyrness 1970, 1975	OR	5-yr old cutslopes	0.5 cm yr^{-1}	1.5	7.5
		1-yr old cutslopes	0.7 cm yr^{-1}	2.3	11
Megahan 1980	ID	45-yr old cutslopes, soil	0.01 m^3 m^{-2}-yr^{-1}	3.1	15
		45-yr old cutslopes, granite	0.011 m^3 m^{-2}-yr^{-1}	3.5	17
Megahan and others 1983	ID		11 mm yr^{-1}	3.3	16
Megahan and others 2001	ID	Cover 0.1-89%; gradient 55-104%	0.1-250 Mg ha^{-2}-yr^{-1}	0.0020-5.1	0.01-25
Fillslope					
Bethlahmy and Kidd 1966	ID	Unvegetated fillslope	94 Mg ha^{-2} for 10 mo	2.3	11
Megahan 1978	ID	12-yr old fillslope	12 Mg km^{-2}-yr^{-1}	0.25	1.2

* Mg = 10^6 grams or 1 metric tonne
** Short measurement period precludes extrapolation to annual yield.

to 1990s when forest road design was more of an issue. In general, the connectivity of the road network to streams will be a function of the precipitation regime, road design, and road maintenance.

Effects of Forest Thinning for Fuel Management

Numerous studies have evaluated the effects of timber harvest on runoff, water quality, erosion, and sediment yields (Binkley and Brown 1993; Stednick 2000). Most studies have focused on commercial harvests using relatively severe treatments such as clearcuts, patch cuts, or heavy selective cuts, while few studies have focused on forest thinning operations. Fuel management treatments are more similar to thinning operations, such as selective single tree selection or group cuts, rather than patch or clearcuts. This means that the observations and conclusions presented here are based partly on inference and extrapolation from studies of more intensive forest harvest operations, and to the extent possible, on the limited data from thinning studies that more closely correspond to the amount of disturbance that might be expected from fuel reduction treatments.

Effects of Forest Thinning on Runoff

Changes in annual water yields

The removal of forest cover decreases interception and transpiration, and in wetter areas, this generally increases annual water yields (Bosch and Hewlett 1982; MacDonald and Stednick 2003). The increases in annual water yield following forest harvest are usually assumed to be proportional to the amount of forest cover removed, but at least 15 to 20 percent of the trees must be removed to produce a statistically detectable effect (MacDonald and Stednick 2003). In areas where the annual precipitation is less than 18 to 20 in (450 to 500 mm), removal of the forest canopy is unlikely to significantly increase annual water yields (Bosch and Hewlett 1982). In drier areas, the decrease in interception and transpiration is generally offset by the increase in soil evaporation, and there is no net change in runoff as long as there is no change in the underlying runoff processes (for example, a shift from subsurface stormflow to overland flow due to soil compaction) (MacDonald and Stednick 2003). For example, removing 100 percent of the forest cover in a snow-dominated area with a mean annual precipitation of 21 in (530 mm) resulted in an initial water yield increase of 1.1 in yr^{-1} (28 mm yr^{-1}) (Bates and Henry 1928), while a 24 percent reduction in forest cover in a snow-dominated area with a mean annual precipitation of 34 in (871 mm) caused an initial water yield increase of 3 in yr^{-1} (76 mm yr^{-1}) (Troendle and others 2001). In wetter environments, the combination of clearcutting and roads may increase annual water yields by 20 in (500 mm) or more.

Extrapolating from these and other results suggest that relatively heavy thinning operations can increase annual water yields in wetter environments. No measurable increase in runoff can be expected from thinning operations that remove less than 15 percent of the forest cover or in areas with less than 18 in (450 mm) of annual precipitation. Since evapotranspiration rapidly recovers with vegetative regrowth in partially thinned areas, any increase in runoff due to thinning operations is likely to persist for no more than 5 to 10 years.

Runoff timing and peakflows

The timing of the increase in runoff due to forest harvest is important because of the potential impact on water supplies, sediment transport capacity, bank erosion, and aquatic ecosystems. If forest harvest only increases low or moderate flows, one would expect little or no change in channel erosion or sediment yields. An increase in larger flows provides a mechanism for increasing annual sediment yields (Lewis 1998; Schumm 1971).

The timing of the increased runoff due to harvesting will vary with the hydrologic/physiographic characteristics and climate regime. If the climate is dry in summer and rainy during the winter, the largest increase in runoff will occur in the fall to early winter. This is due to the increase in soil moisture in late summer after forest harvest and the resulting increase in runoff efficiency because less precipitation is needed for soil moisture recharge. Runoff rates also will increase throughout the winter due to the reduction in interception.

In snow-dominated environments, nearly all of the increase in runoff will occur in early spring. As in rain-dominated environments, forest harvest reduces summer evapotranspiration and increases the amount of soil moisture carryover. Less snowmelt is needed for soil moisture recharge, so more of the early season melt is converted into runoff. The reduction in forest canopy also increases the amount of solar radiation that reaches the surface of the snowpack and the transfer of advective heat, and these changes increase the rate of snowmelt and may slightly accelerate the timing of peak runoff (for example, MacDonald and Stednick 2003; Troendle and King 1985).

An analysis of the changes in flow duration curves due to forest harvest indicates that low flows generally experience the largest percentage change, while the higher flows experience the largest absolute change. In other words, most of the additional water comes during the higher flows (Austin 1999). In rain-dominated areas, the percent increase in high flows is generally much less than the percent increase in low flows. For example, Austin (1999) reported that in rain-dominated areas, the combination of roads and intensive forest harvest increased the larger daily flows by about 10 to15 percent. Studies in the cold snow zone in the Rocky Mountains indicate that 100 percent harvest will generally increase the size of the annual maximum flow by about 40 percent (MacDonald and Stednick 2003). If less than 100 percent of the vegetation is removed, the increase in the size of the annual maximum flow is roughly proportional to the percent of forest cover removed (MacDonald and Stednick 2003). For example, peak flows increased by 20 to 28 percent after removing 30 to 50 percent of the forest canopy in northern Arizona, while peak flows increased by 90 percent after removing 77 percent of the canopy and 170 percent in a clearcut watershed (Brown and others 1974). In areas dominated by snowmelt, there may be little change in low flows (Bates and Henry 1928; Troendle and King 1985).

Several studies indicate that the increase in runoff due to forest management can increase suspended sediment concentrations and annual sediment yields. Suspended sediment loads increased after harvesting 10 sub-watersheds in the North Fork of Casper Creek in northwestern California. This increase was attributed to the increase in channel shear stress and transport capacity as a result of the increase in runoff (total flow volume), as the harvest units, roads, and landings were restricted to upslope locations (Lewis 1998). On the Fraser Experimental Forest in Colorado, the harvest-induced increases in high flows can account for most of the observed increases in annual sediment yields (Troendle and Olsen 1993). These results suggest that flow increases due to forest harvest can increase in-channel erosion and can be the dominant cause of an increase in sediment production, particularly when the amount of ground disturbance is kept to a minimum.

Effects of Forest Thinning on Erosion

Types of timber management activities used for thinning

The effects of forest harvest activities on erosion and sediment yields depend on techniques used, site characteristics, storm event of concern, and skills of the equipment operators. In decreasing order of disturbed area, some common forest harvest practices are clear-cutting, seed tree and shelterwood harvests, single tree selection, and group selection. Light or moderate thinning operations typically cause much less ground disturbance than clear-cuts or shelterwood cuts. On the other hand, a relatively intense thinning operation may require access to more of a stand than a harvest using patch cuts or group selection even though a smaller volume of timber is being cut (Haupt and Kidd 1965). Erosion rates tend to be positively correlated with percent bare soil and the amount of surface disturbance, and these two factors generally are proportional to the number of trees being harvested (Haupt and Kidd 1965). In general, erosion rates are acceptably low when the proportion of bare soil is less than 30 to 40 percent (Benavides-Solorio and MacDonald 2005; Gary 1975; Swank and others 1989).

Like other forest management practices, thinning generally requires road access. The amount of roads needed for commercial thinning will vary with the spatial distribution of the tress being thinned and the yarding techniques used. Non-commercial thinning may require fewer roads than most other forest harvest activities because yarding is not necessary. When evaluating the effects of thinning relative to unmanaged forests, it is essential to consider the effects of the road network—including new road construction, changes to existing roads, and the increase in traffic—in addition to the effects of the thinning activities.

Effects of felling on erosion

Felling is the action of cutting down a tree by machine or hand. Mechanized fellers cut a tree down with a saw blade and then de-limb the tree. Some machines are designed to collect the trees using a specialized attachment (feller-buncher). Mechanized felling is faster and less hazardous than hand-felling, but the trees need to be under a certain diameter and the area has to be machine accessible. Mechanized fellers can disturb and compact the soil, and the use of these machines is a potential source of erosion. However, because they do not drag the logs on the ground, they often generate less erosion than log skidders.

The effects of felling on erosion generally have not been studied independent of yarding. Hand felling can be accomplished by one person with a chainsaw, and the amount of soil disturbance from this activity generally is considered negligible. A comparison of clearcut and thinned plots to control plots showed that hand-felling without mechanized yarding caused minimal surface disturbance and no increase in erosion (McClurkin and others 1987).

Non-commercial thinning to reduce fuel loads is being done on an increasingly large scale using masticating machines. These machines are usually large, rubber-tired or tracked skidders with a mulching or wood grinding attachment such as a Hydro-Ax or a Bull-Hog. Some machines are designed to masticate standing trees, while others fell the trees before masticating the material. Like mechanized fellers, the movement of masticating machines can disturb or compact the soil and thereby increase the potential for erosion. The shredded wood that remains after these operations may increase the amount of ground cover and reduce the erosion potential. The effects of these treatments, including impacts on vegetation, have not been rigorously evaluated.

Effects of yarding on erosion

The amount of disturbed area and bare soil due to thinning and forest harvest will depend largely on the amount and type of yarding activities. Ground-based tractor-yarding generally necessitates an extensive network of skid trails and roads, while full suspension cable yarding will cause much less ground disturbance and generally requires a less dense road network. Tractor yarding generally produces the greatest amount of site disturbance, followed by jammer, high lead cable, skyline, and helicopter yarding (Rice and others 1972; Stednick 1987). This list is slightly misleading in that the amount of disturbance due to logging was not separated from the disturbance due to roads. In some situations, jammer logging (cut trees are cable yarded using a truck-mounted boom) can result in up to 29 percent more road area than tractor-logging (Rice and others 1972), and the higher road density can greatly increase the total erosion rate from the project area. In a northwestern California study, a categorical variable to represent the type of yarding helped to more accurately predict post-harvest erosion rates (Rice and Furbish 1981).

Although thinning a stand of trees to a desired density requires access to the entire stand, non-commercial thinning generally requires little or no yarding and can be one of the least disturbing forest management practices. Commercial thinning requires yarding methods appropriate for smaller trees, such as small skylines with light cables and short towers, small crawler tractors, rubber-tired skidders, horses, tractor-mounted winches, or specialty yarding machines (Small Woodlands Program of BC 2002). The amount of disturbance caused by yarding will depend on the site characteristics, timing of yarding, and the percent of the stand that is being thinned. In most cases, the amount of disturbance from commercial thinning will be similar to selective harvest techniques.

Review and integration of erosion rates from managed watersheds

Several recent studies have summarized erosion and sediment yields from managed and unmanaged forests. These include a summary of erosion and sediment production data from different site preparation and timber harvest activities in the United States (Stednick 2000) and suspended sediment data from areas subjected to forest harvest and road construction (Binkley and Brown 1993). These reviews indicate a general lack of data for non-commercial thinning operations (table 2) and a relatively rapid decline in surface erosion rates after timber harvest activities. For example, in central Idaho, 90 percent of the erosion from skyline and jammer logging occurred within the first 2 years after harvest (Megahan 1975).

Table 2. Changes in sediment and streamflow due to forest management activities. Data are restricted to studies in the western United States.

Reference	Location	Type of logging	Yarding method	Plot size or area	Harvest percent	Sediment response	Streamflow response
Lewis 1998	South Casper Creek, CA	Selective cut	Tractor	424 ha	65%	Suspended sediment increased for 1 yr after road building (335%) and in the 6 yrs following harvest there was a 212% increase in sediment load	NA
Lewis 1998	North Casper, CA	Clearcut	80% cable yarded	~30-473 ha	48%	89% increase in sediment load after treatment	NA
Lopes and others 2001	Beaver Creek, AZ	Ponderosa pine clearcut/ windrowed and burned	Bulldozer	184 ha	100%	Where: Suspended Sediment = k a (Streamflow Discharge) $^{b+1}$, the a and b coefficients were increased by 440 and 54% respectively	30% increase in annual water yield
Lopes and others 2001	Beaver Creek, AZ	Ponderosa pine cut into irregular strips and 25% of residual thinned	Bulldozer	546 ha	40%	Where: Suspended Sediment = k a (Streamflow Discharge) $^{b+1}$, the a and b coefficients were increased by 97 and 44% respectively	20% increase in annual water yield
Lopes and others 2001	Beaver Creek, AZ	Large pinon-junipers were pulled out with bulldozer	Bulldozer	131 ha	100%	Where: Suspended Sediment = k a (Streamflow Discharge) $^{b+1}$, the a and b coefficients were increased by 44 and 21% respectively	No significant increase in annual water yield
Lopes and others 2001	Beaver Creek, AZ	Herbicide on pinon-juniper	None	147 ha	100%	Where: Suspended Sediment = k a (Streamflow Discharge) $^{b+1}$, the a and b coefficients were increased by 58 and 27% respectively	160% increase in annual water yield
Heede 1987	S. Fork Thomas Creek, AZ	Patch cuts and group selection (moderately disturbed)	Crawler Tractors	0.01 - 0.06 ha	NA	Sediment delivery was 10.8 kg ha^{-1} yr^{-1} compared to 8.05 in control	Overland flow was 89% higher than 1.31 mm yr^{-1} control
Heede 1987	S. Fork Thomas Creek, AZ	Patch cuts and group selection (severely disturbed)	Crawler Tractors	0.01 - 0.06 ha	NA	Sediment delivery was 31.8 kg ha^{-1} yr^{-1} compared to 8.05 in control	Overland flow 240% higher than 1.31 mm yr^{-1} control
Williams and Buckhouse 1993	Ag. Research Center, OR	Cut to stand density of 1 stem per 69 m^2	Rubber tired skidders	1 by 5 m plot	NA	No sediment	No runoff (up to 39 mm hr^{-1} rain intensity)
Williams and Buckhouse 1993	Ag. Research Center, OR	Cut to stand density of 1 stem per 132 m^2	Rubber tired skidders	1 by 5 m plot	NA	No sediment	No runoff (up to 39 mm hr^{-1} rain intensity)
Megahan 1975	Horse Creek, ID	Clearcut	Skyline	12 ac total	100%	Sediment rate was 33 yd^3 yr^{-1} mi^{-2} compared to 21 yd^3 yr^{-1} mi^{-2}	NA
Megahan 1975	Horse Creek, ID	Clearcut	Jammer	10 ac total	100%	Sediment rate was 33 yd^3 yr^{-1} mi^{-2} compared to 21 yd^3 yr^{-1} mi^{-2}	NA
Madrid unpublished	Lincoln National Forest, NM	Pre-commercial thinning with slash piled	None	10-1 m^2 plots	Low intensity	Dry simulation = 102% increase in sediment yield from 1.12 to 2.26 kg ha^{-1}; Wet simulation = 200% increase from 0.43 to 1.28 kg ha^{-1}	No significant change in runoff
Madrid unpublished	Lincoln National Forest, NM	Pre-commercial thinning with slash scattered	None	10-1 m^2 plots	Low intensity	Dry simulation = 59% increase from 1.12 to 1.77 kg ha^{-1}; Wet simulation = 123% increase from 0.427 to 0.951 kg ha^{-1}	No significant change in runoff
Brown and MacDonald unpublished	Upper South Platte, CO	Pre-commercial thinning	None	from 0.094 to 0.65 ha	25%	No sediment in control or thinned swales	NA

An ongoing hillslope-scale study is comparing ground cover and erosion rates from intensively thinned areas to undisturbed areas in ponderosa pine forests in the Colorado Front Range. In the treated stands, the use of a Hydro-Ax redistributed existing litter and scattered wood chips over 21 percent of the surface area (Brown and MacDonald 2005; Brown and others 2005). The mean surface cover of wood increased from 3 to 6 percent, but the thinning did not significantly affect the proportion of litter, rock, or live vegetation cover. Percent bare soil increased from 9 to 15 percent, but this was only marginally significant ($p = 0.08$). The 48 plot-years of data from 2003 to 2005 show no evidence of surface runoff or hillslope erosion from either the thinned or the control plots, even though the steepest plots (>50 percent slope) were subjected to a 1.6 in (42 mm) storm with a maximum 30-minute intensity of 2.4 in h^{-1} (61 mm h^{-1}) (Brown and MacDonald 2005; Brown and others 2005).

In northern New Mexico, a series of rainfall simulations (6 in h^{-1} or 150 mm h^{-1} for 1 hour on successive days) was done on control plots, lightly thinned plots with the slash piled, and lightly thinned plots where the slash was scattered (Madrid 2005). The results show that thinning had no effect on runoff or the amount of bare soil, but the mean sediment yields from the second (wet) simulation on the thinned plots was two to three times higher than the mean sediment yield of 1.8 t ac^{-1} (4 Mg ha^{-1}) from the control plots (Madrid 2005).

The spatial pattern and location of the harvest activities relative to the stream network influences the amount of sediment that is delivered from severely disturbed areas. Forest management activities often generate a mosaic of severely disturbed areas (clearcuts, skid trails, and landings) and relatively undisturbed areas (streamside management zones). The former usually are considered sediment source areas and the latter usually serve as sediment sinks. If the runoff and sediment yields from the source areas are less than the absorption capacity of the downslope sediment sinks, it follows that there will be little or no change in runoff and sediment yields at the watershed scale. It is important to recognize that watershed-scale changes in sediment yields tend to be correlated with the amount of disturbance in a watershed, but a high level of disturbance does not always mean that there will be a detectable change in sediment yields at the watershed scale (Haupt and Kidd 1965).

Effects of Prescribed Burning

Prescribed burning is the controlled use of fire to achieve specific forest management objectives (Walstad and others 1990). Prescribed fires are often used after timber harvest operations to dispose of flammable residues and reduce the wildfire risk. Prescribed fires are also used to facilitate tree planting by removing logging slash, debris, and undesirable vegetation; reduce the risk of destructive insect infestation; create suitable environmental conditions for the establishment and growth of desired tree species; manipulate secondary plant succession to favor the development of preferred species; and increase production of understory vegetation for wildlife (Walstad and Seidel 1990). Prescribed burning is increasingly being used to reduce fuel loads in forests.

Fire Effects on Forest Floor

The major factor that determines the effects of burning on runoff and erosion is the amount of disturbance to the surface organic material (commonly referred to as duff or forest floor) that protects the underlying mineral soil. The effects of burning can vary from merely removing some of the litter (low burn severity) to totally consuming the duff layer and organic matter in the upper soil layers (high burn severity). If the duff is completely consumed by a fire, the mineral soil is exposed to rain splash and overland flow (Soto and others 1994; Wells and others 1979). Any loss of organic matter in the uppermost layers of the mineral soil will alter the structure of the surface soil, and the resultant disaggregation of the soil particles can greatly increase its susceptibility to

Table 3. Published first-year sediment losses after prescribed fires (after Robichaud and others 2000).

Dominant plant, location	Treatment	Sediment loss ($t\ ac^{-1}$)	Sediment loss ($Mg\ ha^{-1}$)	Reference
Ponderosa pine, CA	Control	<0.0005	<0.001	Biswell and Schultz 1965
	Prescribed fire	<0.0005	<0.001	
Chaparral, CA	Control, steep slope	0.0009	0.002	DeBano and Conrad 1976
	Prescribed fire, steep slope	3	7	
	Control, gentle slope	0	0	
	Prescribed fire, gentle slope	1	3	
Chaparral, AZ	Control	0	0	Pase and Lindenmuth 1971
	Prescribed fire	2	4	
Larch/Douglas-fir, MT	Control	<0.0004	<0.001	DeByle and Packer 1972
	Slash burn	0.07	0.2	
Ponderosa pine, CO	Low severity	0.16	0.37	Benavides-Solorio 2003
	Moderate severity	0.058	0.13	
	High severity	0.36	0.81	
Ponderosa pine, CO	Low severity	0.30	0.67	Benavides-Solorio 2003
	Moderate severity	0.49	1.1	

erosion (Brown and others 1985; DeBano and others 1998; Robichaud and Waldrop 1994; Ryan 2002; Wells and others 1979). In some vegetation types, a moderate or high severity fire can change or induce water repellent soil conditions at or near the soil surface (DeBano 1981; Huffman and others 2001; Robichaud and Hungerford 2000). The fire-induced soil water repellency and disaggregation of soil particles will reduce the infiltration rate of the mineral soil, and the loss of organic material reduces the water storage capacity above and in the mineral soil. These changes result in increased runoff, especially from short duration, high intensity rain events (Baker 1990). Prescribed fires are generally designed to leave some residual duff to protect the mineral soil and maintain high infiltration rates, which minimizes potential erosion (table 3).

The amount of duff consumption during prescribed fires is controlled primarily by the thickness and water content of the duff prior to burning (Brown and others 1985; Frandsen 1997; McNabb and Swanson 1990; Reinhardt and others 1991; Wells and others 1979). For example, the First Order Fire Effects Model (FOFEM) uses 22 different algorithms to predict percent duff consumption, depth of duff consumed, and percent of mineral soil exposed, and all but one use pre-burn duff thickness and duff moisture content as input variables (FOFEM, version 5.0). Fire managers use models such as FOFEM to help design prescribed burns that will consume much of the fuel load while leaving a protective duff layer over the mineral soil.

Fire Effects on Runoff and Erosion

Prescribed fire

Prescribed fires create a highly variable mosaic of burn severity, duff consumption, and unburned area (Robichaud 2000; Robichaud and Miller 1999). This spatial variability in postfire surface conditions results in spatially varying runoff and erosion rates. Post-prescribed fire variations in runoff and erosion have been assessed by conducting rainfall simulation on small plots. For example, high infiltrations rates and low sediment yields were reported after a spring season, low burn severity prescribed fire in northern Idaho (Robichaud and others 1994) (table 4). In another study, two low burn severity prescribed fires were conducted after timber harvest—one in Idaho (Hermada) and one in Montana (Slate Point), and postfire assessments indicated that only 5 percent of the Hermada site and 15 percent of the Slate Point site had burned at high severity (Robichaud 1996; Robichaud 2000). As expected, the initial infiltration rates in the high burn severity were lower than in the unburned and undisturbed areas. More importantly,

Table 4. Sediment yields from rainfall simulation studies on low severity burned plots (after Benavides-Solorio and MacDonald 2001; Johansen and others 2001; Robichaud 2000).

Dominant plant type, location	Slope [Time since burn] (%)	Rainfall intensity [Rainfall duration] (in h^{-1}) [(min)]	(mm h^{-1}) [(min)]	Sediment yield (t ac^{-1} in^{-1})	(kg ha^{-1} mm^{-1})	Reference
Sagebrush-juniper, CA	— [1 year]	2.6 [30-60]	65 [30-60]	0.034-0.083	3.0-7.3	Simanton and others 1986
Mixed conifer, ID	13-27 [within days]	2.0 [30]	50 [30]	0.028-0.12	2.5-11.0	Robichaud and others 1994
Pinon-juniper, NV coppice	5-8 [1-2 months]	3.3 [60]	84 [60]	0.045-0.11 0.035-0.15	4.0-9.7 3.1-12.8	Roundy and others 1978
Pinon-juniper, NV interspace	[1 year]			0.090-0.29 0.13-0.36	7.9-26.0 11.9-32.0	
Ponderosa pine, CO	21-22 [1-2 months]	3.1 [60]	80 [60]	0.056-0.092	4.9-8.1	Benavides-Solorio and MacDonald 2001
Douglas-fir, MT	30-70 [within days]	3.7 [90]	94 [90]	0.094	8.3	Robichaud 1996
Douglas-fir, ID	40-75 [within days]	3.7 [90]	94 [90]	0.41	35	

the initial infiltration rates in the areas burned at low severity (which comprised the largest proportion of the prescribed burn area) fell within the upper end of the range from the areas left unburned and undisturbed areas (Robichaud 2000). The total sediment yields from the three 30-minute rainfall simulations on the plots burned at low severity were an order of magnitude smaller than the values from the plots burned at high severity (table 4) (Robichaud 1996). Similar differences in sediment yields were reported by Benavides-Solorio and MacDonald (2005).

Runoff and sediment yields were also measured from natural rainfall events at the catchment-scale (17 to 22 ac, 7 to 9 ha) after the same prescribed burns at both Slate Point and Hermada. At both sites, the runoff and sediment yields were generally low from the catchments subjected to both timber harvest and prescribed burning (table 5) (Covert and others 2005). The low runoff and sediment yields were most likely due to the generally low burn severity and the averaging of fire effects (Covert 2003; Robichaud 1996).

Earlier work also noted that erosion after prescribed fires occurred primarily in areas where the fires were locally severe or there was extensive disturbance due to forest harvesting (McNabb and Swanson 1990). A study of 200 permanent 11 ft^2 (1-m^2) plots in northern Idaho have a pre-harvest erosion of 0.04 t ac^{-1} (0.09 Mg ha^{-1}). Prior to any activities, the litter coverage was 83 percent, and this decreased by just 8 percent 1 year after helicopter-logging and broadcast burning. One year after broadcast burning, the total erosion was 0.8 t ac^{-1} (1.9 Mg ha^{-1}); 40 percent of this was attributed to the mechanical disturbance from logging and 32 percent to the broadcast burning (Clayton 1981).

Another study in northern Idaho measured erosion rates of 67.2 yd^3 ac^{-1} yr^{-1} (127 m^3 ha^{-1} yr^{-1}) from a clearcut area that was then burned by a high severity wildfire (Megahan and Molitor 1975). No erosion was measured from an uncut watershed that burned in the same wildfire. Erosion pin data suggested a net soil loss of 0.43 in (11 mm) on the clearcut and burned watershed as compared to a net gain of 0.20 in (5 mm) on the uncut watershed. Rill erosion was observed within 30 days after the fire on the clearcut watershed, while on the uncut watershed, there was some soil movement from rainsplash but there was no evidence of rilling (Megahan and Molitor 1975).

Wildfire

The effects of high severity wildfires on runoff and erosion are generally much more severe than the effects of prescribed fires. High severity fires are of particular concern

Table 5. Mean annual rainfall, runoff, and sediment yield for two catchments that were logged and broadcast burned at low burn severity. Hermada was burned 3 years after harvesting; Slate Point was burned 1 year after harvesting; and Round-Up was burned 2 years after harvesting (Covert 2003; Covert and others 2005; Robichaud 1996).

Catchment name	--------------Hermada-------------				---------------Slate Point---------------				---------Round-Up--------		
size (ac, ha)	22, 9				17, 7				5, 2		
Year since Rx burn	1 yr	2 yr	3 yr	4 yr	1 yr	2 yr	3 yr	4 yr	1 yr	2 yr	3 yr
ground cover (%)	95	98	99	99	98	99	100	100	92	96	98
Annual rainfall											
(in)	34.3	26.5	47.1	18.7	22.4	20.4	28.1	9.53	11.8	21.1	10.4
(mm)	870	673	1196	474	568	519	714	242	300	537	265
Annual runoff											
(in)	3.1	3.5	13	7.1	1.7	2.1	3.8	1.3	2.0	0.51	1.3
(mm)	78	89	320	180	43	53	97	33	51	13	32
Annual sediment yield											
(t ac^{-1})	0.00	0.00	0.00	0.00	0.0015	0.016	0.0052	0.00	0.045	0.045	0.045
(t ha^{-1})	0.00	0.00	0.00	0.00	0.0033	0.036	0.012	0.00	0.10	0.10	0.10

because the loss of protective cover and fire-induced soil water repellency can induce severe flooding and erosion even after moderate rain events (DeBano and others 1998; Neary and others 2005). In severely burned areas, high intensity, short duration rain events have increased peakflows from 2 to 2,000 times (DeBano and others 1998; Neary and others 1999, 2005). Published sediment yields after high severity wildfires range from 0.004 to 49 t ac^{-1} yr^{-1} (0.01 to over 110 Mg ha^{-1} yr^{-1}) in the first year after burning (Benavides-Solorio and MacDonald 2005; Moody and Martin 2001; Robichaud and others 2000). In most cases, the decline in soil water repellency and vegetative regrowth means that these large increases in runoff and erosion diminish quite rapidly. Most long-term studies show no detectable increase in erosion by about the fourth year after burning (Benavides-Solorio and MacDonald 2005; Robichaud and Brown 2000).

Watershed Effects of Fuel Management

It is much more difficult to quantify the effects of roads, timber harvest, and fuel treatments on stream flow and sedimentation at the watershed scale than at the plot scale because of the inherent complexity of the underlying processes and the variability over time. For example, Jones and Grant (1996) tried to determine the changes in stream flow caused by roads and timber harvesting on six watersheds in western Oregon that varied in size from 150 ac to 230 mi^2 (60 ha to 600 km^2). They concluded that:

1. forest harvesting increased peak discharges by as much as 50 percent in small basins and 100 percent in the three large basins;
2. increases in drainage efficiency were due to the connectivity of the road system to the stream channel network; and
3. the entire population of peak discharges was shifted upward by clear-cutting and roads.

Using the same data set, Thomas and Megahan (1998) were unable to detect any effect of cutting on peak flows in one of the large basins and determined that the data were inconclusive for two large basins. They found that the timber management activities had altered stream flows for smaller events on the small watersheds, but there were no detectable differences for the larger events (2-year return interval or greater). The effects of roads and forest harvest decreased over time, but they were still detectable after 20 years on the clear-cut watershed and for 10 years on the patch-cut and roaded watershed (Thomas and Megahan 1998).

Beschta and others (2000) also analyzed the same data set as Jones and Grant (1996) and they concluded that the increases in peakflows after harvest operations (including road building, clearcutting, cable logging, and site preparation) depended on the peakflow magnitude. Peakflow increases averaged approximately 13 to 16 percent after treatment for events with a recurrence interval of 1 year, and by 6 to 9 percent for storms with a recurrence interval of 5 years.

These different interpretations of the same data set reflect the challenges of assessing and understanding the effects of forest management activities at the watershed scale. The differences in site conditions, climatic regime, and treatment intensity mean that different studies have found very different results, and a study can be found to support almost any point of view. For example, 11 watershed studies from sites in British Columbia to California can be cited to show that logging can increase, decrease, or have no effect on the size of peak flows (Harr 1979). Accurate predictions of the effects of fuel treatments are only possible if there is a simultaneous understanding of the underlying processes and how the different effects might be transmitted and aggregated at the watershed scale.

Conclusions

Fuel management in forested areas can involve a number of activities, including the construction, maintenance, and use of forest access roads; timber cutting and removal; non-commercial thinning or mastication; and prescribed fire. The following conclusions reflect our current understanding of the effects of fuel treatments at the plot and watershed scales:

1. Roads greatly increase runoff and erosion rates at the plot and road segment scale. The effect of these increases at the watershed scale depends on the connectivity of the road and stream networks, but several studies have indicated that roads have minimal effect on runoff at larger spatial scales. More studies have shown that unpaved forest roads are chronic sediment sources and that roads can significantly increase sediment yields on small to moderate-sized catchments. Road building, maintenance, and obliteration can generate significant short-term increases in runoff and sediment. The effects of forest roads on runoff and sediment yields can be greatly reduced by improved road placement, road designs that dissipate runoff and direct it away from streams, and the widespread use of erosion mitigation techniques.
2. Non-commercial thinning operations (without yarding) have small, short-lived impacts on runoff and sediment production, even when operations extend over large areas.
3. Commercial thinning and yarding has a greater potential to increase runoff, erosion, and sediment yields because of the more extensive removal of the forest canopy; greater ground disturbance due to skid trails, cable rows, and landings; greater ground disturbance due to more intensive harvest; need for extensive road access; and increase in heavy truck traffic. The potential increases in erosion and sediment yield can be minimized by reducing the area and amount of soil disturbance, establishing buffer strips along stream channels, and minimizing overland flow by restoring severely disturbed areas.
4. High severity wildfires increase runoff and erosion rates by two or more orders of magnitude, while low and moderate severity burns have much smaller effects on runoff and sediment yields. If areas are burned at low severity, the potential for increasing peak flows and erosion rates is relatively small. However, if prescribed fires are conducted under dry duff moisture conditions and larger areas are burned at high severity, there is a much greater risk for significantly increasing runoff and erosion rates. The natural regrowth on severely burned areas means that overland flow rates and sediment yields generally return to pre-burn levels in approximately 4 years. Water yields may remain elevated for a longer period due to the time required for interception and transpiration rates to return to pre-burn levels.

5. Vegetative recovery after fuel treatments is generally very rapid, with erosion rates typically dropping to pre-fire levels within 1 to 2 years. Hydrologic recovery after fuel treatments also tends to be more rapid than after clearcutting or high burn severity fires because a smaller proportion of the forest canopy is being removed.
6. Fuel management treatments generally are needed every 10 to 20 years and the associated cumulative effects occur during each access and treatment cycle. Although hillslope erosion rates recover quickly, the road system, which is typically used and maintained between treatment activities, is a chronic source of sediment. Sediment yields from high severity wildfires are much greater than the increase in sediment yields due to fuel management activities, but the recurrence interval of such wildfires can be hundreds of years. Over longer time scales, the cumulative impacts of fuel treatments, repeated at 10 to 20 year intervals, when combined with the impacts of continuous road maintenance and use, may be similar to the pulse impact from wildfires.

The cumulative effect of fuel management activities is related to their location and concentration within a given watershed as well as the degree and frequency of disturbance for each activity. The watershed-scale impacts of any fuel management activity must consider the associated activities of road use, road maintenance, increased traffic, and multiple entries with various types of equipment as well as the combined effects of all the fuel treatments being applied. However, these effects are complex and interrelated. Few studies have examined the role of different controlling factors, much less the effects and interactions of the different activities on runoff and erosion at the watershed scale. Identifying the cumulative effects of timber harvest activities is a continuing challenge, as it is almost impossible to quantify the relative contribution of each activity at each location. It follows that determining the cumulative effects of fuel treatments, which generally cause less disturbance than timber harvesting, will be even more of a challenge.

Knowledge Gaps

Additional research is needed to understand the cumulative effects of fuel treatments at the watershed scale. Research to date has identified and quantified some of the key factors relevant to fuel treatment operations, but the combined effects of these variables are not well understood. In particular, studies are needed in the following areas to determine the

1. change in peak flows, soil moisture, and sediment yields from repeated entries into the forest for fuel management operations on various soil types, precipitation regimes, and scales—both time and area;
2. extent to which roads, burned areas, and timber harvest units are connected to stream networks and how this connectivity changes over time as a function of the precipitation regime and site conditions; and
3. changes in runoff and sediment yields due to reopening brushed-in, low-use forest roads for fuel management activities.

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Chapter 6.

Cumulative Effects of Fuel Treatments on Channel Erosion and Mass Wasting

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Introduction

Controversy over fuel treatments on public forestlands often focuses on the potential for such treatments to contribute to cumulative watershed impacts. If a fuel treatment project modifies the production or transport of water, sediment, or woody debris through a channel network, downstream habitats and aquatic resources may respond adversely to the changes. If these changes augment impacts from previous or on-going activities, the fuel treatment project will have increased the overall level of impact—the cumulative impact—to downstream resources.

As currently applied, "fuel treatments" include a variety of practices, such as prescribed burning, removal of sub-canopy "ladder fuel" and downed wood, thinning of canopy trees, thinning of understory trees, conversion of fire-susceptible stands, clearing of shaded fuel breaks, post-fire salvage logging, and logging of insect-damaged or at-risk stands. Many of these activities are not economically self-supporting, so they are often bundled with standard timber sales to offset costs. Such projects tend to be subjected to particularly intense public scrutiny, and questions are often raised concerning the extent to which fuel treatments influence erosion.

Considerable research has been carried out on channel erosion and mass-wasting processes, but few studies explore the effects of fuel treatments on such processes. Wondzell (2001) reviewed the literature available as of 2001. However, the scarcity of literature that specifically addresses the issue is not a critical problem. Fuel treatments influence factors controlling process rates in ways similar to activities (such as logging) and events (such as wildfires) that have been more widely studied. If erosion process mechanisms are understood, a large body of literature becomes applicable to the problem. This chapter describes characteristics of channel erosion and mass-wasting processes, describes the environmental factors that most strongly influence erosion processes, discusses the mechanisms by which fuel treatments can influence those controlling factors, outlines strategies for determining whether such influences will occur, and describes how erosion evaluations might be incorporated into a cumulative impact analysis. The erosion processes discussed here include channel-bank erosion, gullying, soil creep, landsliding, and related processes. Sheetwash erosion is considered in chapters 5, 8, and 13.

Characteristics of Erosion Processes

The erosion processes considered in this chapter often occur downstream or downslope of the triggering land use activities. Consequently, they not only can contribute to off-site cumulative watershed impacts, but also can themselves be influenced by multiple upslope or upstream activities. The potential influences of each process on downstream environments can be inferred through an understanding of

1. the factors affecting the distribution and rates of these erosion processes,
2. the kinds of sediment likely to be produced by each, and
3. the likely distribution of sediment inputs in time and space.

Channel-Bank Erosion

Bank erosion generally occurs by direct tractive erosion of a raw bank face or by undercutting and toppling or slumping. At sites where activities impinge on banks, direct disruption can also be important. The rate, mode, and distribution of channel-bank erosion are strongly influenced by bank materials, vegetation on the bank, pore pressures in the bank, channel flows, near-bank activities, and in-channel deflections. Hooke (1979), Thorne and Tovey (1981), and Couper and Maddock (2001) describe processes of bank erosion and evaluate factors that control the erosion rate.

Tractive erosion is most active on sparsely vegetated banks formed of non-cohesive materials such as sand and gravel. Repeated wetting and drying or freezing and thawing of bank materials can reduce their cohesion, contributing to "dry ravel" of the banks and increasing their susceptibility to tractive erosion. Banks formed from cohesive clays are more resistant to tractive erosion but can spall off in sheets when subjected to wet/dry or freeze/thaw cycles. Tractive erosion usually is negligible on bedrock banks unless the rock is poorly indurated. Bank erosion rates at a site generally increase if the duration of inundation increases, and more of the bank face is susceptible to tractive erosion for longer periods if hydrologic changes lead to increased channel flows.

Tractive erosion sources generally are distributed along the channel network—rarely is input dominated by erosion at a single site. Potential rates of tractive erosion are expected to increase downstream with increasing discharge and increasing prevalence of fine-grained bank materials. However, this general downstream trend may be reversed in low-gradient, lowland rivers with banks formed of cohesive materials (Lawler and others 1999). At a reach scale, erosion tends to be most rapid at the downstream outer edge of bends, where high-velocity flow impinges on banks. Often, erodible banks are inundated for only a small fraction of the year, so sediment inputs may be restricted to high-flow events.

Tractive erosion generally produces fine-grained sediment that remains mobile after entering the stream (table 1). Larger clasts are also contributed to the extent that they are present in the bank, but they are usually dislodged by undermining rather than by traction.

Bank failures are often triggered by undercutting, so rates of bank erosion by toppling and slumping depend in part on rates of tractive erosion. Toppling is most pronounced where floodplain- or terrace-surface deposits are cohesive. Roots increase cohesion even in inherently non-cohesive materials, and banks along incised grassland channels are often characterized by toppling failures.

In contrast to tractive erosion, bank failures tend to occur as high flows recede because

1. the susceptible material is then at its highest bulk density due to saturation,
2. pore pressures in the bank are high, and
3. the undercut soil mass is no longer partially supported by the hydrostatic force from inundation (Thorne and Tovey 1981).

Table 1. Selected characteristics of major channel erosion and mass-wasting processes.

Erosion process	Grain size	Sediment input timing	Location	Potential influences [a]
Bank erosion	fine to medium	high flows after high flows	of most concern in moderate to large channels	altered woody debris altered riparian vegetation altered channel form increased channel migration
Gully erosion	fine to medium	periods of runoff early season flows	hillslopes small to medium channels below diversions	altered site productivity lowered water table accelerated runoff more hillslope sediment delivery increased bank erosion altered channel form reduced floodplain connectivity
Soil creep	fine to medium	chronic	pervasive	increased bank erosion
Shallow slides	fine to coarse	high-intensity rain onto wet ground	inner gorges hillslope swales undercut banks certain bedrocks	altered site productivity flow deflection altered woody debris
Debris flows	fine to coarse	high-intensity rain onto wet ground	steep swales certain bedrocks	altered channel roughness flow deflection altered woody debris channel blockage
Deep-seated slides	fine to very coarse	very wet seasons	certain bedrocks	flow deflection
Earthflows	fine to very coarse	very wet seasons	certain bedrocks	flow deflection altered site productivity

[a] All erosion processes can also contribute to aggradation, turbidity, altered bed material, and altered bed stability.

Because bank failures often produce blocks of cohesive sediment and often occur after the flow peak, the sediment introduced may remain close to its source. Tractive erosion during lower flows then gradually mines away the sediment. In forested areas, bank failures often contribute woody debris to channels.

Undercutting can also trigger large streamside landslides that affect more than just bank materials. Such slides can be found wherever channels encroach on valley walls, and they frequently occur along the “inner gorges” (Kelsey 1988) characteristic of many tectonically active areas. Although undercutting usually contributes to the instability, these failures are also susceptible to the same kinds of influences as other landslides. Some streamside landslides initiate at the toe of a slope and propagate upslope over time, while others fail as a single unit.

Streamside landslides can produce large quantities of extremely coarse sediment. Their deposits may remain in place for long periods and can modify the course and character of the channel both upstream and downstream. Where slide deposits deflect channel flows, additional landslides can be triggered by undercutting of new sites, thereby propagating the impacts even farther downstream. The largest slides may create temporary dams, sometimes resulting in dam-release floods capable of scouring and widening channels for long distances downstream.

Direct disruption of channel banks and beds can occur through trampling by animals or people and by land use activities that impinge on stream channels, such as in-stream mining and yarding of logs across channels. The distribution and rates of erosion from direct disruption depend strongly on animal and human use patterns and on the original morphology of the banks. Sites where banks are low, for example, tend to be selected as crossing points. Initial sediment inputs from these sources usually occur during low-flow seasons, when disruptive activity levels are often highest. But because the resulting channel modifications tend to be unstable during higher flows, storm flows usually mobilize additional sediment as they rework the unstable deposits.

Inputs from direct bank disruption are often relatively fine-grained and can be an important source of turbidity during low-flow periods, when natural turbidity levels are low. Such unseasonal sediment inputs can be disproportionately important if they contribute to additional stress on organisms already challenged by low flows or high temperatures.

If a channel reach is not aggrading, incising, or changing form, bank erosion rates along the reach are expected to be roughly equivalent to rates of sediment resupply to those banks (Dietrich and Dunne 1978). Channels can supply fine sediment to banks through overbank deposition, while coarser channel-bed sediments can be incorporated into banks through bar accretion and associated channel migration. Where channels abut hillsides, hillslope sediment transport processes, such as soil creep and landsliding, also can contribute sediment to channel margins. Over a long period, slight imbalances in rates of erosion and deposition lead to the gradual down-wasting of the landscape, while over a short period, larger imbalances lead to temporary, localized changes in channel and bank form. Over an intermediate period, however, conditions tend to average out in most undisturbed settings. An undisturbed headwater channel is expected to look much the same today as it did a hundred years ago, despite relatively continuous bank erosion along the channel.

Under some conditions, bank erosion can occur quite rapidly and may lead to extreme changes in channel character. Major floods or debris flows can significantly widen channels, and sudden or episodically high inputs of sediment can fill channels, leaving flow to spread across the valley bottom in multiple “braided” flow strands. If the aggraded sediment is erodible, each strand then mines the new deposits, gradually shifting the load downstream. Braided channels are typical of glacial outwash plains and alluvial fans.

Unless sediment input remains high or a channel is freely migrating, accelerated bank erosion is usually self-limiting. Banks begin to stabilize once a channel has widened enough that flow becomes too shallow—or impinges on banks too infrequently—to remove eroded sediment.

Gully Erosion

Gully erosion is a particular kind of rapid channel erosion that forms incised, steep-walled channels. Rapidly incised channels small enough to be eliminated by plowing are referred to as “rills,” which are considered a component of sheetwash erosion. In forested settings in North America, gullies are usually of most concern along headwater channels and in meadows. In the semi-arid west, gullying is also important farther downstream, where incision of gullies known as “arroyos” has strongly altered valley-bottom conditions over the past 150 years (Cooke and Reeves 1976). Factors expected to influence susceptibility to gullying include channel gradient, substrate, vegetation cover, and peak-flow regimes. Bull and Kirkby (1997) and Oostwoud Wijdenes and Bryan (2001) discuss gully erosion processes and factors that influence rates of gully erosion.

Gullies can form in unchanneled settings by upslope migration of channel heads, collapse of subsurface soil pipes, or incision of scour holes along previously stable drainageways. Generalized incision or headcut migration can also form gullies along existing channels. Widespread gullying is most often associated with anthropogenic changes, such as introduction of cultivation or livestock. However, gullying can also be triggered by natural events that reduce vegetation cover (such as wildfires), deposit erodible material (such as volcanic eruptions), or generate extreme surface runoff (such as intense thunderstorms). Stratigraphy of valley-bottom deposits in the southwestern United States suggests that climatic shifts have triggered several episodes of arroyo formation over the past 4,000 years (Waters and Haynes 2001).

Gullies commonly grow by upstream retreat of a near-vertical headwall. Water falling over the headwall excavates a plungepool at its base, undercutting the headcut and sidewalls. Undercutting promotes toppling failures, allowing the gully to widen and to progress upstream. Seepage at the base of a headcut can also increase rates of undercutting and in some areas can be the dominant mechanism for headcut retreat (Higgins and others 1990). A depositional lip typically forms at the downstream end of the plungepool, and the channel downstream is often graded to the level of the lip. Channel-fill sediment downstream of a migrating headcut and its associated plungepool may thus represent the upstream progress of deposition on the plungepool lip.

Gully cross sections are modified by the types of bank erosion processes described in the previous section. Often, gully width increases and wall gradient decreases as a function of the distance downstream of a headcut, reflecting progressively longer periods of recovery since passage of the headcut. Upstream headcut migration may halt if a headcut encounters non-erodible material or if the contributing area becomes too small to generate erosive flows.

Gullying most commonly forms intermittent or ephemeral channels, where "intermittent" refers to channels that carry water seasonally, while "ephemeral" usually implies that water is present only during storms. Sediment often accumulates in gullies during dry periods through trampling, dry ravel, and spalling of banks. The first flows after a dry period may then carry particularly high sediment loads as the accumulated sediment is flushed out (Crouch 1990). Gullying tends to be suppressed where soils contain coarse sediment because the coarse clasts can armor the bed and banks and restrict further erosion. Consequently, gullying is generally associated with fine-grained substrates, and sediment produced by gullying usually is readily transportable by channel flows.

Some areas of steep terrain and poorly consolidated bedrock are susceptible to rapid formation of gullies large enough and steep enough that gully-wall failures begin to generate debris flows. At this point, expansion no longer depends on channel flow, and the gully network can form an amphitheater-shaped basin that extends to the ridgeline. These gully-landslide complexes are referred to as "gully slips" in New Zealand, where they formed after conversion of forest to pasture in some regions (Betts and others 2003). Reforestation of gullied watersheds has halted the growth of many New Zealand gully slips, although the forms remain present.

Recovery from a gullying episode tends to be slow, and evidence of gullying may persist for centuries in the form of terraces. Gullies may continue to produce sediment from headcut retreat and wall erosion long after the conditions that initially triggered gully formation have been reversed. Sediment redeposited downstream of actively incising reaches may remain in storage for long periods. Moody and Martin (2001), for example, expect that much of the sediment produced by channel incision after a fire in Colorado will remain in the watershed for several hundred years.

Soil Creep

Soil creep is a gradual mass wasting process that occurs within the soil mantle on most hillslopes. Transport can occur through plastic deformation of the soil mass in some clay-rich soils, but more common is transport by displacement of individual soil particles through root growth, animal burrowing, wetting and drying, and freezing and thawing. These soil disturbances tend to move particles preferentially downslope because of the influence of gravity, and incremental transport of individual particles combines to gradually displace the entire soil profile. Saunders and Young (1983) provide a tabulation of measured creep rates. More recent work (such as Heimsath and others 2002) adds to the measurement record and provides further discussion of creep mechanisms.

Although rates of soil creep are slow, the process is influential because it occurs over most of the landscape. The nature of the motion and its slow rate make measurement difficult, so transport rates are generally estimated by other means. In many quasi-steady-state systems, soil creep is the major source of sediment resupply to stream banks abutting hillslopes, so long-term measurements of bank erosion can provide estimates of creep rates.

Creep rates are expected to depend on hillslope gradient, soil texture, soil moisture, biological activity within the soil, and vegetation, but little is known about the relative influences of these factors because measurement is difficult. Changes in hillslope conditions that increase soil moisture or soil disturbance are likely to increase creep rates, but such effects have not yet been documented in a controlled setting.

Shallow Landsliding

"Shallow landsliding" is a commonly used term that is not recognized in the widely adopted landslide classification system presented by Cruden and Varnes (1996). Some apply the term to any slide that involves only colluvial material, thus corresponding to the "debris" material category of Cruden and Varnes (1996). Others use the term to indicate that the depth to the failure plane is markedly less than the length or width of the slide, corresponding roughly to the "translational" motion category of Cruden and Varnes. Under this second usage, a 5-m deep slide might be considered "shallow" if it is 50 m long or "deep-seated" if it is only 10 m long. When used in forestry related literature, "shallow landsliding" appears to most commonly refer to translational slides. Although these often consist primarily of surficial deposits, some may include significant bedrock. Saunders and Young (1983) compiled measurements of landsliding rates, and many more recent studies provide additional measurements.

Shallow failures generally occur during periods of high-intensity rain that falls onto already wet soils. Relations between slide occurrence and various measures of rainfall intensity have been developed at many sites (for example, Crosta 1998; Crozier 1999; Finlay and others 1997; Nilsen and others 1976; Reid and Page 2003). Slide frequencies are also influenced by hillslope gradient, root cohesion, soil moisture, lateral slope convergence, bedrock type, soil depth, and soil texture. Shallow slides occur most frequently on steep portions of the landscape into which subsurface flow is concentrated, such as headwater swales and inner-gorge slopes. Failure planes of small slides are often within the rooting depth of forest vegetation. Increased landsliding rates are sometimes noted several years after logging, after dead roots have decayed but before new roots have matured (Bishop and Stevens 1964; Swanston 1969). A variety of analytical tools have been developed to identify sites susceptible to shallow sliding (for example, Montgomery and Dietrich 1994).

Shallow slides often mobilize both soil and partially weathered bedrock, so most slide deposits contain a wide range of grain sizes. In forests, deposits usually include woody debris. The proportion of landslide debris reaching a channel tends to decrease with increasing distance between the landslide scar and the stream. However, even distant slides may contribute sediment where slopes are steep or when slides are generated by intense storms. At such times, overland flow generated from the new slide scar can act as a temporary extension to the downstream channel, allowing sediment delivery directly to the stream. Shallow landslides generally occur during periods of intense rain and high flow, so some portion of the landslide sediment reaching a channel is usually transported downstream during the triggering storm.

Once a shallow slide has occurred, the bared scar continues to contribute sediment through surface erosion and gullying until vegetation regrows (Larsen and others 1999). Sediment inputs can be further prolonged as streams rework temporarily stored landslide deposits, contributing both suspended sediment and bedload (Sutherland and others 2002). Because of the relatively slow transport rates for bed material, decades may be required before debris from a major landslide-generating storm is fully evacuated from a moderate-sized channel (Madej and Ozaki 1996). Meanwhile, aggradation from the downstream transport of landslide debris can deflect stream flow into banks, causing secondary failures at downstream sites.

Debris Flows

Shallow failures often displace saturated material, and pore pressures at some failure sites are high enough that landslide debris can lose all cohesion and flow as a liquid. If debris moves as a fluid, the event is referred to as a "debris flow." Field evidence suggests that some debris flows can initiate within channel deposits, and flows have occasionally been found to originate high on slopes without evidence of an initiating landslide (J. McKean, personal communication).

Debris flows in steep forestland channels can entrain large volumes of channel deposits and woody debris (May 2002). Wood-bearing debris flows often come to rest where the wood becomes jammed or the channel gradient decreases to the point that material can no longer flow. These conditions are often present where an affected tributary joins the mainstem channel (Benda and Cundy 1990), so landscapes characterized by debris flows commonly exhibit debris fans at tributary mouths. Debris flow deposits are recognized as unstratified mixtures of diverse grain sizes.

Debris flows and less dense, sediment-laden "hyperconcentrated flows" can also form through rapid incision and entrainment of in-channel deposits (Cannon and others 2001; Cannon and Reneau 2000). In some areas, such flows are common after high-intensity fires that generate hydrophobic soil layers. After a debris flow has occurred, the scoured channel and debris deposits remain subject to accelerated erosion until they are revegetated or become armored by coarse sediment. Log jams formed by debris flows at tributary junctions can accumulate considerable volumes of bed material from both the affected tributary and the mainstem. When the jam eventually fails, a portion of the trapped sediment is released to resume its downstream transport.

Deep-Seated Landsliding and Earthflows

The term "deep-seated landslide" is also not recognized by the Cruden and Varnes (1996) classification system. The term is variously used to refer to landslides having failure surfaces within bedrock or to slides that are deep relative to their length and so have moved by rotation along a curved failure plane (slumps). In the forestry literature, the term is used primarily for rotational slides, though it also often encompasses non-rotational bedrock failures if they seem unlikely to have been influenced by near-surface pore pressures.

Large, deep-seated landslides tend to be more responsive than shallow slides to seasonally high rainfall accumulations and respond less to high-intensity rain bursts or individual storms. The failure surface is often deep enough that root cohesion is inconsequential. The largest features remain visible on the landscape for millennia, and only ancient examples are present in many areas. Controversy persists over the extent to which long-stabilized slides can be reactivated by land management activities.

Once mobilized, large deep-seated slides can remain active for decades or longer. Slide surfaces are often irregular and hummocky, and a depression or sag-pond may be present at the base of the headwall scarp. In some cases, the progressive motion is slow enough that it is most readily recognized by haphazard orientations of mature tree trunks, disruption of road surfaces, or distortion of fences. In other cases, the entire failure may occur over minutes or days, with subsequent activity limited to erosion of the disrupted slide mass and bared scarp. The toes of old slumps often form over-steepened slopes that are now susceptible to shallow landsliding. Sediment contributed by deep-seated slides can include grain sizes ranging from weathered clays to large blocks of intact bedrock.

In some terrains, materials initially mobilized by deep-seated slumps continue to move downslope as earthflows. Earthflows are plastically deforming masses of unconsolidated material that remain active over long periods, ordinarily moving from several centimeters to tens of meters each year. More rapid flows that occur as discrete events are termed "mudflows" or "debris flows." Earthflows generally occur in areas of clay-rich, mechanically weak bedrock, such as shale or argillite. In areas susceptible to earthflows, evidence of past activity is often visible as hummocky terrain, and active earthflows may appear as patches of grassland in otherwise forested areas.

Earthflows are most active during seasonally wet periods. Activity may cease during the dry season or in years with low rainfall and resume when water tables have again risen to a threshold level. On large flows, variations in velocity generally are not associated with individual storms. Flageollet and others (2000) describe the three-dimensional structure of a major earthflow, and Iverson and Major (1987) describe patterns of motion for a similar flow over a 3-year period.

Many large earthflows are bounded by streams at their toes. Activity of the slide can then strongly control the form and sediment load of the stream channel, while channel erosion, in turn, repeatedly undercuts and reactivates the slide mass. A year of rapid motion may constrict the stream, while years of lesser activity may allow the stream to reexcavate its characteristic channel.

Earthflow toes are often the site of intensive surface erosion and shallow landsliding. Consequently, periods of high sediment production are associated with individual storms even though motion of the earthflow itself often is not. Earthflow surfaces often support gullied drainage networks that contribute additional sediment during storms. Because earthflows generally form in areas with weak, clay-rich bedrock, even the coarser blocks tend to be rapidly broken down once introduced to a channel. The largest sediment blocks remain stranded in channels until worn away, while finer sediments can contribute to chronically high suspended sediment loads.

Related Erosion Processes

A variety of other channel erosion and mass-wasting processes can be strongly influenced by fuel management activities and may be important in some settings even though they rarely dominate the sediment supply. Of particular note for forested areas are subsurface channel erosion (tunnel erosion or piping; Jones 1981; Uchida and others 2001), tree throw (Schaetzl and others 1989), and animal burrowing (Gabet and others 2003). Several other processes, such as dry ravel and sheetwash erosion, frequently occur on sites bared by channel erosion and mass-wasting processes and prolong sediment inputs from the primary sources.

Tunnel erosion is common in unchanneled swales in many areas, though its presence is often unnoticed. The process can usually be detected by examining channel heads for soil pipe outlets. Pipeflow is generated primarily by subsurface drainage during storms, though some pipe networks can continue to flow long into the dry season. Pipeflow tends to remain relatively clear of sediment even during storms. Surficial erosion processes do not contribute directly to pipeflow sediment loads unless the pipe's roof is breached upslope. Instead, most tunnel erosion sediment is generated by bank erosion processes and tractive erosion within the pipe. Tractive erosion of the pipe circumference increases with increasing discharge.

Piping is often present in unchanneled swales that are subject to periodic debris slides. In a quasi-steady-state system, soil-creep input of sediment to these swales is largely balanced by the combined activity of tunnel erosion and landsliding. At a smaller scale, pipe diameters are expected to remain relatively constant over time, so erosion of sediment from pipe walls evidently keeps pace with the tendency for pipes to constrict due to soil creep. Tree throw or cave-ins can unroof pipes, temporarily increasing sediment loads and diverting flow to the surface. It is likely that pipe roofs can be reestablished by bridging with forest floor litter and small woody debris, though descriptions of the recovery process have not been published.

Tree throw can contribute appreciable sediment to streams where forested banks or valley walls are steep. At such sites, it may be difficult to distinguish tree-throw events from landsliding because both contribute a mixture of woody debris and sediment and leave similar scars. The distinction rests on the cause of the failure: did a landslide topple the trees, or did falling trees destabilize adjacent soils?

Tree throw is most prevalent during high winds that occur while soils are saturated and trees are in full foliage. Although blowdown of snags is common after fires, such falls often occur by stem breakage and thus do not contribute directly to sediment loads. Whether a tree is likely to break or uproot also varies by species (Veblen and others 2001).

Under most conditions, animal burrowing is an implicit component of soil creep, so creep rate estimates generally account for displacement by burrowing. However, burrowing can also influence sediment production by exposing unvegetated soils to overland flow or by directly contributing sediment to streams where burrow tailings are deposited within the high-flow stream margin. In the Pacific Northwest, for example, mountain beaver burrows often are associated with headwater streams. Populations of burrowing animals vary with stand age, and young stands may provide food sources to support large populations of burrowing rodents.

"Dry ravel" of surface sediments occurs when grains are transported downslope in the absence of flow. Cohensionless particles can be dislodged downslope by even minor disturbances, so dry ravel is common on landslide scars and eroding banks of non-cohesive sediments. Particles on a bare surface can be loosened by cycles of wetting and drying or freezing and thawing, and fire often promotes dry ravel by baring mineral soils and by burning out small woody debris that has trapped sediment on stream banks and hillslopes. After fires, ravel can be an important source of sediment delivery to low-order streams, where the accumulated sediment can then be easily remobilized by wet-season flows (Roering and Gerber 2005).

Factors Influencing Erosion Processes

Although many environmental characteristics can affect erosion processes, several are particularly influential for multiple processes. The distribution of erosion processes, their rates of sediment production, and the timing of sediment inputs are largely controlled by topographic setting, materials, surface conditions, hydrologic conditions, and vegetation. Each of these factors also exerts influence on the others. An understanding of these controlling factors and of how they may be influenced by management activities provides the link needed to evaluate the effects of specific land use activities on erosion processes.

Topography

The susceptibility of a site to various erosion processes can often be inferred from its topographic setting. On hillslopes, local gradient, lateral convergence, and distance from the ridge-top strongly influence which erosion processes will be active. Once processes are activated, their influence depends in part on how far the mobilized sediment travels. Topographic conditions downslope of an eroding site strongly affect the proportion of the mobilized sediment that reaches a channel. Particularly influential are gradient and lateral convergence downslope, presence of topographic irregularities along the travel path, and distance to the stream.

In channels, local topography strongly influences the shear stress imparted by flows on the bed and banks. Deep, high velocity flows on steep slopes develop the highest shear stresses—these are the sites most likely to incise. Along a channel, variations in gradient and channel width can control the distribution of aggradation and incision.

Topographic setting is also important because management-related topographic changes can trigger a variety of erosion processes. On forested hillslopes, most topographic modifications are associated with road construction. Excavation of over-steepened road-cuts, emplacement of fills, construction of stream crossings, excavation of road-side ditches, and deposition of unstable side-cast material can all contribute to increased erosion risk.

Topographic modifications in channels also often result from road construction. Banks may be realigned and armored to protect riparian roads, and levees are sometimes constructed to reduce flooding. Bridges or culverts modify channel cross sections, sometimes restricting passage of woody debris or coarse sediment. Stream crossings on steep slopes are particularly vulnerable to failure because drainage structures can be blocked by woody debris, allowing flow to pond behind the unconsolidated road fill. Overtopping can then lead to rapid gullying, while increased pore pressures within the fill can trigger landslides. At some sites, in-stream structures have been built to divert flow for low-head hydroelectric power generation or to pond water for livestock. Such changes can alter sediment delivery and channel erosion rates downstream. At a larger scale, dams are major controls on downstream channel forms and processes and can strongly influence the effects of upstream activities on downstream environments.

Materials and Surface Conditions

Material characteristics—particularly grain size and cohesion—also influence which processes are likely to be active and how far eroded sediment can move. Coarse-grained sediment tends to have low cohesion and thus is susceptible to dry ravel, but bedrock that weathers to coarse material is not likely to support earthflows. Clays, in contrast, are extremely fine-grained and cohesive. Earthflows usually occur in areas with clay-rich bedrock, but cohesive clay soils are resistant to tractive erosion and ravel. Coarse sediment requires high shear stress for in-stream transport and so may be mobile only during high flows. Coarse sands, gravel, and cobbles thus generally form the bed material in upland streams, while clays and silts are readily transported in suspension and contribute little to the bed material.

Surface characteristics can strongly influence the susceptibility of hillslope or channel materials to erosion. Removal of readily transported fine-grained sediment often leaves a lag of coarse sediment that armors hillslopes or stream beds, impeding further erosion. Soil surface characteristics change radically after fires. Erodible mineral soils are often exposed when protective organic material is burned off. At some sites, fire can generate a surficial hydrophobic layer (DeBano 2000a, b), resulting in rapid runoff and increased surface erosion during subsequent rains. The increased runoff may trigger debris flows, gullying, and increased bank erosion downstream. Ground-disturbing activities can also modify the susceptibility of swales to gully incision by exposing erodible soils.

Hydrologic Conditions

Water affects most types of erosion and sediment transport on hillslopes, and the largest sediment inputs usually occur during major storms. The occurrence of shallow landslides is particularly responsive to the timing and spatial distribution of high pore pressures, which in turn are influenced by soil surface topography, bedrock surface topography, subsurface drainage paths, location along a slope, hydraulic conductivity of soils and bedrock, and the amount of water contributed to the site by precipitation or surface and subsurface drainage. These factors generally control the routing of water down hillslopes and so influence the distribution of gullying, tunnel erosion, debris flows, deep-seated slides, and earthflows.

Shear stress at a channel cross section increases with discharge, thus increasing rates of sediment transport and channel erosion. Because of this dependence, and because sediment inputs from hillslopes also respond to water, most of a stream's annual sediment transport may occur during a few major storms. Changes in the timing, amount, and duration of runoff change the timing, amount, and duration of in-channel erosion, sediment transport, and aggradation.

Land use activities or natural events that modify hillslope hydrology can influence rates of shallow landsliding. Hillslopes may become wetter after logging or wildfire due to decreased transpiration and rainfall interception. The presence of roads often influences hillslope hydrology by rerouting shallow subsurface flows and by diverting road drainage onto hillsides.

Vegetation

Vegetation strongly affects erosion processes by influencing soil strength, surface materials, and hydrology. Root networks in both forest (Schmidt and others 2001) and grassland (Preston and Crozier 1999) can provide additional cohesion to soils of low inherent strength. In some areas, the distribution of shallow landslides reflects this influence. Roering and others (2003), for example, found that slides in their study area were located at some distance from the nearest trees.

Forest-floor litter strongly affects the characteristics of near-surface soil horizons. Organic-rich horizons often have relatively open textures, resulting in high infiltration rates and high moisture storage capacities. Overland flow is uncommon where deep litter has accumulated, and litter shields mineral soils from surface erosion and gully initiation. Removal of a litter layer through mechanical disruption or fire can increase the incidence of overland flow, thus increasing rates of downstream channel erosion.

Channel erosion rates can be particularly sensitive to vegetation changes because of the influence of vegetation on hydrology. Plants use large quantities of water, drawing it from the soil through their root systems and transpiring it into the atmosphere through their leaves. Conversion of vegetation to a community with lower water use increases runoff (Bosch and Hewlett 1982), thereby increasing the potential for channel erosion. Plants also trap rain and snow on foliage, increasing the evaporative loss of water during and after storms (Calder 1990). In areas where cold-season storms can include either rain or snow, warm rainstorms falling onto well-developed snowpacks can generate large flood flows that are often associated with significant in-channel erosion. In these settings, snow accumulations are highest and melt most rapidly in cleared areas, so presence of a forest cover moderates the runoff rate (Marks and others 1998).

Forest stand characteristics in the western United States are changing in response to earlier fire management strategies and on-going global climate change. Given the strong influence of vegetation on erosion processes, erosion regimes are expected to change in response. But even though vegetation change can strongly influence short-term sediment yields, influences on sediment yield over the long term may not be as great because the long-term average soil erosion rate must balance the long-term average soil formation rate if soil depths do not perpetually increase. The soil formation rate, in turn, is influenced by soil depth, erosion processes, and vegetation, and soil depths are controlled by erosion processes and soil formation rates. Because these factors are interdependent and all are influenced by vegetation, a vegetation change alters the balance between them. Where forest was converted to grassland in parts of the East Cape region of New Zealand, for example, deep forest soils are being removed by widespread shallow landsliding (Reid and Page 2003). Over time, a transition to shallower soils typical of grasslands is likely at these sites. The net erosion rate might eventually be nearly the same as before the vegetation change as erosion once again becomes constrained by soil formation rates, but the transition period is characterized by extreme erosion as the volume of sediment stored on hillslopes is rapidly reduced (see also Gabet and Dunne 2003). Such short-term readjustments on hillsides can lead to profound long-term changes downstream, where channels may be choked and floodplains inundated by the sudden influx of new sediment.

Potential Influences of Fuel Treatments on Erosion Processes

Once the factors controlling erosion process rates are understood, the potential influences of various fuel treatments can be inferred by examining their effects on the controlling factors. Forest fuel treatments can be grouped into four categories on the basis of their potential effects on hillslope and channel erosion processes: managed burning, mechanical treatments, logging, and strategic stand design. These activities are supported by the transportation infrastructure and, in some cases, require modification of existing road networks. General patterns of influence are summarized in table 2.

Managed Burning

Fire can be used to reduce ground fuel over wide areas at relatively low cost, whether it is applied through prescribed burning or by allowing wildfires to burn unhindered in particular areas. Activities associated with managed burning often include road use and construction of fire breaks. A small proportion of managed burns escape control or burn more severely than planned, leading to erosional conditions typical of more intense wildfires. Many publications describe the effects of wildfire on erosion processes (for example, Wondzell and King 2003).

Table 2. Potential influences on major channel erosion and mass-wasting processes.

			Treatment[a]				
Process	**Influential attributes**	**Mechanisms of change**	**Brn**	**MT**	**Log**	**Sal**	**Rd**
Bank erosion, gullying	peakflow, runoff	interception, transpiration	+		++		
	“	road, skid trail drainage		+	+	+	++
	“	hydrophobicity	++				
	“	compaction		+	+	+	++
	direct disruption	trampling, trafficking		+	+	+	++
	soil-pipe collapse	trampling, trafficking		+	+	+	++
	rooting density	trampling, trafficking		+	+	+	
	“	canopy removal			+		
	“	burning of groundcover	++				
	surface armoring	burning of litter	++				
	“	less ground-cover vegetation	++	+	+		
Soil creep	soil moisture	interception, transpiration	+		++		
	soil disturbance	trampling, trafficking		+	+	+	+
	rooting change	burning of groundcover	+				
	“	canopy removal			+		
Shallow slides, debris flows	antecedent wetness	interception, transpiration	+		++		
	increased drainage	road, skid trail drainage		+	+	+	++
	“	hydrophobicity	++				
	“	compaction		+	+	+	++
	“	interception, transpiration	+		++		
	undercut toe	disruption		+	+	+	++
	undercut by flow	road, skid trail drainage		+	+	+	++
	“	hydrophobicity	++				
	“	compaction		+	+	+	++
	material	emplaced fill					++
	“	less woody debris	+		++	++	
	root cohesion	canopy removal			+		
Slumps, earthflows	seasonal wetness	interception, transpiration	+		++		
	“	road, skid trail drainage		+	+	+	++
	root cohesion	canopy removal			+		
	undercut toe	disruption		+	+	+	++
	undercut by flow	road, skid trail drainage		+	+	+	++
	“	hydrophobicity	++				
	“	compaction		+	+	+	++

+ indicates a likely influence, and ++ indicates that the influence is likely to be strong.

[a] Treatments and associated activities: *Brn* = Managed burning; *MT* = Mechanical treatment; *Log* = Green-tree thinning; *Sal* = Salvage logging; *Rd* = Road construction or use.

Burning strongly affects soil surface characteristics, ground-cover vegetation, and organic debris on the forest floor, while construction of associated roads and firebreaks mechanically disrupts soils. Depending on the soil type, vegetation type, and burn intensity, burning may induce hydrophobicity in soils (DeBano 2000a, b; Huffman and others 2001; Robichaud 2000). Rain falling on hydrophobic soil may run off as overland flow instead of infiltrating, increasing the likelihood of gully erosion, channel incision, channel-bank erosion, and in-channel debris flows. These processes are also accelerated by burning of soil-surface litter and in-channel woody debris, and by removal of ground-cover vegetation. Canfield and others (2005) describe channel incision after a fire, and Istanbulluoglu and others (2003) describe post-fire gullying. In general, the potential for accelerated erosion is expected to increase with burn intensity (Wondzell and King 2003). Hillslopes may be particularly susceptible to other influences after burning. In one case, for example, trafficking after a low-intensity burn triggered incipient gullying (Saynor and others 2004).

Burning of ground-cover and understory vegetation may reduce transpiration and increase soil moisture. However, these vegetation components usually have shallower roots than overstory vegetation, so the effect is expected to be insignificant by the end of the dry season in areas characterized by seasonal drought. Similarly, changes in rainfall interception would be small unless a significant proportion of the vegetation burns.

If large portions of a landscape are treated, managed fire will burn through small channels and riparian zones. In-channel erosion rates are likely to increase where channels have burned. Sediment in low-order channels is often held in place by small pieces of woody debris. When these burn, trapped sediments are free to move downstream Loss of protective litter in unchanneled swales may allow gullying to progress at these sites, and burning of bank vegetation can increase susceptibility to bank erosion. It may be possible to burn during seasons when naturally high moisture levels might deflect burns from riparian areas or moderate their intensity, but unseasonal burns may defeat faunal strategies for coping with typical wildfires. Spring burns, for example, may disproportionately impact amphibians that are seasonally dispersed across the landscape (Pilliod and others 2003).

Mechanical Treatments

Accumulations of dead wood on the forest floor can be removed mechanically, along with sub-canopy trees that form "ladder fuel" capable of carrying a ground fire to the forest canopy. Dense understories of suppressed conifers form ladder fuel at many sites. Such wood usually is not merchantable and may be treated on site by chipping or chunking, with the pieces then spread as mulch. Alternatively, debris may be piled and burned or marketed for wood chips. Occasionally, logging of the smaller trees (thinning from below) produces stems large enough to be marketable as saw logs or fence posts. In each case, associated activities include road use and may involve road construction.

The primary erosional influences of mechanical understory treatments are likely to be associated with direct disruption to soils through yarding or by high-intensity fire effects under burn piles. Effects are particularly likely where these activities impinge on unchanneled swales or low-order channels. Spreading of chipped materials as mulch may promote infiltration on hillslopes, reducing the potential for increased erosion in swales.

Mechanical treatments are expected to influence hydrology primarily by compaction and disruption of soils due to yarding or other trafficking. Increased surface runoff from compaction may increase the potential for gullying, both in previously unchanneled swales and in downstream channels. Changes in live vegetation density are unlikely to be large enough to significantly influence transpiration or interception unless a high density of green ladder fuel is removed.

Logging

A number of timber sales recently have been categorized as fuel treatments, either because they accomplish canopy thinning (thus reducing the likelihood of spread of a crown fire) or because they remove dead or at-risk stems after wildfires or insect outbreaks. In the case of green-tree removal, such activities entail the same types of erosional consequences as ordinary timber sales, and such effects have been widely studied (for example, Chamberlin and others 1991). McIver and Starr (2000) summarize literature on the effects of post-fire logging on erosion.

After logging, decreased transpiration and interception may lead to increased pore pressures and reduce slope stability. Reduced interception is likely to be the more important of the two mechanisms in the coastal Pacific Northwest, where high interception rates have been measured even during the high-intensity winter rainstorms that generate most sediment in the region (Reid and Lewis 2007). Reduced interception increases effective rainfall, directly increasing the geomorphic impact of individual storms as well as contributing to increased seasonal groundwater levels. In contrast, transpiration changes are most influential during the growing season and so may be most important in areas where the major sediment-producing storms do not occur in winter. Decreased transpiration does not directly increase effective storm rainfall but can augment pore pressures by slowing the reduction of soil moisture and groundwater levels after storms so that antecedent conditions are wetter than usual at the onset of the next storm. Reduced interception and transpiration can thus increase both the activity of deep-seated slides and the risk of shallow landslides and debris flows.

The risk of shallow sliding is further enhanced by reduced root cohesion after green-tree logging. Debris flows may be more mobile if the entrained woody debris does not include boles, which are most likely to lodge in channels and halt the flow. Logging has also been associated with increased activity of existing deep-seated landslides (Swanston and others 1987), and earthflow velocities under forest vegetation in New Zealand were found to be several orders of magnitude lower than in deforested terrain (Zhang and others 1993). Such differences may reflect changes in both hillslope hydrology (Reid and Lewis 2007) and the mechanical behavior of the root-laden soil (Zhang and others 1993). Each of these effects will vary in importance with the proportion of trees logged. Miller and Sias (1998) modeled the effect of altered forest canopy on the stability of a deep-seated landslide.

In areas where rain-on-snow flooding occurs, clearcut logging can increase flood frequencies and magnitudes by allowing deeper snow accumulation and increasing the melt rates. Selective logging is expected to have a lesser effect. At sites where logging has been extensive enough to modify runoff characteristics, small channels with erodible beds and banks are likely to adjust to altered flow regimes through bank erosion, incision, and downstream aggradation. The extent of the adjustment depends in part on the magnitude and persistence of flow changes.

Assessment of the erosional consequences of salvage logging requires consideration of two issues that are not relevent for green-tree logging. First, erosion processes reflect interactions between the salvage logging and conditions left by the disturbance because a site's sensitivity to logging-related impacts may have increased due to the initial disturbance. Hillsides and small channels can become more susceptible to erosion if the litter layer and small organic debris have burned or if surface runoff has increased due to burn-induced hydrophobicity.

Second, removal of dead or dying trees modifies the erosional response to the initial disturbance. Fires and other stand-killing disturbances themselves increase erosion rates. Landslide rates may increase due to reduced transpiration, interception, and root cohesion, and surface erosion increases where mineral soil is exposed. But after disturbance, newly downed wood can trap eroded sediment by adding roughness to slopes and channels, and woody debris may reduce landslide debris mobility by promoting debris jams. Downed wood also can provide soil moisture reservoirs that hasten regrowth. To the extent that post-disturbance management reduces the supply of woody debris, these inherent recovery mechanisms may be impaired relative to an unmanaged condition.

Strategic Stand Design

A fourth fuel reduction approach employs landscape-level design of forest stand distributions to restrict the spread of individual wildfires or to modify fire intensity in critical areas (Graham and others 1999). This strategy arose because there is far more at-risk forest present than can be treated in the near future using the approaches described above. However, if such approaches are used strategically to control future fire behavior at specific locations, the areal extent or overall effects of wildfires might be controllable. For example, one of the methods previously described might be used to establish strips of fire-resistant forests, called "shaded fuel breaks," along ridgelines. Fires originating within a watershed would then be more likely to go to ground upon reaching the fuel break and thus be more readily managed.

Similar strategic planning can be used to protect at-risk structures and communities by concentrating fuel management activities around the area to be protected. Because these approaches are limited in scope, they are less likely to entail widespread erosional consequences than is a more extensive implementation of fuel treatments.

Use of strategic stand design introduces a need to understand the implications of landscape-scale distributions of erosion processes. Local effects can be evaluated on the basis of the particular practices used, while the broader-scale effects reflect the distribution of treatments (in time and space) and the nature of process interactions within and downstream of particular watersheds. At this scale, an understanding of in-stream sediment transport becomes particularly important.

Other Activities Associated with Fuel Treatment

Each fuel treatment approach described above is associated with additional activities such as road use and, in some cases, road construction. Roads are often a major source of management-related sediment through road surface erosion, gullying, landslides, and stream crossing failures. Effects of roads on erosion are described in Chapter 5.

Maintenance of an extensive, functioning road network is sometimes justified in part by the need for ongoing fuel treatment. To the extent that a road network supports fuel treatment, a sediment budget would associate the road-related sediment production with the fuel treatment efforts. For example, if 50 percent of the erosion-generating activity on a road is in support of fuel treatment, then 50 percent of the road-related sediment can be attributed to fuel treatment activities.

Strategies for Evaluating Influences

The previous sections describe how various fuel treatments *could* influence erosion processes. Whether such influences are likely to occur in a particular instance depends on the setting, treatment, and weather. Evaluation of the potential for erosion usually entails

1. examining the landscape to be treated to identify erosion processes already active at the site,
2. examining similar sites over a broader area to identify the less common processes that might occur, and
3. examining similar areas that have undergone similar treatments to identify the types of changes likely.

The questions to be addressed for each erosion process are

1. *In what settings* can the process occur? and
2. *Under what conditions* is it likely to occur in those settings?

Different erosion processes need different strategies for evaluation, require examination of different portions of the landscape, and are most usefully addressed at different spatial and temporal scales. Strategies are described here for evaluating channel-bank erosion, gullying, shallow landsliding, and deep-seated sliding. Similar strategies would be used to evaluate other channel erosion and mass-wasting processes if they are of concern at a project site.

Channel-Bank Erosion and Gullying

Evaluation of in-channel and gully erosion relies on information obtained by examining channel systems at several spatial scales. First, the types of channels and settings at and downstream of the project site are described. This work identifies past and current styles of channel behavior to answer the questions:

- Is there evidence of recent or older changes in channel plan-form?
- Is there evidence of recent or older channel aggradation or incision?

- Is there evidence of recent or older gully activity upslope of the current channel heads?
- Do banks show evidence of recent or older erosion?

If such evidence is found, the spatial and temporal patterns of the occurrences would be identified. Evidence of past episodes of gullying might take the form of bank-like scarps upstream of channel heads in otherwise unchanneled swales, uncharacteristically low width-to-depth ratios along small streams, or the presence of exposed tree roots in banks. If gullying was active in the past, the stream network may be particularly sensitive to small changes in factors promoting channel incision.

The next scale of inquiry examines sites outside of the project area that (1) are likely to be most susceptible to increased channel or gully erosion because of their history or setting and (2) have bedrock and topography similar to those of the project area. For in-channel erosion, these might include channels downstream of road drainage inputs or sites downstream of pervasively logged watersheds. Off-site examinations for gully erosion might focus on burned swales or sites where unprotected ditch-relief culverts empty into steep swales or headwater channels.

This portion of the evaluation describes potential mechanisms of influence on the processes, defines the tolerance of the landscape to change, and identifies the changes in controlling variables to which local erosion processes are sensitive. For example, if channels appear stable at the project site, but similar channels show extensive bank erosion below road surface drainage inputs, the effects of the proposed activities on runoff become a concern. Interpretation of the field observations requires evaluation of the storm history in the area. If no large storms have occurred since a road was constructed, lack of evidence for downstream destabilization cannot be considered evidence that destabilization is unlikely.

Field examination next turns to sites at which the proposed activities—or similar activities—have already been carried out. Here, too, evidence of channel destabilization and gullying is sought, as is evidence for influences on the controlling factors found to be important elsewhere. For example, if gullying was found at sensitive sites, treated sites would be examined for evidence of changes in swale-surface erodibility and surface runoff. In this case, too, sites for which challenging events have not occurred are not particularly useful.

The series of observations described above is intended to document the logic trail needed to support a diagnosis. Information presented might include:

1. descriptions of channel types in the area and downstream,
2. observations of those channel types in potentially erosive settings,
3. discussion of likely destabilization mechanisms at sites where erosion was observed,
4. description of channels and of any evidence for disruption at sites of analogous activities, and
5. description of evidence for the effects of the analogous management activities on conditions likely to influence the mechanisms in (3).

If evidence for an effect is found, or if information from elsewhere suggests that influences are possible, those effects would be further analyzed for the project site. Likely changes in runoff might be estimated through modeling, for example, and this change could then be compared to that present at sites where road drainage has destabilized channels. Particularly important issues for channel and gully erosion often include altered runoff, woody debris, and surface conditions.

In some cases, the weather conditions that provoke erosion may not have occurred recently enough to provide evidence of potential changes. Most channel changes occur during large floods, so conclusions will be weak if a geomorphically significant flood has not occurred in the past decade or so. Similarly, if the combination of disturbances and fuel treatment prescriptions are unprecedented for an area, evidence from past landscape behavior provides a weaker conclusion than it would had analogous conditions existed in the past.

Landsliding

Analysis of shallow landsliding requires special care because

1. such landslides often occur at sites where no evidence of previous destabilization is visible,
2. significant landsliding often occurs only with major storms,
3. the areal density of landslides is usually low even after landslide-generating storms,
4. even a few landslides can cause major impacts in downstream channels, and
5. the process is often of great public concern because it is frequently the most visible erosion process associated with land management activities.

Shallow landslides associated with forest management usually occur because management activities modify conditions at previously stable sites, destabilizing them. Consequently, site inspections cannot reliably reveal whether specific sites will become unstable after activities occur. The diagnosis instead must be made by identifying the site types present at the proposed activity site, assessing the inherent susceptibility of those site types to landsliding, and determining whether the proposed activities will increase that susceptibility. This strategy is essentially the same as that described for assessing susceptibility to channel-bank and gully erosion.

Shallow landsliding is generally associated with particular landscape features such as inner gorges and steep hillslope swales. Such associations simplify the analysis by allowing efficient stratification of the landscape into landslide-prone and stable areas. Analysis is also simplified because, in contrast to gullying and channel erosion, large slides are often visible on aerial photographs. Analysis usually begins with a broad-scale, air-photo-based evaluation of landslide distribution across the landscape using photos that pre-and post-date a major landslide-generating storm. Associations between landslide distribution and landform are first evaluated, then areal landslide densities are calculated for each landform type in areas that had undergone different types or ages of management activities at the time of the storm. Comparison between these values for different management activities provides estimates of the relative influence of the activities on landsliding for particular landforms.

In the hypothetical example shown in table 3, recent logging is associated with an overall landslide frequency 2.5 times that of unlogged areas, and most of the increase results from destabilization of headwater swales. Data for the same 1994 to 1997 storms for older logging suggest that the older sites are largely restabilized, showing only a 22 percent increase relative to unlogged sites. In most locales, the area of remaining old growth forest is too small to allow comparison between unlogged and logged areas, so changes relative to naturally occurring rates cannot be directly calculated. In such cases, definition of the recovery trend as a function of disturbance age can provide an estimate of the minimum change likely, or a simple comparison of rates in older and younger stands (such as the comparison between less than 15-year stands and greater than 15-year stands in table 3) might indicate a minimum likely change if management practices did not differ greatly between the periods.

Susceptibility to shallow sliding can be strongly influenced by hydrologic changes, and such changes can be generated by activities occurring upslope from potentially unstable sites. For example, concentration of road drainage onto a hillside can trigger failures downslope. It may thus be useful to test for relationships between downslope landslide frequencies and upslope activities.

Human activities can also influence characteristic landslide size, and landslides of different sizes often have different effects on impacted resources. If management activities increase landslide frequency and decrease landslide size, for example, biological responses may be important due to the altered spatial and temporal distribution of impacts even if the average rate of sediment input is unchanged (Chapter 12).

Once the association of landslides with particular landforms is defined and the relative susceptibility of those landforms to destabilization is understood, the project

Table 3. Hypothetical example of a rate calculation for shallow landsliding

Landform	Area of landform (ha)	Number of slides[a] 1994-97	Rate (slides/km^2)	Ratio to unlogged	Ratio to >15 yr logging
Unlogged					
Planar slope	1900	3	0.16	1	0.83
Headwater swale	210	2	0.95	1	0.69
Inner gorge	430	4	0.93	1	0.62
Other[b]	450	1	0.22	1	--
Total	*2990*	*10*	*0.33*	*1*	*0.82*
Logged within 15 years					
Planar slope	6300	19	0.30	1.91	1.59
Headwater swale	510	23	4.51	4.74	3.27
Inner gorge	1410	36	2.55	2.74	1.69
Other[b]	1510	3	0.20	0.89	--
Total	*9730*	*81*	*0.83*	*2.49*	*2.04*
Logged before 15 years ago					
Planar slope	3700	7	0.19	1.20	1
Headwater swale	290	4	1.38	1.45	1
Inner gorge	730	11	1.51	1.62	1
Other[b]	680	0	0	0	--
Total	*5400*	*22*	*0.41*	*1.22*	*1*
Grand total	18120	113			

[a] Only slides greater than 100 m^2 are tabulated to provide uniform resolution in different vegetation types.
[b] “Other” includes inherently stable sites such as ridge-tops and floodplains.

area—and areas downslope—can be examined for presence of the types of landforms found to be most important. The change in susceptibility can be estimated as the ratio between rates per unit area of the given landform on forested and treated sites (for example, table 3). The level of concern would be particularly high if the landforms of interest already show evidence of destabilization.

Examination of sites that have undergone analogous treatments is primarily intended to evaluate the types of influences the planned activities may have on controlling variables. Rarely would the sample area be large enough, and the storm history suitable enough, that influences on landslide frequency could be directly estimated on the basis of a site examination.

As with shallow landsliding, it is useful to begin an analysis of deep-seated landsliding with a broad-scale air-photo survey to identify patterns in the distribution of the process. If deep-seated slides are not already present in settings with bedrock and topography similar to those at the project site, they are not likely to be of concern unless significant earthwork is planned.

If deep-seated slumps and earthflows are present in similar settings, the broad-scale survey can be used to detect patterns in their activity levels. In particular, evidence can be sought to determine whether existing features can be reactivated or accelerated by management activities occurring on or upslope of the features. Remobilization of a dormant slump might be recognized by opening of new tension cracks at the base of the headscarp or by increased rates of shallow sliding from the toe. Temporal variations in activity level for earthflows can often be detected by using sequential aerial photographs to track the displacement of surficial features such as trees or roads.

The project site itself is then examined to determine whether deep-seated slumps or earthflows are present and evaluate their current activity levels. If either active or dormant features are present, activities that increase water inputs to the features would tend to increase their activity level. Evaluation of deep-seated features must usually be qualitative rather than quantitative.

Evaluating Cumulative Impacts

The preceding sections focused on understanding and evaluating the influences of fuel treatments on the distribution and rates of channel erosion and mass-wasting processes. Such analyses can provide estimates of the extent to which management activities might increase the erosion risk. However, increased erosion is of concern only if an increased risk of erosion is accompanied by an increased hazard to a resource or entity of concern. It is at this point that evaluation of the cumulative impacts of altered erosion becomes necessary.

From the point of view of an impacted resource, the effects of a land use activity on individual processes are not nearly as important as the overall effect of changes caused by the full distribution of activities through time and across the landscape. These cumulative effects are important to understand in two respects. First, the combined influences on a particular process must be understood if the net change in process rate is to be evaluated. For example, rates of shallow landsliding might increase because of both decreased rainfall interception and decreased root cohesion after canopy thinning. Second, the combined effects of all influences on a potentially impacted resource must be understood if the actual severity of the impact is to be evaluated. For example, increased flood damage caused by heightened rain-on-snow runoff may be further augmented by reduced channel conveyance due to landslide-related aggradation.

The Council on Environmental Quality (CEQ) has provided guidance (CEQ 1997) for preparation of cumulative impact analyses for documents prepared under the National Environmental Policy Act (NEPA), and federal courts have clarified standards for analysis through their opinions on cases involving such analyses (Chapter 14). Both the CEQ and the courts indicate that analysis is intended to evaluate the net impacts on particular resources. Consequently, a cumulative impact analysis for a fuel treatment project would need to assess the extent to which erosion rates influenced by the project might affect the nature and severity of impacts on specific resources.

The CEQ (1997) describes the analytical steps useful for cumulative impact analysis (table 1, Chapter 14), and these are readily applied to analysis for cumulative impacts associated with channel erosion and mass wasting. The procedure first defines impacted resources of potential concern in the area, as well as the resources, values, or issues of importance in the area that may not yet show impacts (step 1). Issues that might necessitate evaluation of erosion processes could include concerns over salmonid populations, downstream flooding, water quality, reservoir sedimentation, or any other impact that can be affected by altered sediment load or channel form.

For each entity of concern, the mechanisms through which impacts might occur are then identified. Potential interactions between impacts and sediment load may begin to become evident at this stage, though identified mechanisms need not directly involve erosion processes. Because the analysis is of *cumulative* effects, the broader context in which sediment-related influences occur must be evaluated. In the case of concerns over increased downstream flooding, for example, potential mechanisms of interest that are not directly related to a project's sediment inputs might include increased peakflows, reduced channel conveyance due to vegetation encroachment, changes in reservoir management strategies due to increased irrigation demands, and increased residential development on floodplains. Sediment-related mechanisms could include reservoir sedimentation and reduced channel conveyance due to aggradation.

An understanding of the distribution of the resources of interest and of the nature of relevant impact mechanisms then allows definition of the spatial and temporal scales needed to analyze impacts on each resource (steps 2 and 3). Because different impacts are expressed at different sites and over different time scales, analysis scales will differ for each kind of impact, and sometimes for each impact mechanism. For example, most influences on flood hazard need to be evaluated over the watershed upstream of sites susceptible to flooding, but prediction of the contribution of changes in reservoir management strategy to flood hazard requires consideration of socio-economic influences over a much broader scale.

At this point, a general overview is useful of the types of activities that have occurred in each relevant analysis area and that may affect the resources of concern (step 4). Evaluation of the nature and timing of past changes is facilitated if the distribution and timing of past agents of change are known. An assessment of cumulative impacts on flood hazard, for example, might evaluate the history of vegetation change in the upstream watershed and assess the timing and distribution of activities that can influence rates of sediment production.

Examination of recent legal opinions suggests that defining the significance of environmental changes is particularly challenging (Chapter 14), and the next analysis steps outline a strategy for addressing this problem. First, the types of coping strategies or responses to each potential impact are described for each resource (step 5). This information provides a basis for evaluating how much of a change is tolerable before a resource becomes impaired. Impacts on some issues of concern, such as municipal water supplies or transportation infrastructure, can be assigned economic values. In other cases, regulations may have established particular thresholds of significance. At this stage, examination of natural disturbance patterns can be very useful. Many resources of concern (such as endangered species) developed in the context of the spatial and temporal variations in conditions that occurred before Euro-American settlement. For these, deviations from the natural patterns define the levels of stress currently experienced (step 6). In some cases, attempts have been made to place current conditions in the context of very-long-term averages for sediment inputs obtained by cosmogenic isotope work. However, if the "range of natural variability" is found to have included an extreme sediment-generating event 2,000 years ago, that event did not occur in the context of the other changes that are present today. If such major events are found to have occurred naturally, the impact assessment would need to evaluate the cumulative effect of the modern land use activities on the ability of impacted resources to recover from such an event.

The CEQ then suggests that baseline conditions be described (step 7), and explains that "*the baseline condition of the resource of concern should include a description of how conditions have changed over time and how they are likely to change in the future without the proposed action*" (CEQ 1997, p. 41). In this context, establishing the baseline requires description of the pre-Euro-American conditions, comparison of those to today's conditions, and description of the current trajectory of conditions. Of these tasks, description of the pre-Euro-American conditions is usually the most challenging because undisturbed conditions are rarely available for comparison. Instead, those conditions generally must be inferred from (1) observation of sites with *relatively* undisturbed conditions, (2) an understanding of the history of change in an area (step 4 above), and (3) historical evidence, including old snapshots, early aerial photographs, oral histories, and so on. For the flood hazard example, baseline conditions would be established for flood frequencies at susceptible sites as well as for the conditions influencing those frequencies. Baseline channel conveyance, for example, might be established by using historical aerial photographs to evaluate changes in channel form and bank vegetation, and old surveyed cross sections may be available from bridge construction sites.

At step 8, the cause-and-effect relations between human activities and impacts are identified. For increased flooding, anthropogenic effects on peakflows and channel conveyance would be evaluated at this stage. Most impacts are influenced by a variety of mechanisms, many of which do not involve sediment production. Anthropogenic effects on channel conveyance, for example, might include vegetation management on channel banks or construction of levees.

Past and on-going land use activities would then be evaluated to determine the extent to which they influence conditions that affect impact levels. The existing cumulative impact on a resource is the overall impairment caused by the combined mechanisms (step 9). At this stage, the potential influences of a proposed activity would be examined to determine whether they could contribute to the identified impacts. The first eight analysis steps outlined potential impact mechanisms relevant to the particular setting, thus narrowing the scope of the portion of the analysis relating specifically to erosion processes. It now becomes possible to match the potential changes likely to be associated with altered erosion (for example, column 5 in table 1) with the types of changes relevant to the impacts of concern. Many potential influences would be discarded at this stage if they are found not to be relevant in the particular system being evaluated.

To evaluate impacts associated with erosion processes, the proposed activities would first be analyzed to identify their potential influences on particular processes (table 2). Each of the impact mechanisms identified for flood hazard, for example, could then be evaluated to determine whether the potential influences from the relevant erosion processes (table 1, column 5) can affect the impact mechanism. For example, a project involving road construction for salvage logging in steep terrain might be expected to generate shallow landsliding (table 2), which could contribute to aggradation (table 1). If analysis has suggested that reduced channel conveyance is of concern, for example, potential changes in aggradation could contribute to a cumulative impact on flood hazard.

The final steps outlined by the CEQ allow for use of the analysis to redesign or mitigate the project (step 10) and for monitoring and modification of the project after it is implemented (step 11). The preceding analysis steps provide much of the information needed to design efficient and effective mitigations and monitoring projects.

Impact analysis under NEPA also requires analysis of the cumulative impacts associated with the "no action" alternative. In the case of salvage activities, such analysis would ordinarily consider impacts of the disturbance event and any rehabilitation work associated with the event. Analysis of the project's effects on channel erosion and mass wasting would need to address how changes from the project would modify the effects of the event on those processes in the absence of the project. At some sites, for example, woody debris contributed to small streams by a fire might trap large volumes of sediment eroded from burned slopes, and salvage of the wood could increase the downstream sediment flux relative to the effect of the fire alone. In addition, if Euro-American management practices influenced the extent or character of the infestation or fire, some aspects of the initial disturbance are themselves anthropogenic, and those influences would need to be evaluated as a component of cumulative impacts.

Analysis of the role of fuel treatment projects in cumulative impact generation will also be required during impact evaluation for future projects of other kinds. At that time, the erosional consequences of past fuel treatment projects—even if carried out under categorical exclusions under NEPA—would need to be evaluated since they will have become "past projects" that provide the context for newly planned projects.

Cumulative impact analysis is necessarily an interdisciplinary exercise. Here we consider effects on mass-wasting and channel erosion processes, but similar analyses would be carried out for effects on hydrology, vegetation, wildlife, and so on. In the real world, all of these influences interact. As analysis proceeds for each of these environmental components, linkages between components are identified and incorporated into the analysis. For example, when likely hydrological changes are identified, their influences on erosion processes can be evaluated. Influences on the impact mechanisms from other previous and potential future activities would also be analyzed using a similar approach. In particular, implications of the combined influence of past and proposed activities in prolonging the duration and spatial extent of the impacts would need to be considered. Interdisciplinary analysis is also important because a change in erosion rate is of interest in cumulative impact analysis only insofar as it influences an impact of concern, so evaluation of the significance of erosional changes may require an understanding of fisheries biology, riparian ecology, structural engineering, or any of a number of other fields.

As in many areas of human endeavor, forest management decisions must be made even though knowledge of the likely outcomes of those decisions is incomplete. Even if knowledge of erosion processes were perfect, precise outcomes could not be predicted because future weather conditions are unknown. The challenge for cumulative impact analysts is to use knowledge from a broad range of disciplines as effectively as possible to allow adequately informed decision-making.

Conclusions

The broad range of potential fuel treatment practices can influence channel erosion and mass-wasting processes in a variety of ways, so the erosional outcome from a particular project depends strongly on the nature and setting of the project. Thus, each application must be evaluated in its own right to assess potential impacts. Even though the effects of particular fuel treatment activities on erosion rates have not been widely studied, information from a variety of other studies is applicable to the problem if the factors controlling the distribution and rates of erosion processes are understood and the effects of particular fuel treatment practices on those factors can be determined. In general, fuel treatment activities that modify soil conditions, hydrologic conditions, vegetation, or hillslope or channel morphology are likely to influence the rates and distribution of channel erosion and mass-wasting processes.

If the resulting influences adversely affect an entity that is experiencing impacts from other sources as well, the fuel treatment project will have contributed to the cumulative impact on that entity. If the relevant impact mechanisms are understood, the potential effect of erosion from a planned fuel treatment project on the cumulative impact can be evaluated by determining the extent to which the types of erosion that are likely to be associated with the project will influence those impact mechanisms.

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Cumulative Watershed Effects of Fuel Management in the Western United States

Chapter 7.

Fuel Management and Water Yield

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Introduction

There have been numerous studies worldwide demonstrating that changes in forest density can cause a change in water yield. Bosch and Hewlett (1982), Hibbert (1967), Stednick (1996) and Troendle and Leaf (1980) have summarized the findings from most of these studies. In general, as Hibbert (1967) observed, reducing forest cover increases water yield; establishing forest cover on sparsely vegetated land decreases water yield; and response to treatment is highly variable and, for the most part, unpredictable.

Although the first two of these conclusions are still accepted, the hydrologic response to changes in forest cover, although variable, is more predictable than Hibbert (1967) concluded (Bosch and Hewlett 1982; Stednick 1996; Troendle and Leaf 1980). This change in thinking results from the increased number of observations available with each successive review and an improved understanding of the factors influencing streamflow response. Streamflow response to a change in forest cover is strongly related to climate, species composition, and the percentage change in vegetation density (fig. 1). The data from 95 watershed experiments conducted in the United States show that, on average, annual runoff increases by nearly 2.5 mm for each 1 percent of watershed area harvested (Stednick 1996). Because runoff is quite variable from year

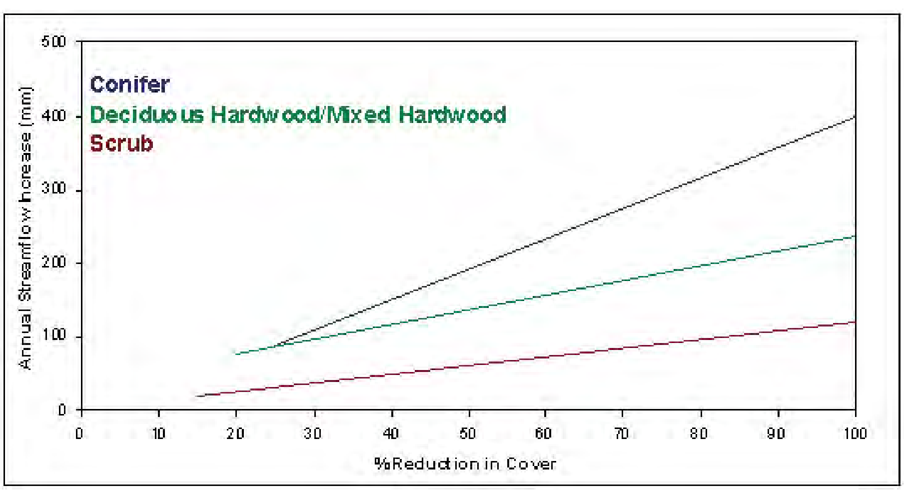

Figure 1. The relationship between reduction in vegetation cover and increases in stream flow for three vegetation types (redrawn from Bosch and Hewlett 1982).

to year, the general conclusion is that approximately 20 percent of the basal area of the vegetation must be removed before a statistically significant change in annual runoff can be detected (Bosch and Hewlett 1982; Hibbert 1967; Stednick 1996). However, as Bosch and Hewlett (1982) suggest, reductions in forest cover of less than 20 percent (fig. 1), particularly in more humid areas, may well produce statistically non-significant increases in streamflow that would presumably decrease to zero increase in streamflow at zero reduction in forest cover.

Much of our understanding about the effects of forest disturbance on water yield has come from paired watershed experiments. Unfortunately, very few of these catchment-scale experiments provide data on the hydrologic response to fuel reduction since the vast majority of the treatments imposed a partial or complete clearcutting of the mature trees rather than a partial cut or thinning (Stednick 1996). Hence, much of our understanding of the hydrologic impacts of thinning and prescribed fire comes from inference supported by various plot and process studies.

Objectives

This chapter has three objectives pertaining to the effects of fuel management on water yield:

1. Determine whether regionalization can help reduce the variability in treatment response that made streamflow change unpredictable according to Hibbert (1967).
2. Assess the effect of forest disturbance on each component of the water balance—interception and evaporation, transpiration, infiltration, and storage—and use this to help infer how fuel reduction treatments may impact annual water yields as well as peak and low flows.
3. Identify tools that can help hydrologists and land managers predict both the on-site and cumulative changes in water yield that may result from vegetative treatments.

Hydrologic Impact of Forest Disturbance on Streamflow Characteristics

Regionalization of Hydrologic Response

The hydrologic cycle represents the processes and pathways involved in the circulation of water from land and water bodies to the atmosphere and back again. An understanding of the hydrologic cycle is fundamental to understanding the effects of different forest practices on key components of the water balance, including soil moisture and streamflow. The hydrologic cycle is often expressed in the form of a water balance or continuity equation:

$$\text{Runoff} = \text{Precipitation} - (\text{Evaporation} + \text{Transpiration Loss} \pm \text{Change in Storage}) \quad (1)$$

Precipitation can be in the form of rainfall, snowmelt, or fog drip; evaporation includes evaporation from both the soil and the surface of plant canopy and litter (interception loss); and change in storage includes soil moisture and groundwater. Evaporation and transpiration are usually regarded as losses and reduce the amount of precipitation that is transformed into runoff. Changes in storage can be very important over short time periods (in other words, seasonal or less than 1 year), but are generally assumed to be zero over long periods unless there is continuing groundwater extraction.

Although the components of the hydrologic cycle are always the same, the relative importance of each component can vary considerably with geographical location and from season to season. The complex interactions between climate and vegetation

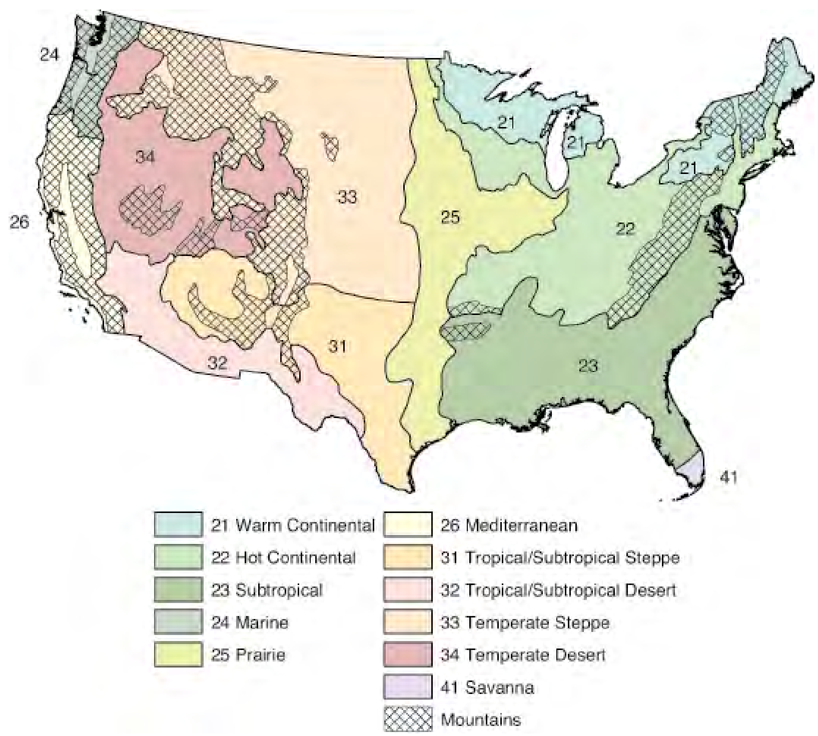

Figure 2. Ecoregion classification based on Bailey (1994).

control the role that the individual components play in the water balance and the influence that forest disturbance has on the water balance.

In the mid-1970s, Bailey (1976, revised 1994) developed an ecoregion map of the United States that depicted the relationship between vegetation patterns, climate, and landscape or topography (fig. 2). Vegetation types with similar moisture and energy requirements were found to be present in reoccurring patterns within and between ecoregions for given site conditions. Differences in vegetation patterns between ecoregions were primarily influenced by differences in the amount and seasonal distribution of water and energy. Precipitation and energy are largely controlled by elevation, latitude, aspect, topography, prevailing wind direction, and proximity to oceans, and the balance between precipitation and energy are the primary controls on streamflow. The concepts used in developing the ecoregion classification for the United States (Bailey 1976) have been successfully applied to North America (Bailey and Cushwa 1981) and the rest of the world (Bailey 1998). Ecoregion classification provides a useful framework for stratifying hydrologic response and predicting the effects of vegetation manipulation on water yield.

In the late 1970s, the ecoregion concept (Bailey 1976) was used to stratify the United States into seven hydrologic regions (fig. 3) in order to better predict the effects of silvicultural activities on non-point source pollution (EPA 1980). The resulting handbook, *An Approach to Water Resources Evaluation of Non-Point Silvicultural Sources*, is commonly referred to as WRENSS. This contained graphical procedures, stratified by region, to predict the effects of forest disturbance on streamflow, erosion, temperature, and nutrients. The same hydrologic regionalization is being used here to help understand and predict the effects of forest disturbance, including fuel reduction treatments, on annual water yield, peak flows, and low flows.

This chapter will emphasize the hydrologic effects of fuel reduction treatments in the Rocky Mountain region (WRENSS Hydrologic Region 4 in fig. 3) for the following reasons. First, this synthesis focuses on the western United States. Second, more than 50 percent of all National Forest System lands are contained within the Rocky Mountain region, and excessive fuel loadings and departures from historical ecological conditions are particularly severe (Romme and others 2003). These factors mean that fuel reduction programs are most likely to be concentrated in this region.

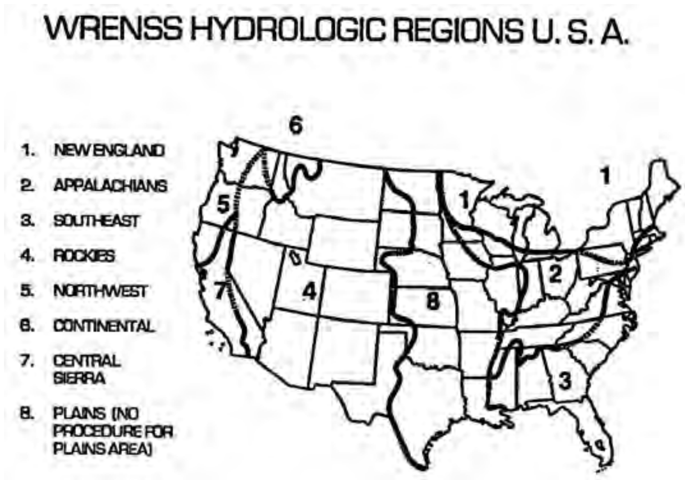

Figure 3.The United States divided into seven hydrologic regions (see Troendle and Leaf 1980).

In addition, the hydrologic effects of fuel treatments are potentially more important in this region because possible changes in water yield may be more important and longer-lived than in other regions. From a process standpoint, the Rocky Mountain region provides examples of how vegetation management strongly influences snowpack accumulation and melt as well as summer evaporation and transpiration. This means that the processes being discussed here will cover the key processes for nearly all forested areas in the United States, even though the relative magnitude will vary by region. This chapter first discusses and explains the observed variations in the hydrologic responses to forest disturbance in the Rocky Mountain region, uses these observations to predict the likely effects on fuel reduction treatments, and then compares these responses to the potential hydrologic responses that may occur in other ecoregions in the western United States.

Water Yield

Numerous publications have quantified the effects of forest disturbance on streamflow in different regions of the United States (for example, Brown and others 1974; Debano and others 2004; Douglas 1983; Gary 1975; Harr 1983; Hornbeck and others 1997; Kattleman and Ice 2004; Keppeler and Zeimer 1990; NRC 2008; Reinhart and others 1964; Troendle and Leaf 1980; Troendle 1983). These studies have shown that the magnitude of change in water yield is most strongly related to the amount of precipitation and the proportion of forest cover that is removed. The increases in flow following forest disturbance can be quite large in the humid southeast, northeast, north central, and northwest regions of the United States. In contrast, increases in water yield due to removing woody vegetation are an order of magnitude smaller in drier areas such as the Southwest. The effect of a decrease in forest density on water yield can occur regardless of whether this is due to disturbance from fire, insects, disease, or timber harvest. Afforestation or an increase in forest density generally has the opposite effect on water yield than does forest removal.

The magnitude of the hydrologic response to disturbance between years will depend on the summed effect of the changes in processes as indicated by equation 1. These include the degree to which the management activity alters net precipitation to the soil by altering interception losses and infiltration characteristics and the soil moisture evaporation and transpiration. The timing of a change in streamflow within a year depends on when precipitation or snowmelt exceeds both evapotranspiration demand and soil moisture recharge requirements. Hence, any effort to predict the effect of forest disturbance on water yield requires an understanding of how the disturbance affects the water balance with respect to the amount and timing of precipitation inputs (whether there are changes in flow pathways) and the degree to which soil moisture storage and recharge requirements have been altered as a result of changes in evaporation and transpiration.

Rocky Mountain Region

The Rocky Mountain region is fortunate to have a long series of carefully controlled paired watershed experiments to evaluate the effects of forest harvest on water yield. Worldwide, the first such experiment was conducted at Wagon Wheel Gap in south-central Colorado on the headwaters of the Colorado River (Bates and Henry 1928; Troendle and King 1987; Van Haveren 1988). This study was followed by a series of watershed studies in the region (Hoover and Leaf 1967; Stednick and Troendle 2004; Troendle and King 1985, 1987; Troendle and Leaf 1981), but the longest running and most detailed study has been conducted on the Fool Creek watershed on the Fraser Experimental Forest in central Colorado (Hoover and Leaf 1967; Troendle and King 1985).

The results of the Fool Creek study are presented in more detail because this is the longest and most comprehensive study in the Rocky Mountain region, and the process-based understanding developed at Fool Creek applies throughout the snowmelt dominated Rocky Mountain region. Mean annual precipitation at Fool Creek is about 760 mm per year while annual evapotranspiration ranges from 450 to 570 mm per year (Troendle and King 1985). At a nearby study site, the average annual evapotranspiration (ET) is directly proportional to precipitation once precipitation exceeds 462 mm (18.2 inches) (Troendle and Reuss 1997).This relationship can be expressed by:

$$ET_{mm} = 462\text{mm} + 0.284\ (\text{Precipitation}_{mm} - 462\text{mm}) \qquad (2)$$

Annual water yields from the Fool Creek watershed were calibrated against East St. Louis Creek for a 15-year period. From 1954 to 1956, approximately 40 percent of the Fool Creek watershed, or 50 percent on the commercially forested area, was clearcut in alternating cut and leave strips. Comparison of the mean annual hydrographs for the 15-year calibration period and the first 15 years after harvest clearly shows that, on average, forest harvest increased both annual and peak flows (fig. 4). Numerous other studies have shown that the changes in runoff shown in figure 4 are typical of the effect of forest disturbance in the cold snowmelt region typical of the Rocky Mountains (Bates

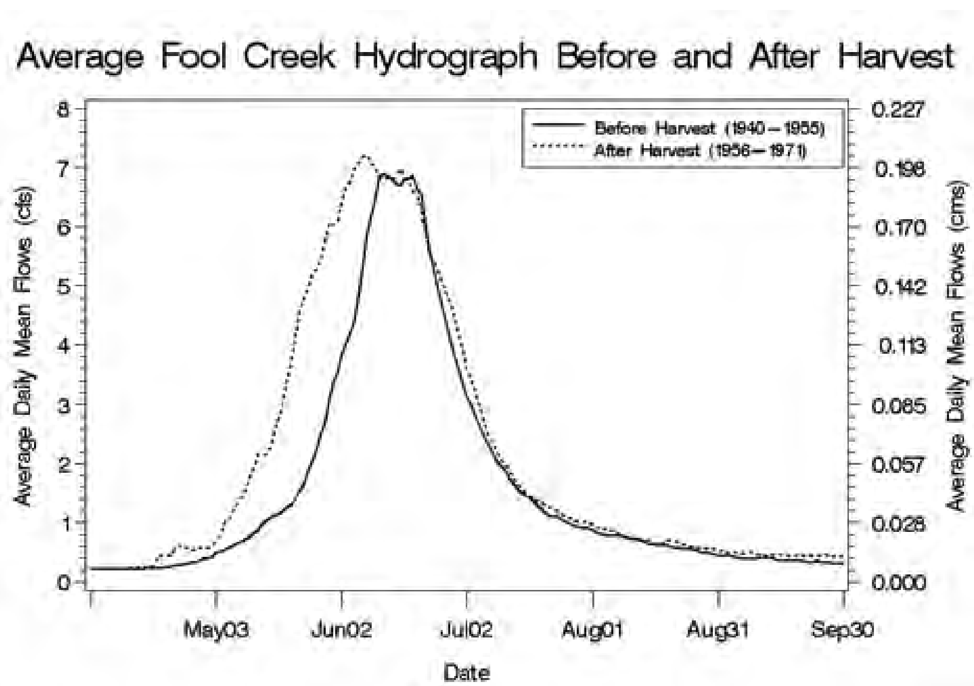

Figure 4.The average hydrograph from the Fool Creek Watershed for the 15-year period both before and after treatment.

and Henry 1928; Swanson and Hillman 1977; Swanson and others 1986; Troendle and Bevenger 1996; Troendle and King 1987; Troendle and Reuss 1997).

Within the first 15 post-treatment years, the observed increases in annual water yield have ranged from a high of 16.2 cm in the wettest year to a low of 3.6 cm in the driest year. The "average" first-year response to the treatment on Fool Creek was equivalent to a 10.0 cm increase in seasonal water yield (Troendle and King 1985). During the 1956 to 1983 post treatment period, on average, these increases are due to the 50 percent reduction in the annual ET that would have occurred on the clearcut portion of the watershed (Troendle and King 1985; Troendle and Reuss 1997). By 1983, 28 years after harvest, regrowth in the clearcuts was causing a significant decline in the average annual water yields.

A month-by-month analysis showed that significant increases in flow occurred primarily in May with only an occasionally significant increase in June. No detectable impact has been documented on flows from July to October (Troendle and King 1985). There is little opportunity for measurable increases in water yield to occur during most of the growing season because summer evapotranspiration is limited by the amount of available water. The high summer water deficit explains why, on average, less than 5 percent of summer rainfall is transformed into streamflow (Bevenger and Troendle 1987; Garstka and others 1958; Troendle and King 1985) and the reduction in summer evapotranspiration does not detectably increase summer or fall streamflow.

The observed changes in the cold snow zone's water yield after forest removal are due to both a decrease in winter interception and a reduction in growing season soil moisture depletion (Dietrich and Meiman 1974; Goodell and Wilm 1955; Potts 1984b; Troendle 1987, 1988; Troendle and Meiman 1986; Troendle and Reuss 1997; Wilm and Dunford 1948). In the cold snow zone, such as at Fool Creek, precipitation accumulates over the winter as snow pack, with minimal melt over this accumulation period. When the snowpack begins to melt in spring, the melt water first recharges the soil by replacing the water that was depleted during the previous growing season. Once soil moisture storage is filled, the excess meltwater is available to become streamflow. At Fool Creek, which is comprised mostly of east- and west-facing slopes, approximately 30 percent of the increase in water yield can be attributed to the decrease in interception and resultant increase in the amount of water contained in the snowpack. The reduced evapotranspiration during the previous summer also reduces the amount of meltwater needed for soil moisture recharge in the clearcut. This process accounts for approximately 50 percent of the increase in water yield. The remaining 20 percent of the observed increase in water yield results from the reduction in evapotranspiration losses during April and May (Troendle and King 1985).

On north-facing slopes, the reduction in winter interception losses can account for more than 50 percent of the increase in annual water yield after forest harvest (Troendle and King 1987; Troendle and Meiman 1986). In contrast, on south-facing slopes, the reduction in winter interception may account for only 20 percent of the observed change in annual water yield. This difference in the role of winter interception is because the snow remains in the canopy almost continuously from November to May on north-facing slopes. The tremendous surface area of exposed snow allows a great deal of evaporative loss to occur despite the lower incoming solar radiation compared to south-facing slopes, and this can be attributed to the low relative humidity and relatively strong vapor pressure deficit. On south-facing slopes, the forest canopy is less dense and intercepts less snow, and the intercepted snow is more likely to melt and fall out of the canopy, thus allowing less time for interception losses to occur. Surprisingly, the 13- to 19-cm change in summer evapotranspiration appears to be independent of aspect and varies primarily with the amount of annual precipitation (Troendle 1987, 1988).

A similar hydrologic response to Fool Creek has been documented following forest harvest for other watershed studies in the Rocky Mountain region, and these include Wagon Wheel Gap in south-central Colorado (Bates and Henry 1928), Dead Horse Creek in central Colorado (Troendle and King 1987), Coon Creek in southern Wyoming (Troendle and others 2001), thinning in South Dakota (Anderson 1980), tree mortality due to insect attacks in northwestern Colorado (Love 1955), beetle kill in southwestern

Montana (Potts, 1984a), and wildfire in northern Wyoming (Troendle and Bevenger 1996). It should be noted that the forest cover at Wagon Wheel Gap was predominantly aspen as compared to mostly conifers in all of the other watersheds.

At lower elevations in the Rockies, there is less snow, summers are hotter and drier, and the dominant tree species are ponderosa pine and Douglas-fir. The fire regimes in the ponderosa pine and mixed Douglas-fir/ponderosa pine forests have been severely affected by fire suppression. It is within these forest types that most of the thinning and prescribed fire treatments are being proposed or taking place. A comparison of the potential water yield changes from these sites to the more extensively studied changes in the cold snow zone helps illustrate the magnitude and causes of the variations that confounded Hibbert in 1967.

Studies in southwestern Idaho indicate that forest harvest causes smaller and less persistent increases in annual water yields than at Fool Creek. The observed changes in water yield are strongly influenced by aspect. At Boise Basin, 42 percent of the area was logged or burned in 1929 and 1930. Ten years later the increase in annual water yield was only 7.6 mm (Rosa 1961). At Silver Creek, 23 percent of the basin was clearcut, primarily on south-facing slopes, and there was no detectable increase in water yield. In contrast, a clearcut and prescribed fire on a nearby 1-ha catchment with a northerly aspect doubled the subsurface flow during spring runoff. This doubling in runoff was due to a 5- to 8-cm decrease in on-site evapotranspiration. As in the cold snow zone, these increases in water yield were smaller when there was less precipitation and greater energy input. These results indicate that the general lessons learned from the cold snow zone apply to lower elevation sites in the Rocky Mountain region, but the predicted increases in water yield from lower elevations must take into account the lower precipitation and greater differences by aspect.

Equation 1 and the results from the Rocky Mountain region show that the increased runoff after forest disturbance is the integrated response to the amount and timing of precipitation and snowmelt inputs, soil moisture recharge requirements, and the evapotranspirational demands at the time of soil moisture recharge. In the humid, rain-dominated regions in the eastern United States, the maximum increases in flow can be as much as 300 or 400 mm per year following clearcutting (Hornbeck and others 1997; Stednick 1996). Unlike the cold snow zone, these increases in streamflow often occur in the late summer and early fall because that is when precipitation begins to exceed the reduced amount of soil moisture recharge in the harvested areas. Once soil moisture recharge is satisfied in the harvested and unharvested areas, the only difference in winter water yield and peak discharges will be due to the difference in rainfall interception losses between the harvested and unharvested areas. In the snow-dominated areas, the timing of the water yield increase would include a spring component similar to the Rocky Mountain region, although there would also be an increase in growing season flows if there is sufficient precipitation. Overall, the magnitude of the changes in annual water yield do not differ greatly between the Northeast (Hydrologic Region 1, fig. 3) and the Rocky Mountain region (Hydrologic Region 4, fig. 3) for a similar reduction in basal area. However, the increases in water yield tend to be less persistent because of the relatively rapid vegetative regrowth after forest harvest, and some long-term studies indicate a decrease in summer and annual streamflow 25 to 35 years after harvest. A plot of the data from the Northeast also suggests that a detectable change in water yield can occur after removing only 10 to 12 percent of the basal area (Hornbeck and others 1997), and this can be attributed to the wetter conditions during the summer growing season.

The effect of timber harvest on water yield from the "warm" snow and rain-on-snow zones of the Cascades of Oregon and Washington (Hydrologic Region 5, fig. 3) and the Sierra Nevada of California (Hydrologic Region 7, fig. 3) have both similarities and differences in response compared to that of the cold snow zone of the Rocky Mountain region (Kattleman and Ice 2004; Rice and others 2004). Snowpacks are generally at or near 0 °C, so snowmelt can occur during the winter and precipitation can occur as snow, rain, or a mixture of snow and rain at any time throughout the winter. Nearly all forests are coniferous, so winter interception losses are higher than in the deciduous forests of the east, but less than the interception losses in the cold snow zone in the Rocky Mountain region because the snow rapidly melts out of the canopy in all but the highest elevation zones. Forest harvest in areas with high annual precipitation and high soil moisture storage capacity can cause a greater reduction in summer evapotranspiration than in the cold snow zone forests like Fool Creek, and this can lead to more soil moisture carryover relative to uncut sites. This difference in soil moisture carryover can lead to larger increases in annual water yields, such as in the first-year values of 300 to 400 mm that have been observed from paired watershed studies in Oregon (Jones and Post 2004; Stednick 1996). As in the eastern United States, the relatively rapid regrowth means that the water yield increases due to forest harvest are typically eliminated in a much shorter time than at Fool Creek, and the rapid regrowth can lead to a decrease in summer and annual water yields within 1 to 3 decades after harvest.

Precipitation and Interception

Throughout much of the Rocky Mountain hydrologic region, the annual hydrograph is dominated by the melting of the winter snowpack. In snow dominated areas of the western United States, the amount of water present in the snowpack on 1 April can explain from 60 to 90 percent of the variation in annual runoff (Bevenger and Troendle 1987; Garstka and others 1958; Troendle and King 1985). Overall, as much as 95 percent of the total annual streamflow in the cold snow zone originates as melting snow, while only 3 to 5 percent of the rainfall becomes stormflow. In contrast, up to 24 percent of the rainfall can be returned as stormflow in some of the more humid areas in the eastern United States, and this can approach 70 percent for some rainstorms under exceptionally wet antecedent conditions (Hewlett and others 1977; Woodruff and Hewlett 1970).

In the cold snow zone of the Rocky Mountains, virtually any reduction in stand density will increase snowpack accumulation (for example, Gary and Troendle 1982; Haupt 1979; Meiman 1970, 1987; Packer1962; Troendle and Kaufmann 1987; Wilm and Dunford 1948). In the higher elevation lodgepole pine and spruce-fir forests, the increase in the snowpack on partially cut stands is directly proportional to the percent of basal area removed and the increases observed in clearcuts (Goodell 1952; Love 1953; Troendle and Meiman 1984; Wilm and Dunford 1948). As noted previously, the increases in peak snow water equivalent in the cold snow zone after forest disturbance

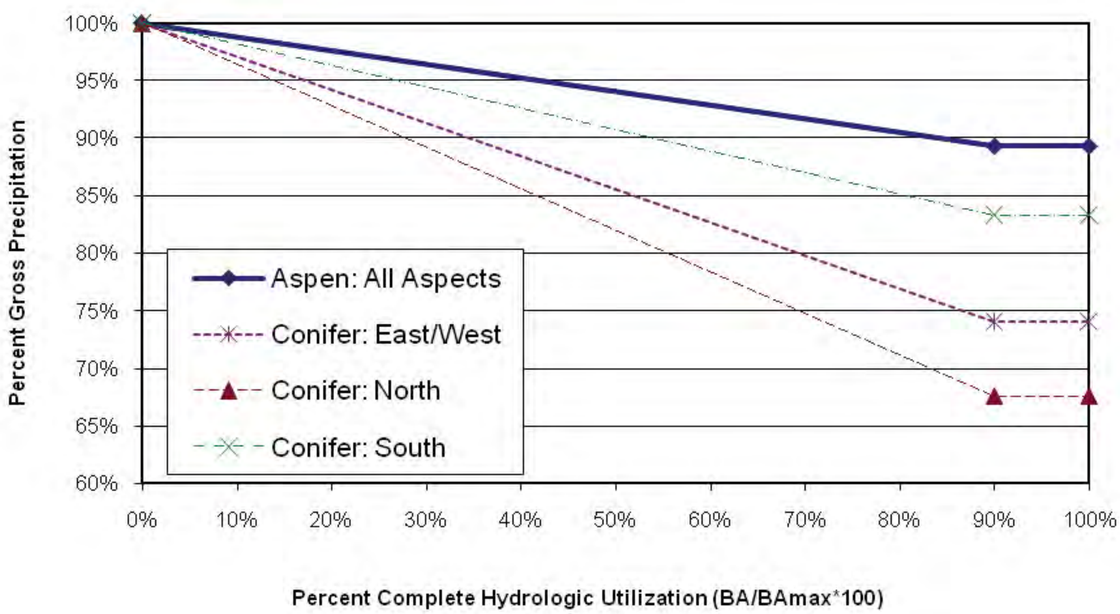

Figure 5. Percent of gross precipitation reaching the forest floor in the Central and Northern Rocky Mountains as stand density increases from 0 (opening) to 100-percent of the maximum basal area for the site (from Troendle and others 2003).

are greatest on north-facing slopes and smallest on south-facing slopes. Increases on east- and west-facing slopes are intermediate (Troendle and others 2003, 2005; fig. 5). In drier ponderosa pine forests, a reduction in basal area did not detectably increase the snow water equivalent on south, east, and west aspects, but did substantially increase the snow water equivalent on north-facing slopes (Haupt 1979).

The amount of interception loss during and after individual snowfall events varies significantly with storm size, storm intensity, wind speed, and location (in the case of small clearings). In coniferous forests in the cold snow zone, one generally can expect that 25 to 35 percent of the winter snowpack will be intercepted and lost to the atmosphere by some combination of sublimation and evaporation. Timber harvest in a deciduous forest, such as an aspen forest, will also increase peak snow water equivalent, but the much smaller amount of canopy reduces winter interception losses to about 10 to 12 percent. Process-based studies have shown that the observed increases in the winter snowpack after forest harvest result from the reduction in interception losses rather than a redistribution of snow during or after a storm event (Schmidt and Troendle 1989, 1992; Troendle and King 1987).

The magnitude and significance of interception losses by forest vegetation to the overall water balance have been documented by Kittredge (1948), Coleman (1953), and others. Interception losses may account for 25 to 35 percent of the annual precipitation, depending on the amount, type, and intensity of precipitation and the type and density of forest vegetation. In the cold snow zone, the effect of a reduction in winter interception on water yield is particularly important because the storm-based changes in interception accumulate over the course of the winter and can represent a significant increase in water inputs during spring melt. The snow interception losses measured in the cold snow zone of the Rocky Mountain region are surprisingly consistent with values from other cold snow regions in the United States (Troendle and Leaf 1980). The increase in net precipitation resulting from forest removal is proportional to the reduction in stand density and can range up to 15 to 30 percent for individual storm events (Kittredge 1948).

Soil Moisture and Summer Evapotranspiration

The effects of a change in stand density or leaf area index on summer evaporation, and especially transpiration, are not as linear as the changes in snowpack accumulation. Clearcutting in the Central Rocky Mountains reduces on-site soil moisture depletion by 13 to 19 cm during the growing season, regardless of aspect (Dietrich and Meiman 1974; Troendle 1987, 1988; Troendle and Kaufman 1987; Troendle and Meiman 1984;

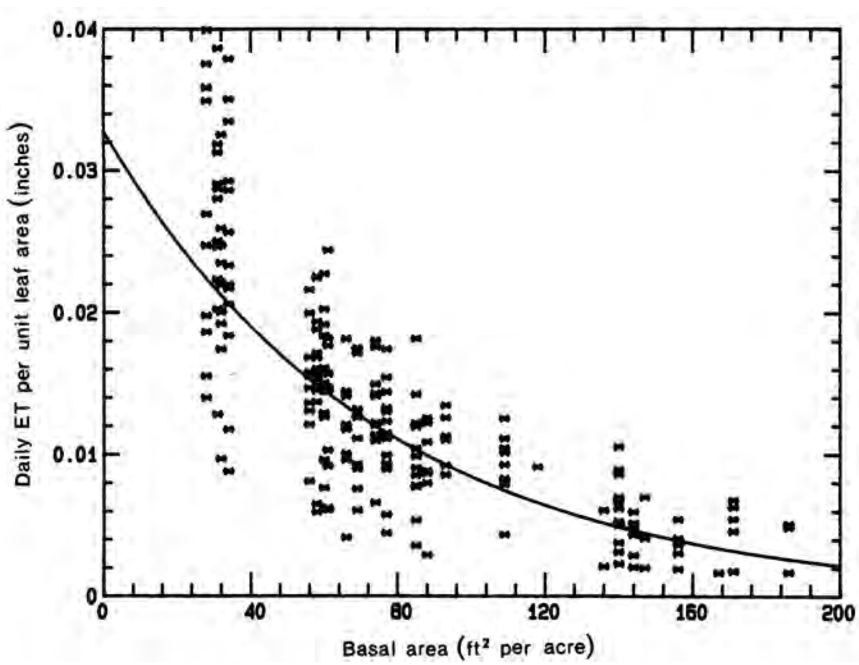

Figure 6. Daily ET (soil water depletion) per unit of leaf area as a function of basal area (from Troendle 1987).

Wilm and Dunford 1948). However, thinning can have very little effect on summer evapotranspiration rates as the residual trees can capture some or all of the savings in soil water by increasing water use (MacDonald 1986; Troendle 1987). The variations in the relationship between leaf area index and daily evapotranspiration (fig. 6) illustrates the ability of trees to adjust their water use in accordance with soil moisture availability as well as other factors. This means that the relationship between stand density and soil water depletion is statistically significant in wet years when there is less competition for soil water, while in dry years, there may be no correlation between basal area and soil water depletion because evapotranspiration from the residual stand may use all of the available water, regardless of the reduction in stand density (Troendle 1987). Hence, the potential for thinning to reduce summer transpiration and increase water yields depends on the amount of precipitation. If the sum of the water stored in the soil and summer precipitation exceeds potential evapotranspiration, thinning may increase the amount of water available for streamflow because of the reduction in summer evapotranspiration. If the sum of the stored water and summer precipitation are less than the potential evapotranspiration, any reduction in evapotranspiration due to thinning or forest harvest will be lost to evaporation from the soil and transpiration by the residual vegetation. If there is not a reduction in summer evapotranspiration there will not be any reduction in the amount of water needed for soil moisture recharge. In water limited systems, such as most of the Rocky Mountain hydrologic region, summer precipitation is low and soil water reserves are often depleted on all aspects and across a wide range of stand densities and forest types. Therefore, it is unlikely that most fuel reduction treatments will sufficiently decrease soil water depletion to cause an increase in annual water yields unless precipitation amounts exceed evaporative demand. In most areas, the only mechanism for fuel treatments to increase water yields is to (1) reduce interception losses and thereby increase rainfall runoff during the winter when soils are relatively wet or (2) increase the snow water equivalent and increase runoff during the spring melt period.

Peak Flows

The effect of forest disturbance on the size of peak flows can be predicted by the changes in the dominant controlling factors, which include:

1. the change in peak snowmelt rates;
2. the change in rainfall interception, particularly when the soils are relatively wet;
3. the degree to which roads and other disturbances intercept water and alter the pathway that water takes to the stream channel,
4. alteration of the infiltration rate to the extent that runoff pathways are changed, and
5. changes in soil moisture content and storage capacity (Anderson and others 1976).

With respect to the change in peak snowmelt rates, the magnitude of the effect of forest disturbance (other than fire) on peak discharges in the cold snow zone is similar to the observed changes in annual water yields. In the case of Fool Creek, peak flow increased by an average of about 20 percent (fig. 4); however, the three largest peaks of the post treatment period, from 1967 to 1998, were not significantly increased (Laurie Porth, personal communication). During those years when snow packs are greater and more long lasting, melt rates in the clearings appear similar to those in the forest and differences in soil moisture resulting from timber harvest are eliminated before the peak, thus diminishing the effects of forest removal on peak discharge for those largest events. Other studies in the cold snow zone have shown a 20 to 50 percent increase in the average peak flows due to clearcutting, but these have also shown no significant increase in the largest peaks of record (Troendle and Bevenger 1996; Troendle and King 1985; Troendle and others 2001).

A comparison of pre- and post-harvest flow duration curves indicates that the flows most affected by forest harvest are those that exceed the lowest 40 percent of the discharges but do not exceed the 90th percentile. However, the duration of bankfull discharge at Fool Creek, which is assumed to equal the 1.5-year instantaneous discharge, increased from 3.5 days to 7 days per year after the timber harvest (Troendle and Olsen 1994). The longer duration of bankfull flows was presumed to increase channel scour as indicated by the observed increases in annual sediment yields after patch clearcutting on the North Fork of Deadhorse Creek (Troendle and King 1987).

There has been considerably more debate over the effect of forest harvest on peak flows in the maritime snow climates of the Pacific Northwest and Continental/Maritime hydrologic regions (Grant and others 2008; Jones and Grant 1996; Thomas and Megahan 1998). This debate is due in part to the fact that the largest peak flows are typically due to mid-winter rain-on-snow events, and forest harvest can affect a series of processes that control the amount of snow in the canopy and on the ground, as well as the amount of heat that is available to melt the snowpack. These additional processes, when combined with the variability in climatic conditions during a storm and within a watershed, can make it very difficult to determine exactly how forest harvest affects peak flows from a given event.

Plot-scale studies have shown that rain-on-snow events accompanied by high winds can dramatically increase snowmelt rates in forest openings (Beaudry and Golding 1983; Berris and Harr 1987; Christner and Harr 1982; Harr 1986; Marks and others 1998; Storck and others 1998, 1999). This increase in melt rates is due to the increased condensation of moist air on the snowpack driven by the high winds and the resulting transfer of heat to the snowpack (Berris and Harr 1987; Harr 1986; Marks and others 1998; Storck and others 1999). Much less research has been done on how thinning affects this process, but basic research on turbulence theory suggests that even widely spaced cylinders (for example, trees) can be effective in reducing turbulence at the bottom surface (Poggi and others 2004a, b). This would suggest that thinning may have little effect on peak snowmelt rates during rain-on-snow events in the transient snow zone.

Forest roads, whether paved or unpaved, typically have very low infiltration rates and, therefore, convert nearly all of the rainfall or snowmelt into overland flow. When they are cut into the hillslopes, they also can intercept the slower moving subsurface flow. Depending on their connectivity with the stream network, roads may deliver this water directly to the channel. The increase in runoff and faster flow velocities act together to increase the size of peak flows (LaMarche and Lettenmaier 2001; Luce 2002; Megahan 1972; Wemple and Jones 2003; Wemple and others 1996) as well as total runoff. Skid trails can also generate surface runoff because of their lower infiltration rates, but generally these are not deeply incised to the hillslopes and, therefore, do not intercept subsurface stormflow. A compilation of published data indicates that the proportion of roads that are connected to the stream network is proportional to mean annual precipitation, as this tends to increase the number of streams and road crossings, which are a primary source of road-stream connectivity. The effect of roads on the size of peak flows can be minimized by outsloping and reducing their density and proximity to stream channels.

As a result of tracked or wheeled vehicles, soil compaction can reduce infiltration rates to the point that overland flow is generated during storm or snowmelt events. Again, this will increase the amount and velocity of runoff and thereby increase the size of peak flows. As with roads, best management practices are usually implemented during mechanical operations to reduce or eliminate this problem, including avoiding operations, such as minimizing high traffic areas during wet weather when the soils are more susceptible to compaction.

Other mechanisms that increase the size of peak flows in rain-dominated areas include the post-harvest increases in soil moisture and rainfall interception. Wetter soils allow a greater percentage of the precipitation to become streamflow, and the reduction in summer evapotranspiration generally results in wetter soils through the growing season. This would cause an increase in the runoff response in the first fall rainstorms. However, once the soil moisture in uncut areas has been fully recharged, there would be minimal differences in soil moisture between the cut (or thinned) and uncut areas and the initial soil moisture effect would be largely eliminated. Since the largest runoff events occur under wet conditions, the change in soil moisture due to timber harvest is unlikely to affect the size of the largest peak flows (Troendle 1987).

Forest harvest or forest thinning will also reduce the amount of rainfall interception by reducing the total leaf area. Any reduction in interception will effectively increase the amount of precipitation reaching the mineral soil, and this change should increase the size of peak flows. Again, this change will be most important in the smaller storms and less in the larger, more intense storms as the forest canopy can generally capture only a few millimeters of water and evaporation rates during large storms are relatively small due to the small amounts of incoming solar radiation and high relative humidity.

A recent review on the effects of forest harvest on peak flows in western Oregon supports these basic principles (Grant and others 2008). First, data from a variety of paired watershed experiments shows that forest harvest has a progressively smaller effect on peak flows as recurrence interval increases, and this is consistent with our general understanding of runoff processes and results from the Rocky Mountain region. The observed changes in the size of peak flows varied by watershed, but the peak flow increases ranged from 0 to 40 percent in the rain and transient snow zones and from 0 to 50 percent in the snow zone. The observed increases in peak flows generally approach the limit of detectability (about a 10 percent change) at a recurrence interval of approximately 6 years (Grant and others 2008). The largest increases in the size of peak flows occur in the fall because of the higher soil moisture carryover in harvested areas. The timing of the largest increases is consistent with equation 1 and the observed changes in the cold snow zone where the runoff increase occurs almost entirely on the rising limb of the snowmelt hydrograph because of the soil moisture carryover from the previous summer and corresponding reduction in the amount of water needed for soil moisture recharge in the following spring.

As in the cold snow zone, the magnitude of the observed changes in the size of peak flows in western Oregon is generally linear with respect to the proportion of the watershed that has been harvested (Grant and others 2008). The effect of roads cannot be clearly disentangled or quantified relative to the effect of timber harvest, although the data and modeling studies suggest that roads can increase the size of peak flows (for example, Bowling and others 2000; Grant and others 2008; Jones 2000). Using the mean response lines from different watershed studies, thinning less than 40 percent of a watershed is unlikely to cause a detectable change in the size of peak flows in rain-dominated areas and would result in only a 14 percent increase in the size of peak flows in the transient snow zone (Grant and others 2008).

In conclusion, both the available data and our understanding of hydrologic processes indicate that thinning should generally have little or no effect on the size of peak flows. In general, the changes in the size of peak flows due to forest management are small relative to the interannual variability in the size of the largest runoff events, and this again makes it difficult to link thinning with statistically significant hydrologic, geomorphic, or ecological changes.

Hydrologic Recovery

The longevity of hydrologic response following timber harvest appears to be unique in the cold snow zone of the Rocky Mountain region relative to other hydrologic regions, and this includes both the changes in water yield and snowpack accumulation. At Fool Creek the “average” 10-cm first-year increase in flow had declined by only 28 percent over the first 28 years following timber harvest (Troendle and King 1985). Recent streamflow data suggest that full hydrologic recovery will require 60 or more years (Laurie Porth, personal communication).

The duration of hydrologic recovery is more speculative when the silvicultural practice is a thinning or individual tree removal as compared to clearcutting strips, patches, or entire watersheds. It is generally assumed that the residual trees will very quickly occupy the site and use the soil moisture savings. However, the significant increases in snow pack accumulation persisted for at least 20 years after a partial cut that removed 40 percent of the basal area at Deadhorse Creek (Laurie Porth, personal communication; Troendle and King 1987). The persistent increase in the snowpack after partial cutting implies that at least a portion of the associated increase in annual water yields and peak flows would also be long lasting in the cold snow zone.

In contrast to the cold snow zone, the hydrologic response to clearcutting and thinning is relatively short-lived throughout the balance of the United States (Beasley and others 2004; Brown and others 1974; Douglass 1983; Harr 1983; Hornbeck and Kockenderfer 2004; Hornbeck and others 1997; Jackson and others 2004; Jones and Post 2004; Kattleman and Ice 2004; Keppeler and Zeimer 1990; Troendle and Leaf 1980). The shorter duration is due to the much more rapid rate of vegetative regrowth, which is due largely to the warmer temperatures and greater availability of water. Paired watershed studies suggest that the increase in annual water yields resulting from clearcutting will drop to zero within 30 years, and there may then be a period of a net decrease in water yields as a result of the active regrowth and changes in species composition (Jones and Post 2004). The persistence of any increase in annual water yields due to thinning or partial cuts will be much shorter due to the tendency for the residual vegetation to uptake any savings, and these are likely to disappear within 5 or possibly 10 years. When this occurs, any increase in low flows is also likely to be very short-lived.

Hydrologic Effects of Prescribed Fire

Prescribed burning is the controlled use of fire to achieve specific management objectives (Walstad and others 1990), and it is commonly used to reduce fuel buildup and the associated risk of severe wildfire (Norris 1990). Between 1998 and 2007, 6.7 million hectares managed by federal agencies in the United States were treated with prescribed fire (NIFC 2008). Relative to wildfires and forest harvest, the effects of prescribed burning have received little study until recently.

The hydrologic effects of prescribed burning are largely a function of fire severity and area burned. High severity burns that consume protective litter and expose mineral soil generally increase runoff and sediment yields, whereas low severity burns that only consume the upper litter layers have much less hydrologic impact (Benavides-Solorio and MacDonald 2001, 2005; Tiedemann and others 1979). Because prescribed fires are typically intentionally set during times when flame lengths are expected to be low, fire residence times are expected to be short, soil heating is expected to be low, and the effects of prescribed fires on soil properties are limited in severity and extent. The percent exposed mineral soil following low severity prescribed burns is generally between 5 and 30 percent, whereas values ranging from 35 to 95 percent have been reported following high severity prescribed burns or wildfires (Benavides-Solorio and MacDonald 2001, 2005; Cooper 1961; Robichaud and others 1993; Swift and others 1993; Robichaud and Waldrop 1994; Van Lear and Kapeluck 1989). The occurrence of surface runoff and erosion after fires is highly dependent on the amount of ground cover. Most studies indicate that little overland flow or surface erosion occurs when there is less than 35 to 40 percent bare mineral soil (Benavides-Solorio and MacDonald 2005; Robichaud and Brown 1999). This may be a useful first-order threshold for predicting whether there is likely to be a significant increase in surface runoff, but the hydrologic effects of fire depend on many other factors beyond burn severity and percent bare soil. These include the amount of vegetation that is killed by the fire, proportion of a watershed that is burned, location of the areas burned within a watershed, soil type, rate of vegetation recovery, and precipitation regime after burning (Luce 2005).

On-Site Effects

The surface condition after a prescribed fire is typically a mosaic-like pattern of low severity, high severity, and unburned patches (Robichaud 2000). The connectivity of runoff producing patches imparts a strong control on water and sediment yields to the stream channel (Doerr and Moody 2004; Luce 2005; Shakesby and others 2000). The patchiness of burn severity allows unburned and low severity patches to infiltrate runoff and trap sediment that is generated on adjacent high severity patches (Biswell and Schultz 1957; Cooper 1961; Swift and others 1993). The patterns of burn severity help control the spatial scale at which the effects of prescribed burning can be detected. For example, strong soil water repellency in high severity patches may have little effect at the watershed scale if only a small percentage of the watershed burns at high severity, or if there are intervening low severity or unburned patches (Huffman and others 2001).

Effects on Streamflow

Since low severity prescribed fires do not cause a high degree of tree mortality or litter combustion, the effects on evapotranspiration and forest floor water storage are generally too small to change watershed-scale water yields. For example, the 1 to 10 percent basal area mortality reported following low severity prescribed burns in ponderosa pine is below the 20 percent threshold at which changes in streamflow are usually detectable (Gottfried and DeBano 1990). The reduction in forest floor water storage due to prescribed burning varies, but the lower-most litter layer must be modified or removed before the water holding capacity of the forest floor is significantly reduced (Agee 1973; Brender and Cooper 1968; Clary and Ffolliot 1969; Cooper 1961). Therefore, prescribed fire is unlikely to increase watershed-scale runoff unless a large proportion of the watershed burns at high severity.

As evidence, water yields did not increase following a prescribed fire that burned 43 percent of an Arizona ponderosa pine watershed (Gottfried and DeBano 1990). The lack of a significant increase in flow was likely due to the fact that the fire killed only 1 percent of the pre-burn basal area and left most of the litter intact (Gottfried and DeBano 1990). Two successive prescribed fires that completely burned four loblolly pine watersheds in South Carolina had no detectable effect on streamflow (Douglass and Van Lear 1983). Similarly, prescribed fires in giant sequoia-incense cedar forests in Sequoia National Park in California had no effect on streamflow in a 100-ha watershed where 60 percent of the area was burned, and in a 20,000-ha watershed where eight fires burned 11 percent of the watershed over a 7-year period (Heard 2005). The absence of any change in water yield was attributed to the low severity burn in the 100-ha watershed and the small proportion that was burned in the 20,000-ha watershed. In contrast, a different prescribed fire in Sequoia National Park did cause streamflow to increase (Williams and Melack 1997). The fire was more severe than the low severity burn described by Heard (2005) and killed most of the younger trees and understory vegetation and consumed the majority of the forest litter (Williams and Melack 1997).

The effects of prescribed fire can vary by cover type. When the cover type is chaparral, the relative intensity of the burn may be greater, a greater percentage of the vegetation is consumed, and a greater percentage of the soils become water repellent. In two chaparral watersheds, burning 80 to 90 percent of the area by moderate and high severity fires increased water yields by 4 and 14 times, respectively, relative to unburned areas (Riggan and others 1994).

These results confirm that light to moderate prescribed fire has little effect on streamflow. This is largely because only a small percentage of the vegetation is affected and net changes in infiltration characteristics are minimal. Since the major components of the water balance are not substantially altered, there is little or no effect on streamflow.

In some vegetation types, particularly chaparral, there is a much greater propensity for prescribed fires to burn at higher severity. In these areas, the use of prescribed fire as a fuel reduction treatment may have a greater hydrologic effect. In each case, the integrated hydrologic response to successive prescribed fires must be compared to the hydrologic response resulting from the likely frequency and severity of a wildfire.

Predicting Changes in Water Yield

The 1980 WRENSS Handbook includes a set of graphical procedures that have proven useful for estimating the hydrologic impacts of various silvicultural activities on water yield and water quality (EPA 1980). The hydrology chapter has regional evapotranspiration estimates based on the hydrologic regions in figure 3. Regionalized curves and modifier functions are then provided to estimate the changes in actual evapotranspiration in response to changes in stand density and stand condition (Troendle and Leaf 1980). The predicted changes in evapotranspiration were assumed to affect the amount of water available for stream flow. For snowmelt-dominated areas, the changes in forest cover alter net precipitation and the amount of evapotranspiration. In rain-dominated areas, the precipitation is not adjusted to reflect stand conditions and the change in evapotranspiration is estimated directly.

The understanding of hydrologic processes in the cold snow zone has evolved significantly since WRENSS was developed, and a new version of the model, WinWrnsHyd, has been produced (Swanson 2004). One of the most significant changes is how this program simulates the effects of forest harvest on snow accumulation. The revised snowpack sublimation and scour routines are more sensitive to wind speed, surface roughness, and opening size. These changes allow one to better link the changes in wind speeds due to removing some or the entire forest canopy to changes in snow accumulation, and to more accurately predict the effects of leaving or removing slash or other forms of roughness on snowpack accumulation. The net effect is to make the model more sensitive and more accurate with respect to the effects of partial cuts and thinning on water yields (Shepperd and others 1992).

WinWrnsHyd is programmed in Microsoft Access and uses database tables as input so that different harvesting scenarios can be created using GIS or other forest planning tools. Data reflecting stand conditions can be input as a series of "snapshots." Alternatively, if growth curves are available, the data for one or more silvicultural prescriptions, occurring simultaneously or at different time intervals, can be input to the model and the effects of regrowth on hydrologic response can be simulated as a time series. The WinWrnsHyd program can also estimate the likely changes in peak flows following forest disturbance. These procedures and updates are particularly relevant because, as noted earlier, the cold snow zone is of tremendous importance for water supply purposes, and fuel reduction treatments in this zone are more likely to affect runoff for a longer period than similar treatments in other ecoregions.

Cumulative Watershed Effects

The concern over cumulative effects arises because the effect of a single activity may not be significant, but the effect may be significant when combined with the effect of other management activities. A cumulative effect can occur spatially, such as the effect of multiple management activities within a basin, or over time, such as the hydrologic effect of one activity persists and the residual effect is superimposed on the effect of a second activity on the same site (MacDonald 2000).

The potential for generating a cumulative effect in space depends on the magnitude of each effect, their persistence over time, and the extent that the effect is delivered to the downstream location. In the case of fuel management activities, the hydrologic effect of a given activity is likely to be relatively small because only some of the forest canopy is being removed. As noted earlier, it has been generally accepted that at least 20 percent of the basal area in a forested watershed must be removed to obtain a detectable change in stream flow. As watershed size increases, it is increasingly unlikely that forest management will affect more than 20 percent of the basal area in a watershed before hydrologic recovery eliminates the hydrologic change due to some of the management actions. The implication is that the hydrologic effect of a fuel management activity is most likely to be detectable immediately below the activity, and the rate of hydrologic recovery will make it difficult to detect the effect of multiple activities over time and space, especially in larger watersheds (MacDonald 2000). As an example, clearcutting 36 percent of the North Fork of Deadhorse Creek sub-basin in the Fraser Experimental Forest in central Colorado caused a significant increase in streamflow. However, this change was not detectable a few hundred meters downstream at the main stream gauge on Deadhorse Creek, as the harvest in the North Fork watershed affected less than 6 percent of the area in the Deadhorse Creek watershed (Troendle and King 1987).

The potential for cumulative watershed effects due to fuel management will also be limited because most of the fuel management activities in the western United States will be concentrated in the drier forest types that have the greatest risk of high severity wildfires. As noted earlier, most studies have shown that forest harvest will not result in a detectable change in streamflow when mean annual precipitation is less than 18 to 20 inches. In contrast, thinning has been demonstrated to cause moderate increases in streamflow in the central Appalachians where precipitation greatly exceeds 18 to 20 inches (Reinhart and others 1964). But in humid areas, the hydrologic recovery is quite rapid (Hornbeck and others 1997). In general, the absolute changes in runoff due to fuel reduction activities in the drier forest types will be small or undetectable relative to the potential changes in more humid areas. The potential for cumulative hydrologic effects is further limited because the persistence of a hydrologic change due to thinning or a partial cut will generally be relatively short everywhere except in the cold snow zone, but these forest types are less likely to be the focus of fuel reduction treatments (Romme and others 2003).

Another issue in assessing the potential cumulative hydrologic effect is whether a given change in flow will be transmitted downstream to the location of interest. In most mountainous areas, any change in flow generated by forest harvest or fuel management activities should not be substantially altered by downstream transmission losses. However, seepage losses may become significant when streams and rivers flow onto broad, semi-arid alluvial plains. In such areas, the streams are likely to be losing water to the underlying alluvial aquifer during at least the drier portions of the year. In other words, the increase in streamflow may be “lost” to groundwater storage. The potential for transmission losses will be a function of the scale of the analysis, relative and absolute magnitude of the changes in flow, and specific watershed characteristics.

In most cases, the measurement and detection issues mean that the magnitude of a potential cumulative hydrologic effect will have to be assessed by modeling. As an example, Troendle and Nankervis (2000) and Troendle and others (2003) estimated the changes in average annual water yield resulting from long-term vegetation changes in the North Platte River Basin (table 1). Current vegetative conditions in the river basin were extrapolated backwards in time from United States Forest Service stand condition records, and water yields under the different forest conditions from 1860 to 2000 were simulated using the WRENSS model. The results suggested that streamflow from National Forest System lands has decreased 3 inches (76 mm) from 1860 to 2000 as the result of an increase in forest density (table 1). Although these decreases are difficult to detect using the existing streamflow records on the North Platte River, they are considered real and significant by water users and planners in the Platte River Basin. A separate study using a combination of precipitation, snowpack, and streamflow records reached similar conclusions (Leaf 1999).

Table 1. Water yield from National Forest land in the North Platte River basin from 1860 to 2000.

Year	Area (ha)	Predicted water yield (mm)
1860	448,418	376
1880	448,418	343
1900	448,418	366
1920	448,418	340
1940	448,418	307
1960	448,418	302
1980	448,418	307
2000	448,418	300

In a more recent assessment, Troendle and others (2007) used the WRENSS Hydrologic Model to predict the changes in water yield resulting from proposed fuel management treatments in the Upper Feather River watershed in the Sierra Nevada of California. Because proposed treatments influenced only a small percentage of the total vegetation on the entire study area, the cumulative impact on water yield was minimal. However, the GIS-based modeling was useful for demonstrating that the treatments could have an on-site or local affect on annual water yields.

Conclusions

One can conclude that fuel reduction treatments in forested watersheds will probably have little detectable impact on water yields either on-site or downstream. Most prescriptions are not likely to remove the 20 percent of basal area that is needed in most areas to generate a detectable change in flow. As Bosch and Hewlet (1982) concluded and subsequent data (Hornbeck and others 1997) and modeling (Troendle and others 2003, 2007) support, removing less than 20 percent of the basal area may also result in a change in flow, but this change will not be detectable. In cases where there is a detectable hydrologic response to fuel management treatments, the observed response will be greatest in wet years and smallest or non-detectable in dry years. Fuel reduction treatments that are carefully implemented and do not induce overland flow as a result of skid trails or compaction should generally have little or no detectable effect on peak discharges. With the exception of the cold snow zone in the Rocky Mountain region, any change in flow due to fuel reduction treatments will be short-lived.

Prescribed fires, when designed and used as a fuel reduction tool, are probably less likely to influence water yield than mechanical treatments because of the smaller reduction in basal area and lack of ground disturbance by heavy machinery. Prescribed fires that kill a significant proportion of the mature canopy or expose more than 35 to 50 percent of the mineral soil may have a significant, detectable effect on annual water yields or storm runoff.

Simple models are available to simulate the on-site and cumulative hydrologic impacts of virtually any individual or combination of forest disturbance scenarios. The use of these models should be a required component of the planning process in order to assess both on-site and cumulative impacts over time.

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Chapter 8.

Effects of Fuel Management Practices on Water Quality

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Introduction

Fuel management practices in the Rocky Mountain region may include prescribed fire, timber harvesting (patch cuts, thinning, high-grading, or selective logging), mechanical treatments (mulching, chipping or chunking), chemical treatments, or grazing to reduce undesirable species (Chapter 4). The application of any of these treatments has the potential to affect water quality. Understanding the effects of land use practices on hydrologic processes is of primary importance when assessing water quality effects. Unlike agriculture where there are often many activities each year, fuel managements practices occur once every year to once over several decades. Fuel management activities should be implemented with best management practices (BMPs) to minimize or prevent water quality changes or nonpoint source pollution.

Fire

Research has largely focused on the effects of wildfire on water quality. Few address prescribed or controlled fires as smaller watershed level effects are expected. In general, wildfires are more intense (EPA 2005) and more extensive in area than prescribed fires, resulting in potentially greater effects on watershed processes. Watershed effects from fire depend on several variables, including fire size, fire severity, soils, watershed slope, vegetation, vegetation regrowth, precipitation, physical location on the watershed, and proportion of watershed burned.

Temperature

Soil heating may occur following the removal of cover (vegetation, litter and duff, and organic material) by fire (Wells and others 1979). The magnitude of heat pulse into the soil depends on fuel loading, fuel moisture content, fuel distribution, rate of combustion, soil texture, and soil moisture content (Chapter 9). The movement of heat into the soil is not only dependent upon the maximum temperature reached, but the length of time and the heat source that is present. Because fuel is not evenly distributed, a mosaic of heating occurs. The highest soil temperatures are associated with areas of greatest fuel consumption and longest duration of burning. In forested areas, high subsurface soil temperatures usually occur beneath fuel accumulations, with the highest temperatures most likely found in association with consumption of large piles of harvest residue or windrow or very thick duff layers.

Rangelands have lighter fuel loadings, resulting in fires of shorter duration and less subsurface heating. The greatest subsurface heating likely occurs where thick, dry litter layers are consumed beneath shrubs and isolated trees. The soil heat pulse, including both amount and duration, is instrumental in eventual effects of fire on plants (DeBano and others 1998). Excessive soil heating can kill plants and decrease vegetative cover and influence stream temperature from loss of riparian cover to soil water heating.

Exposure of small streams to direct solar radiation is the dominant process responsible for stream temperature increases (Tiedemann and others 1978). Other mechanisms include increased air temperature, channel widening, soil water temperature increases, and streamflow modification (Ice 1999). Streams with smaller surface areas may be more susceptible to heating, but usually return to expected temperature within 500 ft (150 m) downstream (Andrus and Froehlich 1991). Maintaining shade in riparian zones can be used to avoid most temperature increases in small streams. As stream width increases, more of the water surface is exposed to sunlight, consequently reducing the influence of riparian canopy on stream temperature.

The ability of a forest fire to change the temperature of any particular watercourse or water body depends on the amount of water subject to heating. More precisely, it depends on the affected unit's surface-area-to-volume ratio. In essence, this means that temperatures rise faster in smaller and shallower water bodies than in larger and deeper ones. All else equal, the magnitude of any temperature change depends on both the amount of heat directed at the water surface per unit time and the duration of heating. As fire burns in surrounding vegetation and woody debris, it can raise the temperature of water in forest streams (Amaranthus and others 1989; Cushing and Olson 1963; Feller 1981; Hall and Lantz 1969; Helvey 1972; Levno and Rothacher 1969; Spencer and Hauer 1991; Swift and Messer 1971).

The best management practices for prescribed fire are to schedule burning when the soil moisture conditions will minimize heat conductivity into the soils. Streamside management zones or buffers along stream channels can provide shade for stream temperatures and provide filter strips for sediment and nutrients as described later. Streamside buffers are often difficult to exclude from a prescribed burn, but the soil and vegetation are usually moist and do not burn.

Sediment

Watershed responses to prescribed fire may include changes in runoff characteristics, sediment yield, and water chemistry. Under pre-fire conditions, grasses, brush, and the forest canopy intercept precipitation and release it as throughfall, supporting infiltration. Infiltration reduces direct overland flow from precipitation. Runoff is generated through the variable source area concept where infiltration exceeds the saturation potential of soils. As the erosive potential of overland flow is minimized, nutrients and sediments are retained on site. In the absence of vegetative cover, runoff becomes flashier as more streamflow is generated by overland flow, resulting in sharper, higher peak flows and often lower baseflows. With less infiltration, vegetative uptake and retention of water, total water yields from burned watersheds are higher. Once runoff begins, loose soils and ash are quickly removed from steeper slopes. Fire-associated debris is swiftly delivered directly to streams in large quantities. The first storm of the year may produce a 'rolling black' that is a storm event high in suspended sediment and ash. Suspended concentrations over 40,000 mg/L were measured in the first storm event after the 2000 Bobcat fire in Colorado (Kunze and Stednick 2005).

Organic compounds in litter, probably aliphatic hydrocarbons, are volatilized during combustion, migrate into the soil profile, and condense on soil particles, forming a water repellant layer (DeBano and others 1998). The phenomenon is more evident in dry, coarse textured (sandy) soils. It also appears that high temperatures, above 550 °F, destroy the compounds responsible for water repellency. These data suggest fires that heat soils to an intermediate range of temperature (400 to 500 °F) are more likely to cause the formation of a non-wettable layer than fires that heat only the soil surface or those that cause deep penetration of high temperatures. In addition, certain plant communities, such as those containing chaparral species, are more likely to be affected. It is important to recognize that hydrophobicity occurs naturally (DeBano 1981) and may develop under prescribed fire conditions (Huffman and others 2001); however, the effect is not long-lived. Repeated measurements of hydrophobicity after fire suggested the phenomenon lasted up to 22 months in forest soils of the Colorado Front Range, but is usually gone after less than 1 year (Huffman and others 2001).

Suspended sediment is the major nonpoint-source pollution problem in forests, most often associated with forest roads (MacDonald and Stednick 2003). Sediment and turbidity are the most significant water quality responses associated with fire (Beschta 1990). Erosion resulting from prescribed burning is generally less than that resulting from roads, skid trails, and site preparation techniques that cause soil disturbance, which are often a necessary component of prescribed burn projects (EPA 2005).

A controlled burn is usually designed to modify a vegetation type (Chapter 3), while uncontrolled wildfires are less selective in modifying vegetation type or age class. Erosion rates following fires may increase from decreased vegetative cover and/or modified soil properties, including decreased infiltration, hydrophobicity and movement of ash or debris and increased rill erosion from hillslopes directly to the stream channel. Soil erosion may cause decreases in soil nutrients, but unless soil erosion rates are excessive, more nutrients are usually “lost” through the consumption of vegetative fuel. Actual soil erosion and nutrient loss varies by site as a function of vegetation type and recovery, soil type, fire severity, topography, slope position in relation to surface waters, and climate. Significant climate modification has been linked to large area fires. For example, the Bobcat Fire was severe and subsequent storms that occurred had low recurrence intervals, resulting in higher frequency peakflows and higher soil erosion rates (Kunze and Stednick 2005).

Burned areas are sometimes seeded to rapidly establish plants or are given physical treatments to quickly stabilize the soil (Moench and Fusaro 2004). Following severe wildfire, the Forest Service and other land managers may implement Burn Area Emergency Rehabilitation (BAER) treatments to reduce the risk of high runoff and sediment flows. The effectiveness of the most widely used BAER practice, contour-felled log barriers, has not been systematically studied (Robichaud and others 2000). The second most used BAER practice, postfire broadcast seeding with grasses, has been studied and the majority of studies found that this treatment did not significantly reduce erosion during the critical first 2 years after fire (Robichaud and others 2000). Research on the effectiveness of other watershed restoration treatments is ongoing.

Reseeding with grasses is not a reliable technique for erosion control after severe wildfire. Additionally, when an area is seeded with nonnative grass species, native plant species may be effectively excluded leading to questions about long-term stability. Firelines, particularly those that are created by bulldozers, are potential areas of increased soil erosion and establishment of non-native plants. Firelines may be difficult to stabilize with vegetation because much of the nutrient-rich surface soil is cast aside. Hence, they are likely to be slow to revegetate with perennial vegetation. Application of native seed and fertilizer is an effective way to protect firelines (Klock and others 1975; Tiedemann and others 1979).

Nutrients

There are regional differences in the effects of fire on water quality. Of the few studies available for the southeastern United States, results have shown either no effect or small increases in stream nutrients following fires (Richter and others 1982). This contrasts with regions in the western United States where fires have a notably larger effect on water quality (Gresswell 1999; Neary and others 2005; Spencer and others 2003; Stednick 2000). Dissolved nutrients in streamflow are derived primarily from weathering, decomposition of plant material, and anthropogenic sources. Vegetative communities accumulate and cycle large quantities of nutrients (Tiedemann and others 1979). Fire can disrupt this cycle and cause nutrient leaching, volatilization, and transformation.

The concentrations of inorganic ions often increase in streams after a fire (DeBano and others 1998). Studies indicate that changes in chemistry and flow conditions after forest fires are temporary, usually lasting less than 5 years (Chorover and others 1994; Covington and Wallace 1992; Fredriksen 1971; Hauer and Spencer 1998; Ice and others 2004). Early reestablishment of vegetative ground cover after a wildfire is an important factor controlling the recovery.

Water from forested watersheds is typically lower in nutrients than water draining from other land uses. Forest management activities, such as forest cutting and harvesting, may increase annual water yields (Bosch and Hewlett 1982; Stednick 1996) and disrupt the natural cycling of nutrients (Stednick 2000). Several chemical constituents are likely to increase after forest and rangeland burning. The primary constituents of concern are nitrate (NO_3^-), phosphate (PO_4^{3-}), calcium (Ca^{2+}), magnesium (Mg^{2+}) and potassium (K^+). Nitrate is a mobile ion and easily leached from burned areas. Stream nitrate responses to prescribed fire are generally lower than for wildfire (Stednick 2000). Conversely, phosphorus binds readily to sediment and is thus predominately transported with soil erosion. The bulk of phosphorus transport is as total phosphorus, and orthophosphate concentrations are low (Stednick and others 1982). Changes in concentrations of sulfate, pH, total dissolved solids, chloride, iron, and other constituents have been measured. If organic compounds leach into surface waters, water color, taste, and smell may be affected.

Nitrate-nitrogen (NO_3-N) concentrations are usually quite low (0.002 to 1.0 mg/L) in streams draining undisturbed forest watersheds (Binkley and Brown 1993a,b). Concentrations are low because nitrogen is rapidly used by ecosystem biota, and nitrate formation (nitrification) is relatively slow in forest soils. Slow rates of organic matter decomposition, acid soil conditions common in forest environments, and bacterial allelopathy all decrease rates of nitrification. Organic matter and anaerobic conditions in saturated riparian soils allow for denitrification, the reduction of nitrate to nitrogen gas, which may be lost to the atmosphere.

Often, fires will create soil environmental conditions that are favorable for increased microbial activity (Ballard 2000). These include near neutral pH, increased soil moisture (because there are no interception or evapotranspirational losses), a food or carbon source, and soil temperatures. The increased microbial activity often results in a short-term increase in nitrogen availability. Depending on the monitoring frequency and site specifics, an ammonium pulse may be seen, but usually a pulse of nitrate is measured. The short increase in nitrogen availability helps new or existing vegetation become established. Increased nitrogen mineralization rates persisted for 1 year in range grassland and up to 2 years in a shrub community (Hobbs and Schimel 1984).

If vegetation is quickly reestablished, nutrient exports are short-lived and usually do not represent a threat to water quality or site productivity. There are a couple of possible exceptions. Nitrogen deposition can accumulate in forest soils over time, especially in areas with air quality concerns (Riggan and others 1985; Silsbee and Larson 1982). If timber harvesting occurs in these areas, mobilization of accumulated soil nitrogen may result in higher nitrate concentrations and outputs in the streamwater. Values for nitrate generally increased after fire but not to a level of concern, except in nitrogen-saturated areas. Nitrogen-saturated areas are where the atmospheric inputs of nitrogen compounds from precipitation and dryfall exceed the plant uptake requirement, and thus, excess nitrogen moves through the system. The most striking response of nitrate concentration in streamflow after wildfire was observed east of Los Angeles in southern California (Riggan and others 1994).

Immediately after a fire, stream pH may be affected by direct ash deposition as oxides form from the volatilization of metallic cations. In the first year after fire, increased soil pH may also contribute to increased streamwater pH (Wells and others 1979). In most studies, pH values were little changed by fire and fire-associated events (Landsberg and Tiedemann 2000). Transient pH values up to 9.5 were measured 8 months after the Entiat fires in eastern Washington (Tiedemann 1973, 1981).

Measures that reduce on-site soil erosion and stream vegetative buffers, such as riparian areas, will minimize effects of fire on water quality.

Timber Harvesting

Timber harvest, whether marketable or not, is often used as a tool in fuel management (Chapter 4). The effects of timber harvesting on water quantity and quality are well known. Most water quality studies are conducted at small watershed levels in order to decipher treatment effects from variability in water quality data. The effects of timber harvesting as a thinning, selective cut, or other partial canopy removal treatment, will have less of an effect than complete canopy removal. Less site disturbance will result in less erosion potential and remaining vegetation will quickly utilize increased available nutrients and water from evapotranspiration savings.

Temperature

Surprisingly, few recent studies have been published on the effects of silvicultural practices on water temperature, and most of these were conducted in the 1970s. These studies include harvesting with and without streamside vegetation buffers (Beschta and others 1987; Binkley and Brown 1993a; Swank and Johnson 1994).

Literature on the effects of timber harvesting on stream temperatures shows daily maximum stream temperature increases from 1.5 to 8 °C in eastern forests and 0.6 to 10 °C in western forests. The range in temperature increases reflects a range in streamside vegetation buffers from no buffer to a 100-m buffer. Changes in minimum nighttime stream temperatures (during the winter or dormant season) range from no change to less than 1 °C in the East and from zero to less than 2 °C in the West (Stednick 2000).

Temperatures in small streams may increase when the streamside vegetation canopy is removed. Providing streamside buffers or management zones can mitigate this effect. Several studies have reported temperature increases with streamside buffers, but the increases are much smaller than those of fully exposed streams. The lack of documentation on buffer characteristics makes extrapolation difficult. Different measurements of stream temperature also make direct comparisons difficult. Attributes needed to estimate the contribution of forest overstory to stream surface shade include stream width, distance from vegetation to stream, stream orientation, height and density of vegetation, crown or canopy measurement, latitude, date, and time (Quigley 1981).

Generally, forest practices that open small stream channels to direct solar radiation increase stream temperatures. Retention of streamside vegetation appears to mitigate potential temperature changes, especially temperature extremes. These principles are well documented by research throughout the country. Streamside canopy removal may also decrease winter stream water temperatures, since radiation losses may be increased. For small streams, temperature returns to expected levels within a short distance downstream of where canopy shade is reestablished (Andrus and Froehlich 1991). In general, removal of streamside vegetation cover has the potential to increase streamwater temperatures during the day in the summer. In certain settings, the vegetation removal may allow for decreased nighttime temperatures, especially in the winter. Temperature changes return to pretreatment levels as the streamside vegetation reestablishes. The maintainence of streamside vegetation as a thermal cover is key to maintaining stream temperatures at existing levels.

Sediment

Fuel management practices that result in soil disturbances may increase soil erosion. Soil erosion is the detachment and movement of soil particles, measured as tons/acre/year (Mg/ha/yr). Suspended sediment is eroded soil material transported in the water column of a stream. It is measured as a concentration such as mg/L or as turbidity, an optical measurement of the water's ability to diffract light expressed as Nephelometric Turbidity Units (NTU) (Stednick 1991).

Site properties that affect erosional processes include vegetative cover, soil texture, soil moisture, and slope (Falletti 1977; Renfro 1975). The sediment load of streams (both suspended and bedload) is determined by characteristics of the drainage basin such as geology, vegetation, precipitation, topography, and land use. Sediment enters the stream system through erosional processes, often as pulse events during storms. To achieve stream stability, an equilibrium must be maintained between sediment entering the stream and sediment transported through the channel, thus resulting in a stream profile that neither aggrades or degrades over time. A land use activity that significantly changes sediment load can upset this balance and result in physical and biological changes to the stream system (State of Idaho 1987).

Undisturbed forest watersheds usually have erosion rates from near 0 to 0.25 tons/acre/year (0.57 Mg/ha/yr) (Binkley and Brown 1993a). Erosion rates have been estimated as less than 0.1 tons/acre/year (0.2 Mg/ha/yr) for three-quarters of eastern and interior western forests (Patric and others 1984). Typical timber harvesting and road construction activities may increase erosion rates to 0.05 to 0.25 tons/acre/year (0.11 to 0.57 Mg/ha/yr). More intensive site preparation treatments, such as slash windrowing, stump shearing, or roller chopping, may increase soil erosion rates by up to 5 tons/acre/year (11.2 Mg/ha/yr). Soil erosion from a single precipitation event from a wildfire burned watershed was 0.42 tons/ac (0.95 Mg/ha) and accounted for 90 percent of the estimated annual erosion (Kunze and Stednick 2005). Erosion from unpaved road and trail surfaces may be higher.

Numerous studies have been done on the effects of different forest management practices on erosion rates or sediment production. In general, increased site disturbance will result in increased soil erosion and subsequent sediment production. The type and magnitude of erosion depend on the amount of soil exposed by management practices, the kind of soil, steepness of the slope, weather conditions, and any treatments after the disturbance (Swank and others 1989).

Logs are moved (skidded) from the stump to a landing by tractor, cable, aerial systems, or animals. Tractor skidders may be either crawler or wheeled units, both of which are frequently equipped with arches for reducing the extent of contact between log and ground. Site disturbance will vary greatly with the type of skidding or yarding system. Crawler tractors generally cause the greatest amount of site disturbance, followed closely by wheeled skidders. On some sites, use of wheeled skidders can result in more compaction than crawler tractors. One method of decreasing the amount of soil disturbed by crawler tractors or wheeled skidders is through careful layout of skid trails. Location of skid roads away from the stream channel and off steep slopes can greatly decrease the impact of tractor logging. Logging slash placement on used skid trails increases surface roughness and may decrease soil and water runoff. Cable logging systems will result in less site disturbance because yarding trails are established to the yarding tower machinery, which is restricted to road surfaces. Cable systems can be ranked in order of decreasing soil disturbance as follows: single drum jammer, high lead cable, skyline, and balloon (Stone 1973). Helicopters and balloons will likely result in minimum site disturbance, but both are costly and subject to operational constraints.

Unlike many other land uses that disturb soil for long periods, any increase in sediment yields from timber management activities is usually short-lived. Surface soil disturbances provide a sediment supply, but once the finer materials are transported and revegetation occurs, the site is less apt to continue eroding. Sediment yields or measured suspended sediment concentrations decrease over time as a negative exponential (Beschta 1978; Leaf 1974; NCASI 1999). This time factor should be considered when assessing watersheds for effects on water quality (Stednick 1987).

Most timbering operations will involve the use of forest roads for site access and removal of wood products. Roads are recognized as a potential source of erosion and sediment. BMPs related to roads include road location, road design, time of use, road construction and maintenance, and road obliteration. Roads are addressed by Luce and Reiman in Chapter 12.

Streamside vegetation or filter strips have been used to prevent overland flow and soil erosion from reaching surface waters. The filter strip, or equivalent, decreases the velocity of the overland flow by creating surface roughness. The decreased velocity allows sediment to settle out and overland flows to infiltrate into the undisturbed soils. The streamside vegetation filters were originally used to control or limit road-derived sediment from reaching forest streams. The filter had a recommended width of 10 to 100 m and was dependent on hill slope. These filter strips are effective in sediment removal unless an extreme precipitation or overland flow event exceeds the sediment detention/retention capacity. The characteristics that determine filter strip efficiency include width, vegetative and litter cover, surface roughness, and microtopography.

Fuel management by forest thinning is a relatively new practice and few studies have been conducted to assess the influence of these practices on water quantity and quality. A recent study in New Mexico on thinning in pinyon-juniper forests showed that water yield increased more on slash piled plots than scattered plots, when compared to a control. Similarly, sediment yields were higher on the slash piled plots than scattered plots. When slash was scattered, erosion was lower than the control plots (Madrid 2005).

Nutrients

Cutting vegetation disrupts the nutrient cycle and may accelerate dissolved nutrient leaching and loss via streamflow. Exposing sites to direct sunlight may increase the rate of nitrogen mineralization. Phosphorus is commonly associated with eroded soil particles and sediment and may be lost from the site (Swank and others 1989). Usually, there is minimal opportunity for a buildup of these nutrients in the stream system after a timber harvest because of the normally brief period of increased nutrient flux to the stream (Currier 1980). Throughout the United States, studies have found that nutrient losses from silvicultural activities are minimal and water quality (in terms of nutrients) was not affected (Aubertin and Patric 1972; Chamberlain and others 1991; Hornbeck and Federer 1975; Reuss and others 1997; Sopper 1975; Stednick 2000).

Catchment studies have produced a large body of information on streamwater nutrient responses, particularly from clearcutting. Changes in streamwater nutrient concentrations vary substantially among localities, even within a physiographic region. In central and southern Appalachian forests, nitrate-nitrogen (NO_3-N), potassium (K^+), and other constituents increased after harvesting, but the changes were small and did not affect downstream uses (Swank and others 1989). Clearcutting in northern hardwood forests may result in large increases in concentrations of some nutrients (Hornbeck and others 1987). Research on catchments has identified some of the reasons for varied ecosystem response to disturbance (Swank and Johnson 1994).

In general, nutrient mobility from disturbed forests follows the order: nitrogen > potassium > calcium and magnesium > phosphorus (Stednick 2000). Thus, forest harvesting or other disturbances, such as fire, generally produce larger differences in nitrogen concentrations in streamwater than in other constituents. Possible exceptions are the loss of calcium and potassium documented in the northeast United States when precipitation inputs had greater acidity from fossil fuel combustion (Federer and others 1989). Phosphorus is often associated with sediments and increased sediment inputs to the stream may increase phosphorus concentrations.

If vegetation is reestablished quickly, nutrient exports are short-lived and do not represent a threat to water quality or site productivity. Minimization of site disturbance areas will reduce potential soil erosion and allow for quick vegetation establishment. Use of streamside vegetation zones or buffers are effective in removing sediment from upslope overland flows and nutrients from surface and subsurface flows.

Fertilization

As noted earlier, there are some instances where site restoration or revegetation may require fertilization. The most common fertilizer used in wildland management is nitrogen, usually in the form of urea. Urea fertilizer is highly soluble in water and readily moves into the forest floor and soil with any appreciable amount of precipitation. Under normal conditions, urea is rapidly hydrolyzed (4 to 7 days) to the ammonium ion (NH_4). When moisture is limited, urea may be slowly hydrolyzed on the forest floor. Fertilizer is usually applied in the spring or fall to take advantage of seasonal low intensity and short duration precipitation events (Stednick 2000). If the fertilizer stays dry, the soil surface pH favors formation of ammonia (NH_3), which is lost by volatilization. These losses may be significant, and ammonia absorption by surface water is minimal (USDA Forest Service 1980).

The reported effects of forest fertilization on water quality, particularly nutrient concentrations in streams are variable (reviews by Binkley and Brown 1993b; Binkley and others 1999; Fredriksen and others 1975). Nutrient retention by forest soils is excellent and nutrient concentrations in surface waters after forest fertilization are usually low. Fertilizers may enter surface water by several routes. Direct application of chemicals to exposed surface water is the most significant. Identification of surface water bodies prior to the application essentially eliminates this entry mode.

The effects of forest fertilization on water quality, particularly nutrient concentrations in streams are variable (reviews by Binkley and Brown 1993b; Binkley and others 1999; Fredriksen and others 1975). Nutrient retention by forest soils is excellent and nutrient concentrations in surface waters after forest fertilization are usually low. Fertilizers may enter surface water by direct application of chemicals to exposed surface waters. Identification of surface water bodies prior to the application essentially eliminates this entry mode. Any ammonium concentrations in surface waters are rapidly reduced through aquatic organism uptake and stream sediment sorption. Streamside vegetation zones that are not fertilized are generally protective of surface waters.

Nitrate concentrations, if measured in surface waters, usually peak 2 to 4 days after fertilizer application (USDA Forest Service 1980). The magnitude of the peak concentration may depend on the presence and width of streamside buffers and the density of smaller tributaries to the streams. Peak nitrate-nitrogen concentrations usually decrease rapidly, but may remain above pretreatment levels for 6 to 8 weeks. Winter storms may also result in peak nitrate-nitrogen concentrations, but these peaks usually decrease over successive storms and concentrations decrease quickly between storms (Stednick 2000).

Careful delineation of application areas will avoid direct stream inputs of fertilizer. Fertilizer application should be timed to avoid high precipitation periods as fertilizer might be moved directly to surface waters. When fertilizer is properly applied at a rate and time when vegetation can benefit, fertilizers do not adversely affect surface waters. Streamside vegetation is an effective nutrient removal system and any increase in nutrient concentrations in surface waters from fertilizer applications is usually short-lived.

Mechanical Treatments

When vegetation is too thin for prescribed fire, logging is not economical, or fire is not acceptable, mechanical treatments can be effective in fuel management (Chapter 4). Properly used, mechanical treatments reduce fire hazards, increase plant diversity, control noxious weeds, and improve the quality and quantity of vegetation for wildlife and livestock (Zachman 2003). Treatment increases ground cover, which often results in increased infiltration rates and decreased surface runoff and soil erosion.

Roller chopping is a mechanical treatment that is frequently applied to mountain shrub types and pinyon-juniper stands with stem diameters up to 20 cm. The method is effective for knocking down brush and trees and chopping up the slash. Roller chopping can be done when the soil is firm and dry enough to support the heavy equipment.

Low-pressure tires or tracked vehicles can be used on soils that may be subject to compaction.

A cylindrical roller or drum, equipped with several full-length blades, is towed behind a crawler-type tractor or "cat." The roller chopper may be pulled straight or at a diagonal to increase the chopping action. Two roller choppers are sometimes towed in tandem and at slightly contrasting angles. The cat will usually have its blade positioned low to the ground to push over trees and brush. The heavy weight of the roller chopper crushes the trees and brush, while the blades chop them and help roughen the ground surface (Zachman 2003). The increase of litter and the increased soil surface roughness will increase infiltration and decrease soil erosion.

The use of a Hydro-axe is a mechanical treatment that is frequently applied to mountain shrub types and pinyon-juniper stands. This method is effective for knocking down brush and trees and chopping up the slash. A Hydro-axe, also known as a Hydro-mower, is an articulated tractor with a mower-mulcher mounted on the front of the machine. The Hydro-axe has rubber flotation-type tires that cause little disturbance to the surface of the ground. The machine can move around trees to treat selected areas (Zachman 2003).

The vegetation/soil litter following this treatment is much finer than that resulting from other mechanical treatments. The Hydro-axe allows the operator to be precise in the areas and vegetation treated. The mulch creates a protective vegetal layer for the rubber tire tractor to travel over, thus reducing surface disturbance. Large safety zones are required when using the machine since materials of varying size are frequently thrown from the machine

Depending on the fuel load, other site conditions, and the effectiveness of the chipper or mulcher, woody material is reduced to an organic layer of various thicknesses. Some of these organic horizons have been observed to be up to 30 cm in depth. Any increase in the organic horizon will reduce overland flow potential and hence erosion, but the deep layer may decrease soil temperatures and decrease vegetation establishment (USDA Forest Service 1979). To avoid the potential decrease in soil temperatures and to allow organic matter to decompose over a longer time period, some land mangers are using "chunking." Chunking is the mechanical breakdown of woody materials to larger sizes rather than the less than 3 cm on a side. Some operations produce woody debris from 15 to 30 cm on a side.

Mechanical treatments are new as a fuel management practice, and few studies are completed that determine their effect on water resources. Nonetheless, best professional judgment would suggest that if soil disturbance is minimized by limiting the number of tractor passes, avoiding steep slopes (greater than 35 percent), and scattering the woody material, overland flow and soil erosion will not be a problem. Vegetative cover reduction will temporarily increase on-site water quantity, which can be utilized by the remaining vegetation. If sufficient watershed area is treated, channels with intermittent flows may become perennial.

Grazing

Grazing animals can be used to remove vegetation. The management objective may be to remove understory vegetation or reduce noxious weeds. The effects of the grazing practices on water quality would be expected to be minimal if best management practices are followed. Most grazing lands are in ponderosa pine or pinyon-juniper type communities.

The ponderosa pine range is the most extensive forested range in the western United States. It occupies the low elevations of the mountains and foothills in many areas, but mixes with other tree species at moderate elevations. This type of community is associated with an understory of bunchgrasses and shrubs. As the tree density increases, there is generally a curvilinear decrease in understory production. This range commonly serves as spring, summer, and fall range for cattle. Both rest-rotation and deferred-rotation grazing systems, under proper stocking rates, benefit these forested ranges in terms of maintaining vegetative cover and precipitation infiltration (Leininger and Stednick 2002).

The pinyon-juniper range is located between the ponderosa pine forest and desert shrub or grassland. The pinyon-juniper range generally occurs on rocky, poorly developed soils, and in many locations it alternates with big sagebrush, which occupies deeper soils. Cattle and sheep frequently graze this range in spring before moving to higher-elevation summer ranges and again in fall as they return to their wintering areas.

Fire suppression and overgrazing by livestock have allowed woodlands to expand both upslope and downslope over the past 100+ years (Gruell 1999). Prescribed burns and mechanical removal of pinyon and juniper trees by chaining—large tractors pulling anchor chains or cables over the land—are frequently used to reduce this invasion. Desirable grasses are also commonly seeded into recently treated areas to increase forage for livestock and wildlife.

The most important deleterious effect of improper range management on water quality is soil erosion and the subsequent suspended sediment production. Vegetative cover and soil properties determine the infiltration rates of precipitation water and the amount of streamflow that occurs on grazed lands. Vegetative cover is the dominant factor in controlling runoff and water erosion from agricultural lands and rangelands. Livestock grazing may alter the natural infiltration-runoff relationships by reducing vegetative cover, reducing and scattering litter, and compacting the soil through trampling. The magnitude of these changes is determined by topography, climate, vegetation, stocking rate, and animal species.

This reduction in vegetative cover may in turn increase the occurrence of overland flow and contribute to the desertification of marginal rangelands. Water yield due to overland flow may be increased by decreased infiltration rates and capacities due to soil compaction. As use of an area increases, so does the probability of soil compaction. Animal bedding grounds, stock trails, watering locations, and salt licks are areas of potential soil compaction. Soil texture, moisture, and the amount of organic matter influence the degree of compaction. Soil compaction may also reduce plant growth or range productivity through changes in soil aeration and soil moisture.

Animal activity along stream channels or other open waters may change the chemical and bacterial quality of water. Specifically, animal feces may contaminate waters with bacteria or act as sources of nitrate and phosphate. Studies of two adjacent pastures along Trout Creek in central Colorado indicated only minor chemical effects of cattle grazing on water quality. The bacterial contamination of the water by fecal matter, however, increased significantly. After the cattle were removed, bacterial counts quickly dropped to background levels (Johnson and others 1978).

The removal of plant cover by grazing may increase the impact of raindrops, decrease the amount of organic matter in the soil, increase surface crusting (puddling), decrease infiltration rates, and increase erosion. Increased overland flow, reduced soil moisture, and increased erosion translate into greater concentrations of suspended sediment. Other water quality concerns, such as increased bacterial and nutrient concentrations, do not appear to be a problem with grazing systems, except perhaps in riparian zones. The impact of livestock grazing on watersheds has recently become a resource management issue of national proportions. Research project data have often been evaluated emotionally or according to the political advantages offered rather than by scientific and objective thinking. Recent interest in federal grazing practices, particularly grazing allotments, may bring a reevaluation of the environmental and economic implications of grazing systems on watershed resources (Leininger and Stednick 2002).

Changes in the chemical quality of water due to grazing activities are generally not significant or long-lasting unless animals and their waste products are concentrated in one area. Grazing under best management practices does not adversely affect water quality (Leininger and Stednick 2002).

Best management practices for grazing include vegetation monitoring. Most water quality related problems result from loss of vegetative cover. Other practices include off-channel water sources, salting, and pasture or allotment rotation.

Pesticides

Vegetation management usually refers to the treatment of competing vegetation to allow the release of the desired species, for example, spraying of hardwoods to release conifer regeneration or growth. Vegetation control or removal by herbicides can be considered a fuel management practice when the target vegetation represents a contribution to the site fuel load. Similarly, removal of noxious species by herbicides may improve the existing vegetation used by grazing. Noxious weeds control is often accomplished with herbicides. Noxious weeds are usually nonnative species that, lacking natural controls, spread quickly and take over or reduce habitat for native species. Vegetation management often includes the protection of desired vegetation from pathogens, competing vegetation, insects, and animals (Michael 2000). Pesticides provide management with an effective and often inexpensive method to achieve these goals. The Federal Insecticide, Fungicide, and Rodenticide Act as amended (PL92-516) provides for the registration of pesticides in the United States. An integral part of protecting public health and environmental values is the requirement that pesticides must be applied according to directions approved by the U.S. Environmental Protection Agency and on the label of every registered pesticide. The USDA Forest Service requires training of personnel who recommend and use pesticides, applicator certification, and safety plans to assure the safety of personnel and the protection of environmental values (Michael 2000).

In most situations, herbicide applications are infrequent and often may be a one-time treatment. Monitoring for chemicals in water bodies depends on the type of pesticide, rate of application, area soils, and precipitation events following the application. Water quality monitoring for chemicals after pesticide application using best management practices shows that little to no chemicals are detected in water bodies. Studies of the effects of forest herbicide use (applied under regulatory guidelines) on streamwater element concentrations revealed that no levels were high enough to warrant concern (Binkley and Brown 1993b; Michael 2000). In general, when pesticides were detected in surface waters after their application, concentrations were well below the threshold of concern.

Today's more commonly used pesticides rapidly degrade in the natural environment, often a half-life of days. Degradation of pesticides includes biological, hydrolytic, and phtolytic processes that occur in the soil and water. Probably the most important process is the breakdown of organic chemicals by soil microorganisms. Most pesticides have a high affinity for clay and organic matter and may be removed from the soil water as they are bound to soil particles. Once bound, pesticides are often difficult to desorb (MacKay 1992; Michael 2000).

When pesticides are applied to wildlands near surface waters, a buffer zone is usually left between the application area and the water resources. The width of the buffer varies with site conditions, site sensitivity, and state or local regulations. Little research has been done on the buffer width necessary on forested landscapes; more work has been done on agricultural lands.

Hand application of pesticides is easily controlled and site personnel can be advised to avoid streams or other sensitive areas. Pesticide analysis is expensive and any monitoring program can use surrogate assessments. Spray cards can be used to assess pesticide coverage and drift. Often the pesticide carrier (diesel) can be looked for in water quality samples to determine if overspray or drift resulted in pesticides entering surface waters.

Conclusions

A variety of fuel management practices are available to decrease fuel load or improve forest heath condition. These treatments have the potential to affect water quality, but the implementation of best management practices (BMPs) will minimize or eliminate potential water quality effects. There is a relationship between the amount of area disturbed and the amount of potential erosion, thus the amount of disturbed area should be minimized. Streamside management zones or streamside buffers are effective in capturing overland flows, removing sediment and nutrients, and aiding in maintaining stream temperature.

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Cumulative Effects of Fuel Treatments on Soil Productivity

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Introduction

Soil quality, function, and productivity potential are interrelated concepts that cover the range of soil properties and their associated ecological processes. Since the passage of the National Forest Management Act in 1976 (NFMA) and related legislation, management of National Forest lands must be done is such a way as to maintain their productive potential as demonstrated through implementation, effectiveness, and validation (research) monitoring. However, the concept of site productivity is not well defined, and the impacts of timber removal or fire on soil productive potential are not well understood or easily measured (Powers 2006). Two main factors make it difficult to define: (1) the variability in soil and climatic conditions across forest sites and (2) the length of time it takes for trees to reach a predictive age. If tree (or vegetative) growth is used as an indicator of productivity, it may take more than 20 years before the consequences of various management practices in many North American ecosystems can be evaluated (Morris and Miller 1994).

In response to this problem, a number of soil-based indices have been proposed as indirect measures of forest site productive potential. For example, Burger and Kelting (1998) suggest that soil monitoring should vary by soil, site, and management practice. Powers and others (1998) recommend establishing a baseline from a soil survey, then use one physical (soil strength), one chemical (anaerobically mineralized nitrogen N), and one biological (soil fauna activity) index to monitor changes in soil properties. Other soil measures of site productivity have been proposed (Burger 1996; Curran and others 2005a; Herrick 2000; Powers and others 1990), but the link between soil indices and site productivity are not conclusive (Curran and others 2005b; Powers and others 1990, 2004). The data from these studies show that soil compaction and organic matter (OM) removal are important drivers in many ecosystem processes, and the maintenance of adequate soil porosity and OM content is important for continued site productivity and ecological function (Jurgensen and others 1997; Powers and others 2004).

Active fire suppression in the western United States during the 20th century has led to OM accumulation in many forest stands that historically supported a regular fire return interval (Oliver and others 1994; Page-Dumroese and others 2003). Forest stands high in OM levels are usually undesirable because of the increased risk from high intensity wildfires and slower OM decomposition rates (Covington and Sackett 1984). These accumulations of woody residue and surface OM from fire suppression activities are likely above the range of natural variability for these ecosystems and would be susceptible to a correspondingly higher loss during a wildfire (Mutch 1995; Page-Dumroese and others 2003). However, previous human disturbances make it difficult to determine baseline, stand level OM values. Fire suppression has also altered tree density, growth, vigor, and susceptibility to diseases and pests (Kilgore 1981), but the effects of this practice on soil properties are unclear (Monleon and others 1997). For instance, fire suppression can result in stagnant nutrient cycles and, therefore, decreased nutrient availability and tree growth (Biswell 1973; Covington and Sackett 1990). Conversely, current growth of ponderosa pine trees on some sites is higher than growth predicted from yield tables developed shortly after fire exclusion (Cochran and Hopkins 1991), which is attributed to a negative impact of fire on soil productivity.

Since the enactment of the Healthy Forest Restoration Act in 2003, forest management decisions to reduce wildfire risk have increasingly relied on partial cuts and prescribed fire to remove small diameter trees and surface OM from forest stands. Low intensity prescribed underburning, thinning, and combined thinning and burning practices are major components of the restoration effort underway in many forests to reduce fuel loads and fire hazards (Stephens and Finney 2002). Such repeated burns and multiple stand entries by mechanical equipment may have cumulative impacts on ecosystem productivity and sustainability at different scales, such as within a cutting unit or an entire watershed (Curran and others 2005a). In this paper, we discuss the effects of mechanical thinning and prescribed fire on soil compaction and OM pools and the impact this could have on residual fuel loads, soil erosion potential, and long-term site productivity (Elliot 2003; Harden and others 2000; Neary and others 2000).

Thinning

Many studies have documented the impacts of clearcut harvesting on soil physical properties, especially compaction (Miller and Anderson 2002; Page-Dumroese and others 2006; Powers 2006; Powers and others 2004). Similar harvesting equipment is also used in thinning operations and could result in soil compaction on repeatedly trafficked areas. Single equipment passes under specific soil moisture and equipment loading conditions (for example, moist soil, fully mechanized harvesting as demonstrated in Curran and others 2005a). Compaction increases soil bulk density and strength, decreases water infiltration and aeration porosity, restricts root growth, increases surface runoff and erosion, and alters heat flux (Greacen and Sands 1980; Williamson and Neilsen 2000). Total pore space is also reduced, especially the volume of large pores (macropores), which are usually filled with air (Siegel-Issam and others 2005). Poor aeration due to compaction is often cited as a cause of declining root growth (for example, Ruark and others 1982; Zaerr 1983). The susceptibility of soil to compaction is a function of soil texture and original bulk density (Page-Dumroese and others 2006; Powers and others 2005; Williamson and Neilson 2000), soil moisture content (Froehlich 1978; Moehring and Rawls 1970), soil OM content (Adams 1973; Howard and others 1981), the number of machine passes (Soane 1990), and the type of machine applying the load (Han and others 2006). Compaction alters air filled pores, which restricts O_2 movement and creates anaerobic conditions (Linn and Doran 1984), causes the accumulation of CO_2 (Conlin and van den Driessche 2000), and reduces the physical habitat of soil macro- and micro-fauna (Hassink and others 1993).

It is assumed that minimizing soil compaction during a timber harvesting operation is critical for maintaining the productive capacity of a site (Powers and others 2004). While soil compaction can cause substantial declines in tree growth and health in some stands (Conlin and van den Driessche 1996; Froehlich and others 1986; Gomez and others 2002; Heninger and others 2002), they can have little or no impact on growth in others (Powers and others 2004). Growth reductions may occur on both coarse- and fine-textured soils (Cochran and Brock 1985; Froehlich and others 1986; Smith and Wass 1980); however, the reduction of macropore space on course-textured soils may increase soil available water holding capacity and thereby increase tree growth (Gomez and others 2002).

Compaction from repeated trafficking on the same plot of land is the most common cumulative soil effect of mechanical site treatments (Geist and others 1989). However, traffic over many portions of a watershed may also lead to dispersed cumulative impacts in the form of lighter compaction affecting a larger area (Curran and others 2005a). Thinning method also has a strong influence on the degree and extent of soil compaction. For example, cut-to-length logging, particularly on slopes less than 35 percent, can result in spatially dispersed traffic patterns if harvesting machine operators can choose their route to a landing. While this type of logging may show fewer surface impacts (displacement and visible machine tracks or ruts) than thinning with designated skid trails, most compaction occurs in the first few passes and soil damage may be more widespread (Curran 1999; Williamson and Neilsen 2000). Log-forwarder impacts occur mostly on main trails without slash mat protection or near landings, locations where the forwarder makes repeated passes. Using skyline logging systems to thin a stand usually results in soil compaction at the landings or is associated with dragging heavy logs. In northeastern Oregon, both skyline logging and harvester/forwarding operations produced less than 10 percent soil compaction on a number of sites (McIver and others 2003). This amount of compaction is much lower than that found in other harvesting studies from the northwest United States (Allen and others 1999; Froehlich and others 1986; Geist and others 1989). These variable results could be due to differences in harvesting techniques, which in turn affects the amount of soil compaction. Leaving slash from thinning or other harvest activities on skid trails has the potential to help buffer machine traffic to lower the impacts on the mineral soil (Han and others 2006), as does thinning a stand when the soil is dry (Han and others 2006), frozen (Bock and van Rees 2002), or has adequate snowpack (Curran 1999). Consequently, managers have a number of options when they need to reduce fuel over large areas.

Another soil disturbance that may occur as a result of compaction and displacement is soil erosion. When surface moisture is impeded from infiltrating it can result in increased overland flow that can cause erosion and effect off-site resources and water quality. Prudent attention to drainage control and access network planning, construction, and maintenance can help minimize risks associated with erosion. An erosion hazard key is discussed later, under planning and monitoring.

Underburning

Underburning is a low intensity prescribed fire that is used to reduce fuel loads and fire hazards in overstocked stands (Monleon and others 1997). Since fire suppression caused a shift in forest structure, frequent underburnings are one method used to restore stands to pre-European settlement fire regimes (Bork 1985). The impacts of prescribed underburning on fuel loads and surface soil conditions can vary considerably depending on fuel characteristics and loading, soil climatic conditions at the time of burning, and resulting soil burn severity (Gundale and others 2005). Nitrification and N-mineralization showed strong positive correlations with fine fuel consumption after underburning in a Montana ponderosa pine (*Pinus ponderosa* Dougl. ex P. & C. Laws) stand (Gundale and others 2005). In contrast, underburning a ponderosa pine stand in central Oregon resulted in a long-term (12 years) decrease in available N, even though short-term increases were found in the surface (0 to 5 cm) mineral soil immediately after the fire (Monleon and others 1997). This lowering of soil N levels and subsequent decrease in tree growth after underburning may support the supposition that fire suppression will increase soil fertility (Cochran and Hopkins 1991).

Underburning alone or in combination with thinning can alter microbial communities in a forest stand by increasing the temperature of the post burn soil surface or changing the availability of organic substrates (Gundale and others 2005). Many studies have shown that soil heating during the burn results in a substantial short-term loss of microbial biomass or a shift in community structure (Choromanska and DeLuca 2002; Korb and others 2003; Pietikainen and Fritze 1995). These changes, and their duration, are the result of the interactions of fuel load, fuel moisture content, weather conditions, landscape position, light-up sequence, and resulting fire behavior and resident time combined with heat transfer variability within the soil profile (Busse and others 2005; Hungerford and others 1991). If a prescribed underburn occurs after a stand is thinned, the increased fine fuel load usually results in a higher intensity fire, more OM loss, and changes in soil C and N (Pietikainen and others 2000). Total C in the surface OM can also be significantly higher after thinning alone as compared to thinning and underburning (Gundale and others 2005), but is dependent on the amount of C in the undisturbed stand (Page-Dumroese and Jurgensen 2006). The lowering of surface C and N pools by underburning is normally short lived, as OM accumulates from the residual trees (Gundale and others 2005).

The intent of underburning is to produce a low intensity, fast moving fire that leaves much of the humus layer intact (McCandliss 2002) to protect the mineral soil from raindrop splash and erosion. However, if large fuel (>7 cm diameter) are dry during the underburning, there can be a significant reduction in the amount of coarse wood on the soil surface (Youngblood and others 2006), which may affect many species of fungi, cryptogams, invertebrates, and vertebrates (Harmon and others 1986). The amount of woody residue remaining in a stand after underburning will vary depending on fuel load, moisture content, fire intensity, residence time, and suppression activity. Compared to other methods of fuel treatment (thinning, thinning and underburning combined), underburning alone usually results in the lowest quantity of residual coarse wood. For example, underburning resulted in less than 30 logs/ha, thinned and burned stands ~50 logs/ha, thinning alone ~150 logs/ha, and the control stand had larger than 200 logs/ha (Youngblood and others 2006). Of these residual logs, decay class 5 logs (Triska and Cromack 1979) comprised 18 percent of the coarse woody residue in the thinned only and control treatments, but were only 7 percent in the underburned and the thinned and burned treatment (Youngblood and others 2006).

Planning and Monitoring

The development of a hazard assessment process to determine how sensitive a soil may be to mechanical and/or fuel reduction treatments can help minimize risk on forest sites and watersheds. For example, the Forest Practices Code of British Columbia, now replaced by the Forest and Range Practices Act (Province of BC 2004), defines site hazards as a combination of soil texture, coarse fragment content and soil moisture regime. This, in turn, can help guide practitioners in deciding on the appropriate types of equipment to be used, the harvest and maintenance schedule, or type of harvest operation (see Erosion Hazard key as an example in table 1, which is based on science and rationale presented in Carr and others 1991, with updates based on the research of Commandeur 1994). The Weyerhaeuser Company assesses risk to site productivity from all types of management activities to site productivity for each soil mapping unit, largely based on soil physical properties (Heninger and others 1999). The risk ratings are based on modal soil characteristics for each soil series and site factors. Principles behind risk rating with further examples are discussed in Curran and others (2005b, 2007). Compaction, displacement, erosion, and slope stability risks are often interpreted from soil mapping (that needs to be verified onsite) or site specific data collected for harvest planning (for example, as per Curran and others 2000) and prescribed fire assessments. Harvesting,

Table 1. Example of a hazard rating system for surface soil erosion within a cutting unit (adapted from the British Columbia Ministry of Forests from the Forest Practices Code soil disturbance hazard guidebook, currently available in Curran and others 2000).

	Degree of contribution of factors			
Site factors	**Low**	**Moderate**	**High**	**Very High**
Climate precipitation factor	Low	Moderate	High	Very high
(points)	**2**	**4**	**6**	**8**
Topography				
slope gradient (%)	0-10	11-20	21-50	>50
(points)	**1**	**3**	**6**	**9**
length/uniformity	Short broken	Short uniform	Long broken	Long uniform
(points)	**1**	**2**	**3**	**4**
Depth to water-restricting layer (cm)	>90	61-90	30-60	<30
(points)	**1**	**2**	**3**	**4**
Surface soil detachablity (0-15 cm)	SC, C, SiC	SiCl, Cl, SCL	SL, L	Si, SiL, fSL, LS, S
(points)	**1**	**2**	**4**	**8**
Surface coarse fragments (0-15 cm)	>60	31-60	16-30	<16
(points)	**1**	**2**	**3**	**4**
Subsoil permeability (16-60cm)	S, LS, SL, fSL	L, SiL, Si	Cl, SCl, SiCl	SC, SiC
(points)	**1**	**2**	**3**	**4**
Erosion hazard rating	**Low**	**Moderate**	**High**	**Very High**
(point total)	**<16**	**16-22**	**23-31**	**>31**

thinning, and underburning strategies have been described for meeting soil disturbance standards under site conditions in western Washington and Oregon by Heninger and others (1997) and for Interior BC by Curran (1999). The objective is to match site treatment to site disturbance sensitivity. Ground based equipment may be restricted to designated trails or allowed to travel overland depending on the soil and climatic conditions (in other words, dry soil, frozen soil, or snowpack).

Assessing soil changes associated with management is a critical step toward understanding which sites are amenable to trafficking or burning treatments. Generally, monitoring after underburning or thinning activities is collected through transect sampling of continuous line or point data (for example, Howes and others 1983; BC Ministry of Forests 2001, respectively). However, soil quality evaluations must also assess cumulative management impacts at a landscape scale, which is much harder to accomplish than a simple point sampling methodology. When working in larger areas, sampling schemes can be stratified (for example, by soil texture, parent material, vegetation type, harvest methods, etc.) to improve sampling efficiency and reduce costs (Herrick 2000).

Visual disturbance class indicators (Curran and others 2005b) have been used to assess soil displacement or compaction severity after mechanical operations. Such visual class systems are also amenable to the collection of burn severity categories (fire caused changes to soil hydrologic function as evidenced by soil characteristics) and to visually evaluate the extent of burning into the mineral soil and loss of forest floor and surface fuel (Ice and others 2004). The visual assessment of surface OM changes after thinning or underburning is often used as a surrogate or proxy for changes in soil properties. These properties are associated with loss of soil aggregates and increased erosion, which could indicate a loss of site productivity. Placing management impacts in the broader context of the range of natural variability observed before harvesting is another appropriate method for evaluating the consequences of thinning and underburning (Bock and Van Rees 2002; Grigal and Vance 2000; Landres and others 1999; Pennock and van Kessel 1997). Using baseline data from non-harvested stands will help quantify the magnitude of variability so that change in a soil property can be gauged against this variability and help define the processes that thinning or underburning operations influence (Grigal and Vance 2000; Page-Dumroese and others 2000).

Changes in soil OM can also be used as an indicator of soil biological activity and, indirectly, the effect of thinning and underburning on soil quality and site sustainability (Weil and others 2003). Weil and others (2003) developed a simple method to measure active soil C and they note that a change in the labile OM fraction can give an early indication of soil degradation.

All of the methods listed above need to be applied in an adaptive management framework that will allow for changes in methods and procedures as new information or techniques become available (Curran and others 2005c). This adaptive process will ensure that the monitoring of thinning and underburning treatments is using best management practices, coordinating development of training materials and tools, and reporting post treatment evaluations.

Long term research projects are one of the best methods for quantifying the consequences of fuel reduction treatments and evaluation of monitoring strategies. Development of effective and practical methods for assessing changes in soil productivity has been the major focus of the North American Long Term Soil Productivity (LTSP) study (Powers and others 2004). Although designed to measure the long-term impacts of compaction and OM removal after clear-cut timber harvest, this study will also help to validate soil quality standards and monitoring changes in soil productivity after fuel reduction operations. While the LTSP study did not have a fire component, the Fire and Fire Surrogate study was established nationwide to evaluate the ecological impacts of thinning and burning treatments on vegetation, fuel, soils, and other ecosystem functions (Weatherspoon 2000).

Conclusions and Management Implications

Restoration treatments used to restore or enhance ecological processes and/or structure to a forest stand usually involve some variations of thinning and burning. Numerous soil impacts can occur from these treatments, but the impacts can be quite variable, depending on both manageable factors and inherent site sensitivity factors, which together dictate the severity and extent of compaction and burn severity. Manageable factors include equipment configuration and use, decisions on fuel arrangement and moisture levels, light-up sequence, and resulting fire behavior, all timed to take advantage of seasonal soil conditions to minimize impacts. Inherent site sensitivity depends on soil texture and mineralogy, coarse fragment content and arrangement, and organic matter levels and rooting, among other factors. The impacts of commercial or pre-commercial thinning operations (with or without burning) on residual tree growth will have to be measured to calibrate (validate) soil disturbance proxies and feed results into practice improvements to ensure sustainable productivity. When pre-treatment data is available, post-treatment monitoring can use soil disturbance proxies to provide an indirect measure of the impact that a fuel reduction treatment will have on soil properties that are currently considered to control productivity and hydrologic function. The results from these monitoring studies need to be validated against subsequent tree or stand growth. In contrast to clear-cut harvesting, the impacts of thinning operations on changes in soil quality can be difficult to quantify. Although the impacts of thinning operations on soil properties can be assessed relatively easily, the associated changes in site productivity are not documented. Thinning reduces total stand biomass, but can increase the growth of individual trees (Karlsson 2006; Liechty and others 1986). If the response of stand productivity to thinning is only measured on the residual trees, the negative impacts of soil compaction could be masked by the increased growth of the remaining trees.

Increasingly, managers must balance biomass removal to reduce wildfire risk with maintaining soil productivity. Thinning and underburning treatments require accurate monitoring of soil impacts using proxies that are calibrated against longer term effects over time. In the interim, these proxies need to be based on best available science and disturbance limits conservatively set to ensure that productivity and hydrologic function will be maintained. The wide variability in forest soil properties makes this a challenging task. However, by using risk rating systems, various soil factors affecting site sensitivity (response) can be organized and managed during planning and operations. The objective is to identify the inherent site sensitivities and/or seasonal soil conditions that create vulnerability to negative impacts of the selected fuel reduction treatments.

These include factors such as specific soil texture, rock content, low soil fertility, a high proportion of OM pools on the soil surface, or topographic features. Successful fuel reduction monitoring protocols must use proxies that integrate the correct combination of chemical, physical, and biological properties and are calibrated to demonstrate the maintenance of long-term productivity. Use of best management practices (for example, site characterization, Curran and others 2000), detailed soil inventory, use of models to predict erosion (for instance, WEPP), thinning and underburning strategies to minimize disturbance, climatic considerations, soil disturbance monitoring, and prudent use of rehabilitation, all in an adaptive management approach (Curran and others 2005a and b), will help limit localized soil damage and reduce the potential of cumulative fuel reduction effects within a watershed. Ultimately, net primary productivity is the measure to determine the positive or negative impacts of thinning and underburning treatments and will have to be measured in controlled experiments that also calibrate the operational disturbance proxies. Consequently, the results from the North American LTSP network, the Fire and Fire Surrogate study, and other long-term studies must be an integral part of the effort to evaluate both short and long-term impacts of fuel reduction treatments on soil productivity and the validation of monitoring protocols and standards.

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Chapter 10.

Potential Effects of Fuel Management Activities on Riparian Areas

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Introduction

A significant increase in fuel management treatments is underway as the Forest Service and other natural resource agencies implement the National Fire Plan (USDA USDI 2001), the Healthy Forests Restoration Act (GAO 2003; HFRA 2003), and the President's Healthy Forest Initiative (Dombeck and others 2004; Graham and others 2004; Stephens and Ruth 2005). One of the four goals of the National Fire Plan Comprehensive Strategy is to reduce hazardous fuel, thus potentially decreasing the risk of severe wildfire and modifying fire behavior so that some wildland fires may be more readily and safely suppressed (Graham and others 2004; USDA USDI 2002).

The general objective of this report is to provide resource managers and specialists with a summary of existing knowledge that they can use to evaluate the impacts of proposed fuel treatment projects, particularly the cumulative effects on watersheds. Cumulative watershed effects are defined as "the environmental changes that are affected by more than one land use activity…" (Reid 1998). Cumulative impacts can result from individually minor but collectively significant actions that occur over a period of time (Belt and others 1992). In other words, the effects may prove to be additive or interactive. Riparian areas can act as both moderators and integrators of activities that occur within a watershed. Consideration of the potential effects of fuel treatments on ecological functions of riparian areas is essential in determining cumulative watershed effects.

The objective of this chapter is to synthesize the current state of knowledge about the potential impacts of streamside and upland fuel management on the structure and function of riparian areas. Although research is underway, little has been published on these topics, and most examples from the literature are derived from studies that investigated the effects of forest harvest or wildland fire. Although findings from studies conducted throughout the nation are presented in this chapter, the focus is on riparian areas in mountainous regions of the western United States The influence of fuel management practices on surface water quality and aquatic biota are addressed elsewhere in this report (Chapter 8; Chapter 11; Chapter 12).

Riparian Areas: Definition, Natural Variability, and Management

Definition of Riparian Areas

Riparian areas have been ecologically defined as "three dimensional zones of direct physical and biotic interactions between terrestrial and aquatic ecosystems, with boundaries extending outward to the limits of flooding and upward into the canopy of streamside vegetation" (Gregory and others 1991). The first dimension of riparian areas is the longitudinal continuum from headwaters to the mouths of streams and rivers and ultimately the oceans (Vannote and others 1980). The second is the vertical dimension that extends upward into the vegetation canopy and downward into the subsurface and includes hyporheic and belowground interactions for the length of the stream-riparian corridor (Edwards 1998; Stanford and Ward 1988). The third dimension is lateral, extending to the limits of flooding on either side of the stream or river (Stanford and Ward 1993). The vertical and lateral dimensions include the distinct microclimates often associated with riparian areas. In this ecological framework, riparian areas are viewed in terms of spatial and temporal patterns of hydrologic and geomorphic processes, terrestrial plant succession, and aquatic ecosystems (Gregory and others 1991; Naiman and Decamps 1997). In the scientific literature the terms "riparian habitat," "riparian area," and "riparian ecosystem" are used somewhat interchangeably, and pertain to the ecologically defined area adjacent to streams (Knutson and Naef 1997). In this chapter, we use the term "riparian area" when referring to the three dimensional streamside zone (Gregory and others 1991). We focus on riparian areas bordering streams, rivers, and springs, although much of the information presented in this chapter also pertains to vegetated areas surrounding lentic waters such as lakes and wetlands.

To assist in managing riparian areas, numerous administrative definitions have been developed along with terms such as "streamside management zones," "riparian habitat areas," "riparian buffers," and "riparian management zones". Most definitions are based on attributes that differentiate streamside areas from adjacent uplands (Belt and others 1992; Knutson and Naef 1997), such as moist soils and occurrence of plant species and communities that are adapted to them or may rely on somewhat arbitrary boundaries such as a fixed distance on each side of a stream channel (Belt and others 1992). We use the term "riparian buffer" when referring to any administratively defined area adjacent to flowing or lentic surface water, including those that are specified by a given distance from the stream or presence of certain ecological attributes.

As suggested by both ecological and some management definitions, riparian areas and influence do not stop at a uniform distance from the stream bank. Instead, they are composed of mosaics of land forms, plant communities, and environments that vary in width and shape within the larger landscape (Gregory and others 1991; Naiman and Decamps 1997) and are not always easily delineated. The Federal Interagency Stream Restoration Working Group recognized the following three components of the stream corridor: the *stream channel*, with flowing water at least part of the year; *the floodplain*, a highly variable area on one or both sides of the stream channel that is inundated by floodwaters at some interval; and the *transitional upland fringe*, a portion of the upland on one or both sides of the floodplain that serves as a transitional zone or edge between the floodplain and the surrounding landscape (Federal Interagency Stream Restoration Working Group 1998). In this chapter, we use the term "stream-riparian corridor" when referring to the stream channel, adjacent floodplains, and the transitional upland fringe. Each of these components should be considered in fuel management because of the linkages and feedbacks that occur among the channel, riparian area, and uplands (Federal Interagency Stream Restoration Working Group 1998).

Natural Variability Among Riparian Areas

Stream-riparian corridors are highly variable and characterized by multidimensional spatial gradients. The effects of fuel reduction treatments on riparian areas will depend largely on the location of the treatments within a watershed, that is, if they are adjacent to the channel or in the uplands or headwaters, middle or lower portion of the drainage, and positioned relative to tributaries in the stream network. The factors that vary in different portions of a watershed, including soil characteristics, slope, vegetation cover, moisture, and microclimate, also influence the behavior of wildland fire and the potential responses of riparian areas to fuel reduction treatments. Effects will also vary considerably depending on the type of treatment. Some treatments in close proximity to riparian areas may have little effect, such as a relatively cool prescribed fire with low flame length, whereas other treatments may significantly influence riparian areas.

The longitudinal profile of many streams in the western United States can be roughly divided into three zones, which are described based on a simple model of dominant erosion processes: the steep *headwaters*, central *transfer zone*, and low elevation *depositional zone* (Schumm 1977). Each zone is also characterized by riparian plant species, growth forms, and communities that reflect the elevation, geomorphic position, hydrologic and sediment regimes, and past disturbance within a watershed (Carsey and others 2003; Crowe and Clausnitzer 1997; Youngblood and others 1985). The three-zone model is frequently presented for mountain streams; however, the general erosion and geomorphic patterns are also applicable to drainages with less topographic relief. Other stream classifications based on physical processes also help to explain the interactions between the distribution of riparian vegetation and watershed variables and emphasize the role of temporal and spatial scales in understanding the interdependence of physical and biotic processes (Frissell and others 1986; Montgomery 1999; Montgomery and Buffington 1998; Poole 2002; Rosgen 1994; Ward and Stanford 1995).

The diversity of riparian areas is also attributed to the temporal variability in physical events and natural disturbances, such as floods, debris flows, landslides, and wildland fire along with the subsequent successional changes in riparian plant communities over time (Gecy and Wilson 1990; Naiman and others 2005). Fire is a critical disturbance that has shaped the structure of forests and rangelands throughout the western United States (Agee 1993, 1998; Stephens and Ruth 2005). Although limited research has investigated the role of fire in structuring streamside vegetation, riparian plant communities evolved within the ecological context of regional fire regimes (Arno 2000). Studies in several parts of the western United States have revealed that historical fire frequencies in uplands and riparian areas were often comparable (Macdonald and others 2004; Olson and Agee 2005), whereas in others, riparian fires were less frequent but more severe than those in uplands (Arno 1996; Everett and others 2003). Moreover, dendrochronological analyses often detected the same fire events in upland forests and adjacent riparian areas. The decline in fire frequencies in both areas corresponded with the onset of effective fire suppression (Everett and others 2003). Effects of both wildland fire and fire suppression have likely influenced riparian vegetation and functions and should be acknowledged during planning and implementation of fuel reduction treatments. Predicting the potential impacts of fuel treatments on riparian areas requires consideration of the fire history and natural fire regime along elevation gradients throughout the treated watershed and surrounding landscape (Agee 1991, 1993; Arno 2000).

Current attributes and condition of riparian plant communities reflect the historically recent (approximately 100 to 200 years) physical conditions of the landscape as well as land management activities (NCASI 2005). Forest harvest, livestock grazing, road construction, inadequate road maintenance, flow alteration (dams and diversions), and recreation have altered composition and structure of riparian plant communities (Kauffman 2004; NCASI 2005). Removal of beaver has changed stream and floodplain hydrology in watersheds throughout the western United States and directly and indirectly influenced riparian vegetation and nutrient and organic matter dynamics. Mining activities, particularly dredging and hydraulic mining, have left a lasting legacy on the geomorphology and hydrology of many western stream riparian corridors (Wohl 2001).

Although portions of many riparian areas along perennial streams are currently protected, the lingering effects of land management prior to the establishment of buffers are likely to influence the structure and composition of riparian areas for decades to centuries (Young and others 1994). Legacies of past management within watersheds could potentially confound responses to fuel reduction treatments.

Best Management Practices and Protection of Riparian Areas

Riparian areas cover a relatively small area, yet they are disproportionately important for maintenance of water quality and quantity (water storage and aquifer recharge), habitat for aquatic and terrestrial biota, sediment retention, stream bank building and maintenance, and provision of services of economic and social value such as livestock grazing and recreation (table 1) (Gregory and others 1991; Naiman and Decamps 1997; Naiman and others 2005; Prichard and others 1998). On National Forest lands, protection of riparian areas is often governed by special rules, stated as Standards and Guidelines in the Forest Plan for each National Forest, which frequently include sets of best management practices (BMPs) (Belt and others 1992; Gregory 1997; Mosley and others 1997). BMPs are officially approved practices and techniques that are generally cost effective and practicable means of reducing management impacts on streams, valued riparian functions, or ecosystem services (Belt and others 1992; Mosley and others 1997). The management of riparian areas can generally be defined as custodial, in which the riparian areas are protected to maintain specific functions (table 1) (Gregory 1997). The general objective of most BMPs is to protect water quality and habitat along streams from timber harvest, road construction, grazing, recreation, and other land use activities (Belt and others 1992; Mosley and others 1997) and is often accompanied by the designation of riparian buffers (Norris 1993).

Riparian buffers contribute to watershed protection by restricting management activities and other human caused disturbances that alter ecological conditions of stream riparian corridors (Norris 1993). Riparian influence decreases with distance from the stream channel (fig. 1) (FEMAT 1993). Depending on stream width, location within a drainage basin, and management concerns, the required riparian buffer width may vary from 5 ft to 300 ft on each side of the stream (Belt and others 1992; Lee and others 2004). Streams used for domestic water supplies are accorded wider riparian buffers to protect downstream reservoirs from non-point pollution resulting from forest management (Belt and others 1992). Many federally listed plant and animal species (frequently selected as management indicator species) require riparian areas as habitat. Streams that are important for spawning, rearing, or migration of sensitive fish species often receive additional protection in the form of wider buffers (USDA 1995). Existence of a riparian buffer, however, does not preclude all types of management activities. Lee and others (2004) noted that about 80 percent of state and provincial jurisdictions permitted riparian timber harvest. Regulations on public lands are somewhat more restrictive but still allow active riparian management.

The effectiveness of BMPs in mitigating the impacts of land management varies considerably depending on local conditions, management guidance and practices, and the stream or riparian feature of concern (Belt and others 1992; Weller and others 1998). Implementation of BMPs and establishment of riparian buffers have generally decreased the negative effects of forest harvest activities on surface water quality (Belt and others 1992; Norris 1993; Osborne and Kovacic 1993). However, less is known regarding BMP effectiveness in protecting other riparian functions (table 1). For example, in western Washington, Brosofske and others (1997) found that forest harvest strongly affected the riparian microclimate despite designated buffers (mean buffer width, 72 ft; range, 40 to 236 ft). Whereas riparian buffers and BMPs will likely assist in mitigating some impacts of upland fuel reduction treatments, additional precautions and actions may be necessary to protect particular riparian functions. In burned watersheds or areas that have experienced insect caused mortality, riparian buffers may consist of mostly dead trees, and streamside fuel loads may cause concern about fire risk and potential fire behavior. Although the utility of such buffers is questionable for functions such as

Table 1. Functions of riparian areas and key relationships to ecological service (modified from NRC 2002; Naiman and others 2005).

Riparian functions	Indicators of riparian functions	On-site or off-site effects of functions	Valued goods and services provided
Hydrology and sediment dynamics			
Short-term storage of surface water	Connectivity of floodplain and stream channel	Attenuates downstream flood peaks	Reduces damage from floodwaters
Maintenance of high water table	Presence of flood-tolerant, hydrophytic, & mesic plant species	Maintenance of distinct vegetation, particularly in arid climates	Contributes to regional biodiversity through provision of habitat
Retention and transport of sediments; riparian vegetation decreases stream bank erosion	Riffle-pool sequences, point bars, floodplain terraces, and bank stability	Contributes to fluvial processes	Creates predictable yet dynamic channel and floodplain features
Biogeochemistry and nutrient cycling			
Riparian vegetation provides source of organic carbon (allochthonous inputs to streams; organic matter inputs to soils)	Healthy mosaic of riparian vegetation	Maintenance of aquatic and terrestrial food webs	Supports terrestrial and aquatic biodiversity
Transformation and retention of nutrients and pollutants	Water quality and biotic indicators	Interception of nutrients and toxicants from runoff; water quality	Improvement and maintenance of water quality
Sequestration of carbon in riparian soils	Occurrence, extent, & distribution of organic-rich soils	Contributes to nutrient retention and carbon sequestration	Potentially ameliorates global warming; provides source of dissolved carbon to streams via subsurface flow paths
Distinctive terrestrial and aquatic habitat			
Contributes to overall biodiversity and biocomplexity	High species richness—plants and animals	Provides reservoirs for genetic diversity	Supports regional biodiversity
Maintenance of streamside microclimate	Presence of shade-producing canopy; healthy populations of native terrestrial and aquatic biota	Provides shade and thermal insulation to stream; provides migratory corridors for terrestrial and aquatic species	Maintains habitat for sensitive species (amphibians, cold-water fishes, others)
Contribution to aquatic habitat; provision of large wood (CWD/LWD inputs)	Aquatic habitat complexity (pool-riffle sequences, debris dams); maintenance of aquatic biota	Maintenance of aquatic biota	Maintenance of fisheries, recreation
Provision of structural diversity	Availability of nesting/rearing habitat; presence of appropriate indicator wildlife species (for example, neotropical migrants)	Maintenance of global biodiversity; provides migratory corridors for terrestrial and aquatic species	Recreation: bird watching, wildlife enjoyment, and game hunting

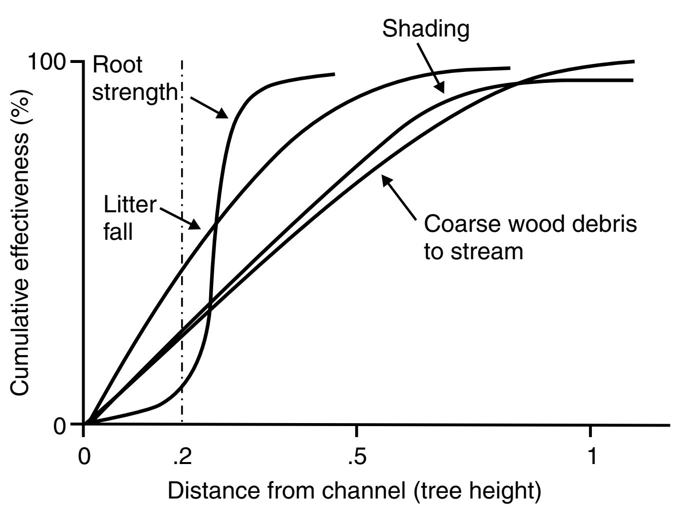

Figure 1. Generalized curves indicating cumulative percent effectiveness of riparian ecological functions occurring with varying distance from the stream channel (FEMAT 1993).

maintenance of stream water quality, they may provide critical wildlife habitat and an important source of large wood for streams and floodplains. In these cases, as well as for prescriptions that are being planned and conducted within riparian areas, managers may need to develop and implement additional on-site BMPs and riparian-specific prescriptions to protect streams and valued riparian functions.

Potential Effects of Fuel Management Activities on Riparian Areas

Fuel Management Treatments

Fuel reduction treatments are being planned and implemented throughout the western United States (http://www.fireplan.gov). Most treatments have the overall goal of decreasing the risk of high severity fire by fragmenting the forest canopy, removing ladder fuel, and reducing the abundance of ground fuel (Peterson and others 2005). Forest fuelbeds can be categorized into six strata:

1. forest canopy,
2. small trees and shrubs,
3. low vegetation,
4. dead wood,
5. moss, lichens, and litter, and
6. duff (Sandberg and others 2001).

Fuel reduction treatments typically target crown, ladder, and surface fuel (Peterson and others 2003) and include prescribed fire, thinning and other silvicultural operations, and chemical and biological treatment (Graham and others 2004). There is considerable variation within each treatment type. For example, a controlled burn prescription may include different burn intensities and different preparation procedures. Also, various combinations of different treatments are used to modify vegetation in each stratum and depend on project objectives, targeted fuel, current condition of the vegetation, past management, and logistics (Peterson and others 2005) (Chapter 4). Each treatment type and combination could have very different individual and cumulative environmental effects on ecosystems processes and attributes, ranging from negative to positive to benign. Also, fuel reduction projects usually require a sequence of multiple treatments staged over a period of time. Discussion of the variation in fuel reduction treatments and potential impacts of each type are beyond the scope of this review chapter. However, the effectiveness of projects in reducing site specific fire hazard and minimizing negative environmental consequences will depend on knowledge of natural fire regimes and existing data on current and historical forest structure and fuel distribution (Peterson and others 2005). The current management of natural ignitions or wildland fire use must also be integrated into planning for fuel reduction treatments and considered in assessment of cumulative impacts.

For most riparian plant communities, few data are available on fuel loads, characteristics, or distribution (Dwire and Kauffman 2003); however, there is a perception that current fuel quantities in some riparian areas are hazardous and constitute a fire risk. This has likely resulted from the recognition that fire in some riparian areas was historically common and that fire suppression has contributed to the accumulation of fuel in riparian areas as it has in uplands, particularly in forest types that historically supported low intensity, high frequency fire (Everett and others 2003; Olson and Agee 2005). Given data limitations on historical composition and structure of riparian vegetation, managers are encouraged to consider the natural fire regime and fire history of the watersheds to be treated when they define target fuel loads for riparian areas.

Despite the uncertainty, fuel reduction treatments are underway in riparian areas (http://www.fireplan.gov/reports) and for some projects objectives extend beyond the reduction of fire risk. For example, prescribed fire has been used to control invasive species along streams and rivers (*Tamarix* spp. in the Bighorn River Basin, Wyoming, Bureau of Land Management Worland Field Office) and enhance wildlife habitat through the regeneration of willows (Bridger-Teton National Forest and Grand Teton National Park, Wyoming). Mechanical methods have been employed to protect structures (for example, restrooms, interpretive displays, and developed campsites) at riparian recreational sites such as picnic areas and campgrounds (for example, along the Colorado River near Moab, Utah, Bureau of Land Management Moab Field Office). These projects are generally quite small (less than 5 acres), and the ecological effects are likely to be fairly local. Most projects are being implemented in riparian areas that have undergone considerable management and disturbance, including wildfire, infestation by exotic species, timber harvest, and road and recreational development. These fuel reduction projects are providing managers with opportunities to reduce fire risk, remove invasive species, and restore streamside areas to conditions that support valued riparian species.

Riparian Vegetation

Riparian plant communities frequently constitute the most floristically and structurally diverse vegetation in a given region (Naiman and others 1993, 1998, 2005; Pollock and others 1998; Tabacchi and others 1998). Because of their transitional location at the land water ecotone, riparian plant communities may include upland, riparian, and wetland species, and thus maintain high levels of beta and gamma diversity (Pollock and others 1998), and a range of life forms and functional groups (NCASI 2005; Pabst and Spies 1999). Numerous vascular plant species of concern occur in riparian habitats (CNPS 1997; Eastman 1990). Riparian plant species have an array of morphological, physiological, and reproductive adaptations for survival in variable and frequently disturbed environments. Specific adaptations include those related to flooding, erosion, sediment deposition, seasonally saturated soil environments, physical abrasion, and stem breakage. Patterns of riparian plant community development and structure are driven by responses to disturbance, hydrologic and geomorphic variables, soil and substrate characteristics, and biological attributes related to succession (Baker 1989). Characteristics of vegetation structure are similar to those used to categorize fuelbed strata (Peterson and others 2005) and include age class, structural type, size, shape, and spatial distribution (vertical and horizontal) of vegetation components (Spies 1998).

Limited research has been conducted on the effects of fuel reduction treatments on riparian vegetation. However, results from studies of prescribed fire and more extensive forest harvest treatments in upland and riparian areas may be helpful in evaluating potential impacts (table 2) (NCASI 2005). Bêche and others (2005) sampled riparian vegetation before and after a fall prescribed burn along stream segments in the central Sierra Nevada Mountains of California. They found that ground cover taxa richness decreased more in the burned plots than unburned plots, diversity (Simpson's D) decreased in both, and ordination results showed little difference in community composition between burned and unburned riparian plots (table 2) (Bêche and others 2005). Similar results have been observed in other locations following prescribed fire (Elliott and others 1999) and may partly be due to patchy burning. In the Oregon Coast Range, riparian herbaceous plant diversity did not differ significantly between unharvested riparian buffers surrounded by logged uplands and undisturbed riparian forests located in unharvested watersheds (table 2) (Hibbs and Giordano 1996; Hibbs and Bower 2001). In forested uplands of the Cascade Mountains (Oregon and Washington), clearcut logging and other types of forest harvest have tended to reduce plant diversity initially, although most shrub and understory species recover with time as succession proceeds (Halpern and others 1992; Halpern and Spies 1995). It should be noted, however, that certain rare species have been locally extirpated by forest harvest (Halpern and Spies 1995; Hansen and others 1991). As expected, plant cover and structure were dramatically reduced in

Table 2. Effects of wildfire, prescribed burning, and forest harvest treatments on diversity of forest and riparian vegetation.

Source	Time scale[a] (years)	Treatment and study type	Location	Findings
Andrus and Froehlich 1988	+2 to + 135	Logging, wildfire, logging + wildfire; retrospective sampling in riparian plots	Western hemlock–Douglas fir forest type; Oregon Coast Range (28 streams)	Rapid regeneration of shrub and herbaceous species; initial increase in exotic species; overall increase in alder cover/ dominance
Nierenberg and Hibbs 2000	+ 145	Stand-replacing wildfire; retrospective sampling in riparian plots	Western hemlock–Douglas fir forest type; Oregon Coast Oregon (9 streams)	Understory shrubs and red alder dominate initially; eventually replaced by conifers
Bêche and others 2005	-1 to +1 (with unburned controls)	Fall prescribed burn in riparian plots; experimental study	Mixed-conifer forest type; Sierra Nevada, CA;	No clear treatment effects in riparian community composition; diversity decreased in both burned and unburned plots
Elliott and others 1999	-1 to +2	Spring prescribed fire on hillslope gradient including riparian cove; experimental study	Mixed-oak and pine/ hardwood forest types; No. Carolina	No change in riparian species composition
Hibbs and Giordano 1996	+1 to + 32 (with controls)	Unharvested alder-dominated buffers across chronosequence of upland harvest compared to alder-dominated riparian forests undisturbed by upland logging; retrospective sampling in riparian plots	Western hemlock–Douglas fir forest type; Oregon Coast Range	No difference in herbaceous species richness, evenness, or diversity between buffered and undisturbed plots
Hibbs and Bower 2001 Pabst and Spies 1999	+1 to +33	Unharvested riparian buffers across chronosequence since upland harvest; retrospective sampling in riparian buffers; compared buffer results to those from unmanaged riparian areas	Four overstory canopy types: pure conifer (western hemlock–Douglas fir); conifer dominated; pure hardwood (alder, maple), hardwood dominated; Oregon Coast Range	Understory shrub and herbaceous diversity strongly correlated with canopy cover type; no strong differences in shrub and herbaceous cover or composition between riparian buffers and undisturbed riparian forests.
Halpern and Spies 1995	Varied; for most plots, before (-1) and after (+2 to +20) logging	Clear-cut logging, slash burning, thinning; permanent plot and chronosequence sampling in managed and unmananged upland forests	Douglas fir-dominated (young, mature, old-growth); West Cascades, Oregon, and Washington	Temporal trends varied; for most plots, understory richness was reduced following logging, but recovered over time

[a] Time scale relative to treatment (year of treatment = 0)

the first few years following prescribed burning and forest harvest treatments (Bêche and others 2005; Halpern and Spies 1995).

Management activities have also increased the vulnerability of riparian areas to invasion by nonnative species (DeFerrari and Naiman 1994; Fleischner 1994; Parks and others 2005; Planty-Tabacchi and others 1996). Following forest harvest, the occurrence of nonnative species has increased at some sites (Andrus and Froehlich 1988; Halpern and Spies 1995). Livestock grazing has led to the introduction of both non-indigenous pasture species and noxious range weeds throughout the western United States, including riparian areas (Fleischner 1994; Hessburg and Agee 2003). Many stream valleys serve as transportation corridors, and roads and trails—known to be major conduits for dispersal of nonnative plants—are frequently located within floodplains (Forman and Alexander 1998; Gelbard and Belnap 2003; Trombulak and Frissell 2000). The potential for introduction or increased cover of invasive species is an important consideration in the planning and implementation of fuel reduction treatments (Harrod 2001).

An immediate goal of most fuel reduction treatments in upland and riparian areas is to change vegetative structure, although the longer term changes on plant community composition, as well as other ecological consequences, are difficult to predict. In many cases, fire managers are able to implement controlled burns by prescription to obtain the desired effects (for example, no tree mortality or mortality of certain size classes). Reports of successful implementation of prescribed burns are not yet generally available. Monitoring has been minimal and little is known about meeting the longer term project objectives, particularly for riparian areas. Many riparian species appear to be fairly resilient to disturbance, particularly fire (Dwire and Kauffman 2003). Because of the ecological importance of riparian vegetation and the experimental nature of most fuel reduction treatments in riparian areas, monitoring before and after treatment to evaluate achievement of objectives, including the response of streamside plant communities, will assist in advancing our understanding and avoiding litigation.

Terrestrial and Aquatic Habitat in Stream-Riparian Corridors

Terrestrial Habitat

The critical importance of riparian areas for wildlife, particularly in arid portions of the western United States, is well recognized (Kauffman and others 2001; Kelsey and West 1998; Raedeke 1988; Thomas and others 1979). Characteristics of stream-riparian corridors that are important for wildlife are related to the transitional nature of the interface between upland and aquatic habitats, the resulting microclimates and provision of water, food, and cover, and the generally linear shapes with high edge to area ratios that serve as routes of seasonal migration for many vertebrate species (table 1) (Kauffman and others 2001; Kelsey and West 1998). Structurally and spatially complex riparian vegetation provides important habitat for some species, including large and small wood on the ground, snags, multiple and diverse vegetative strata and canopy layers (cover), and complex branching patterns (Steel and others 1999).

Managers designing fuel reduction treatments need to consider the riparian features required by wildlife species of concern as well as potential conditions that might promote increases in undesirable nonnative species (Pilliod and others 2006; Strohmaier 2000; Tiedemann and others 2000; Wales 2001). Wildlife species that use riparian areas are generally divided into riparian obligates, riparian generalists, and exotic species (Kelsey and West 1998). Riparian obligates require or depend highly on riparian and aquatic resources to the extent that they are likely to be locally extirpated with loss of riparian habitat. Such species include some amphibians, reptiles, and small mammals and numerous bird species (Kelsey and West 1998). Riparian generalists utilize both riparian and upland habitats, and include some salamander species, reptiles, large and small mammals (particularly bats), and birds (Kauffman and others 2001; McComb and others 1993; Raedeke 1988). Riparian areas support nonnative animal species, such as introduced game birds, as well as undesirable exotic wildlife species, such as nutria (*Myocastor coypus*) (Hayes and Jennings 1986). The fragmentation of native riparian cover types influences the distribution of certain wildlife species, often favoring opportunistic species over those with more specific habitat requirements (Knopf and others 1988; Raedeke 1988). In some regions, breaks in riparian corridor continuity can impact animal movement (Smith 2000). Narrow corridors that are essentially edge habitat may encourage generalist species, nest parasites, and predators (Knopf 1986; Knopf and others 1988).

Research on the influence of prescribed fire, wildland fire, and forest harvest on wildlife species and habitat has shown mixed results that vary considerably for different taxa and by region (table 3) (Raedeke 1988; Smith 2000). Forest management practices primarily affect fauna in the ways that they affect habitat, including nesting, rearing, and food availability (Lyon and others 2000; Tiedemann and others 2000). Some wildlife taxa (or certain life stages of some taxa) may benefit from a particular forest management practice while others may be harmed. For example, certain mammals and birds have been shown to increase in species numbers with forest harvest, while reptiles and

Table 3. Effects of wildfire, prescribed burning and forest harvest treatments on riparian and aquatic habitat and biota.

Source	Time scale[a] (years)	Treatment and study type	Location	Findings
Pilliod and others 2003	Review of monitoring and research results	Wildland fire, prescribed fire	Range of vegetation types, largely conifer or hardwood forests; USA and Australia;	Limited research; declines in several amphibian species following wildland and prescribed fire
Bury 2004	Review of monitoring and research results	Wildland fire, various fuel treatments	Mixed conifer; Pacific Northwest forests	Limited effects on terrestrial amphibian species and riparian generalists; negative effects on riparian obligates
Bury and Corn 1988	Review of monitoring and research results	Various forest harvest practices	Mixed conifer; Pacific Northwest forests	Declines in amphibian populations following logging; severity of decline depended on species
Hicks and others 1991	Review of research results	Removal of trees from riparian areas and various upland harvest practices	Mixed conifer; Pacific Northwest forests	Negative impacts on native salmonid species; degradation of habitat and reduction in number of fish
Smith (ed.) 2000	Review of monitoring and research results	Wildland fire, prescribed fire	Range of vegetation types, largely conifer or hardwood forests; USA	Wildfire and prescribed burning affect habitat and food availability; impacts vary by species and with time since fire
Bêche and others 2005	-1 to +1 (with unburned controls)	Fall prescribed burn in riparian plots; experimental study	Mixed-conifer forest type; Sierra Nevada, CA	Periphyton biomass initially lower in the burned stream, but exceeded biomass in unburned streams within 1 year; Aquatic macroinvertebrate communities showed no detectable response
Huntzinger 2003	+1 to +10	Wildland fire, prescribed fire; experimental study	Mixed-conifer forest types; Yosemite National Park, California; Southern Oregon	More butterfly species in burned areas (wildfire and prescribed fire) relative to controls
Hawkins and others 1983	+2 to +25	Clearcut logging	Mixed conifer; Pacific Northwest forests	Initial increases in fish and salamander populations, followed by declines
Brosofske and others 1997	-2 to +2	Hillslope clearcut/ harvest	Douglas–fir dominated; West Cascades, Washington	Harvesting affected riparian microclimate gradients; increased air temperature, decreased relative humidity; riparian environments became more similar to uplands
Li and others 1994	Review of multiple management impacts on aquatic habitat	Multiple—cumulative effects	Range of vegetation types, mostly mixed conifer; Northeast Oregon	Cumulative effects of grazing, forest harvest, water diversions result in increased stream temperature, degraded fish habitat

[a] Time scale relative to treatment (year of treatment = 0)

amphibians have decreased (Raedeke 1988; Salo and Cundy 1987; Thomas and others 1979). In addition, beneficial effects of forest management on wildlife are sometimes difficult to separate from those that are detrimental (Raedeke 1988) and may change over successional time. Recent reviews have summarized the general patterns of bird responses to fire (Pilliod and others 2006; Saab and Dudley 1998; Saab and Powell 2005). Our understanding of forest harvest on wildlife species is limited, but even less is known about the potential effects of fuel reduction treatments (Bury 2004). Thinning and prescribed burning may significantly impact some wildlife by reducing the amount of down wood (cover), reducing numbers of older snags (nesting sites), and altering the plant species composition of the treated stands (cover and food) (Tiedemann and others 2000). However, fuel reduction treatments may also benefit certain species or multiple species at certain times. For example, riparian burning and thinning resulted in increased butterfly species richness and diversity along streams in the Sierra Nevada Mountains of California (table 3) (Huntzinger 2003).

As land managers proceed with fuel reduction prescriptions, wildlife habitat issues may be among the most contentious and vulnerable to litigation (Bury 2004). The presence of threatened or endangered wildlife species will likely preclude fuel reduction treatments in particular areas, including some riparian areas. However, if goals for treatments include both reduction of fire risk and the return to more historically natural conditions that support riparian habitat (Arno 1996), potential impacts to a range of wildlife species need to be evaluated. The basic life history traits and riparian habitat elements required by rare wildlife species need to be considered at different spatial and temporal scales. Wildlife species will likely respond differently to various prescriptions and successional changes following fuel reduction treatments, as has been observed with other management practices (Bury 2004; Knopf and others 1988; Pilliod and others 2006; Raedeke 1988). Although there may be short-term risks to some riparian habitat, fuel reduction treatments (and the reintroduction of fire to riparian areas) may result in a more spatially diverse range of habitat components with long-term benefits for certain species. Given limitations of current knowledge on the effects of prescribed fire and mechanical treatments on wildlife, monitoring the response of species of concern before and after fuel treatments may be essential to avoiding litigation in some locations.

Aquatic Habitat

Streambank stability: Riparian vegetation can directly affect stream channel characteristics, particularly streambank stability (Davies-Colley 1997; Gregory and Gurnell 1988; Pollen and others 2004; Simon and Collison 2002). Root systems can armor stream banks (Stokes and Mattheck 1996; Abernathy and Rutherford 2001) and bind bank sediment, thus contributing to bank stabilization, reduction of sediment inputs to streams (Dunaway and others 1994), and development and maintenance of undercut banks (Sedell and Beschta 1991). Studies have shown marked differences among riparian species and vegetation types in root characteristics and their influence on bank stability (Lyons and others 2000; Simon and Collison 2002; Wynn and others 2004). Removal of woody riparian vegetation with beneficial rooting characteristics can result in erosion of alluvial streambanks. Removal of herbaceous vegetation can decrease retention and accumulation of sediment, possibly influencing floodplain soil development (Thorne 1990). Local alterations to riparian vegetation that affect bank stability and other geomorphic processes may have effects that extend downstream.

The contribution of woody roots to streambank stabilization was modeled for forested reaches and predicted to extend approximately one-half the average crown diameter (fig. 1) (Wu 1986). Trees growing along the banks are important for maintenance of streambank stability in most locations, and we suggest that they be retained and protected during mechanical fuel reduction treatments. Prescribed fire may top kill certain riparian trees and shrubs but is unlikely to negatively affect belowground structure (Dwire and Kauffman 2003). In planning fuel reduction treatments in riparian areas, managers need to consider rooting characteristics of the plant species treated and the likely replacement species, the nature of streambank sediments, and potential impacts on streambank stability.

Aquatic foodwebs: By altering riparian vegetation, fuel reduction treatments have the potential to influence stream-riparian organic matter dynamics and aquatic trophic pathways. Autochthonous organic matter is generated through photosynthetic production by autotrophic organisms of the aquatic community (vascular plants, bryophytes, algae, bacteria, and protists) and is driven by the amount of light reaching the stream surface. Removal of riparian vegetation can result in increases in stream temperature and light, thus promoting autotrophic production (Bisson and Bilby 1998). In contrast, allochthonous organic matter originates directly from riparian or upland vegetation in the form of leaves, twigs, and other fine litter and indirectly as terrestrial invertebrates (Bisson and Bilby 1998). The input, use, retention, and transport of allochthonous organic matter in streams may drive carbon and nutrient dynamics and affect biota (Webster and Meyer 1997). For most low order streams in forested watersheds, much of the energy for aquatic food webs is derived from allochthonous inputs (Fisher and Likens 1973; Sedell and others 1978; Vannote and others 1980; Newbold and others 1982). Different plant sources vary widely in nutritional quality, and require different degrees of in-stream processing and conditioning by microbes and invertebrates (Allen 1995; Webster and Benfield 1986). In some areas, seasonal inputs of terrestrial insects from riparian areas are an important food source for drift feeding fish species (Young and others 1997). Such inputs are highest from closed canopy riparian areas dominated by deciduous plant species (Baxter and others 2004, 2005; Edwards and Huryn 1995; Nakano and others 1999). For floodplain forests, it has been suggested (FEMAT 1993) that the effectiveness of riparian vegetation in providing allochthonous inputs to streams declines at distances greater than approximately one-half a tree height away from the channel (fig. 1).

Research from studies on the impacts of fire and forest harvest on aquatic food webs have shown mixed results, depending on location, season, and species of interest (Bisson and Bilby 1998). Following a streamside prescribed fire in the Sierra Nevada, periphyton biomass was initially lower in the burned stream, but within 1 year of treatment, exceeded biomass in the unburned streams. Aquatic macroinvertebrate communities showed no detectable response to prescribed burning (table 3) (Bêche and others 2005). Significant alteration in the quality or quantity of allochthonous inputs—such as those occurring following fire (prescribed fire of wildfire) and forest harvest—has led to changes in aquatic trophic pathways that affect fish productivity (Bisson and Bilby 1998; Bisson and others 2003a; Edwards and Huryn 1996). In forested watersheds of the Pacific Northwest, the removal of riparian trees has had negative consequences for some native salmonid species (Hicks and others 1991). However, several studies have shown increases in summer biomass of fish species in headwater streams of the Pacific Northwest after logging (Bilby and Bisson 1992; Bisson and Sedell 1984). In these systems, the fish communities appear to be largely supported by autotrophic food pathways, that is, by invertebrate groups that ingest algae and algal conditioned organic matter. Increased productivity in summer populations of salmonids have also been observed following losses of riparian vegetation caused by other land uses such as livestock grazing (Chapman and Knudson 1980). This seasonal increase in fish productivity is attributed to more light reaching the stream, which stimulates autotrophic production and supports secondary production of algal dependent invertebrates (Bisson and Bilby 1998). In locations where fish bearing streams are management priorities, resource managers need to consider potential impacts of fuel reduction prescriptions on riparian vegetation that influences aquatic food webs and stream-riparian nutrient and organic matter dynamics.

Stream temperature: Fuel reduction treatments could potentially affect water temperature by altering vegetative shade that attenuates the input of solar radiation to streams. Direct sunlight warms streams, particularly during periods of low flow. During winter, lack of cover can affect stream temperature by permitting radiant cooling to the sky, potentially resulting in the formation of anchor ice (Ashton 1989). For many low order streams, riparian shading moderates these thermal fluctuations. Stream temperature has tremendous ecological importance for aquatic biota and ecosystem processes such as productivity and nutrient cycling (Allan 1995; Sweeney 1992). Water temperature strongly influences growth, development, and behavioral patterns of aquatic biota both directly and because of its influence on dissolved oxygen concentrations (Sweeney 1993). Stream temperature is an important factor determining the distribution of fish in freshwater streams, and most species of concern have limited temperature tolerances (Torgersen and others 1999).

Stream water temperature varies markedly within and among stream systems (Poole and Berman 2001). Natural drivers of water temperature include topographic shade, upland and riparian vegetation, ambient air temperature and relative humidity, altitude, latitude, discharge, water source, and solar angle and radiation (Poole and Berman

2001; Sweeney 1993). Streams in different regions and stream segments in different parts of a drainage basin vary in response and sensitivity to specific human activities that alter these drivers (Poole and Berman 2001). In addition, effectiveness of vegetation in providing stream shade varies with topography, channel size and orientation, extent of canopy cover above the channel, and vegetation structure. However, stream shading by riparian and upland vegetation is one of the few factors that can be actively managed to achieve stream temperature targets. The curve presented in figure 1 generalizes the relationship between distance from the channel and shade provided by riparian trees. In western Oregon and Washington, riparian buffer width has been designed to correlate with degree of shade (Beschta and others 1987), and riparian buffers of 100 ft or more have been reported to provide as much shade as undisturbed late successional/old growth forests (FEMAT 1993). Less is known about the effectiveness of buffer widths in providing adequate shade in other regions. In locations where particular stream temperature regimes are management goals, the short- and long-term impacts of fuel reduction treatments on shade (provided by both upland and riparian vegetation) and adequacy of buffer width need to be explicitly addressed.

Large wood dynamics: Fuel reduction treatments could potentially affect aquatic habitat by altering recruitment of large wood to streams. The role of large wood in aquatic ecosystems has become increasingly recognized over the last several decades (Bilby and Bisson 1998; Gregory and others 2003; Harmon 2002). Large wood affects geomorphic, hydrological, and ecological processes in streams and rivers, and its numerous roles link aquatic, riparian, and upland portions of watersheds (Gregory 2003; Lienkaemper and Swanson 1987). Large wood strongly influences channel form in small streams, creating pools and waterfalls and affecting channel width and depth (Montgomery and others 2003). Many species use pools formed by large wood as habitat and in-stream wood for cover (Bilby and Bisson 1998; Dolloff and Warren 2003; Wondzell and Bisson 2003). The presence of large wood in streams affects erosion, transport, and deposition of sediment, the creation and growth of gravel bars, and channel and floodplain sedimentation (Montgomery and others 2003). Dams formed by accumulations of large wood increase channel complexity and facilitate deposition of organic matter, thus providing a food source for numerous invertebrate species and contributing to nutrient cycling and retention (Bilby and Bisson 1998; Wondzell and Bisson 2003). Chronic inputs of large wood to stream channels occur as a result of bank cutting, windthrow, and mortality of individual trees from adjacent riparian areas (Bragg and Kershner 2004; McDade and others 1990). Large pulses of wood may originate from near channel sources following fire, windthrow, or insect infestations, or be transported from distant sites by debris torrents, avalanches, or landslides (Benda and others 2003; Bilby and Bisson 1998; Bragg 2000). In forested landscapes, riparian areas are important sources of large wood for streams and floodplains. However, riparian forest stands are frequently patchy, and variation from all these sources can lead to spatial variability in large wood distribution that is often not recognized in management prescriptions for a given amount of large wood per unit length of stream (Young and others 2006).

The temporal variation in large wood loads creates additional complexity. Following disturbance such as fire, contributions of large wood to channels and riparian areas can be very high in the first few decades thereafter, but the storage in each area may differ substantially. In stream channels, peaks in large wood transport may coincide with increases in contributions because of declines in stream channel stability and increases in discharge following fire, leading to rapid depletion of large wood loads during early phases of post disturbance succession. As riparian trees age, they become large enough to resist transport and breakage once they fall, and large wood loads can slowly build to pre-disturbance levels (Bragg 2000; Minshall and Brock 1991). In riparian areas, the decay of fallen trees can be surprisingly swift (Spies and others 1998; Mackensen and others 2003). In addition, recurrent fire may consume some riparian large wood (Skinner 2002) but leave pieces in the stream channel largely unaffected. Because large wood dynamics in streams and riparian areas are complex and remain poorly understood, we suggest that managers proceed with caution in altering fuel loads near streams, particularly in watersheds that have been logged.

Land use and management practices have led to marked decreases in the quantity of large wood in channels in some forested regions. Historical practices, such as removal of wood from rivers for navigation and fish passage, splash damming, tie drives, and clearing of riparian trees has resulted in simplification of stream channels and streambanks, reduction in the areal extent of riparian areas, and local decreases in amounts of large wood (Sedell and Froggatt 1984; Young and others 1994). More recent research has focused on the consequences of streamside logging (table 4). Studies conducted in forested portions of the western United States have shown marked long-term reduction in recruitment of large wood to streams in basins where forest harvest has been conducted (Lisle 2002). In western Oregon and Washington, the probability that a falling tree will enter the stream is low at distances greater than about one tree height away from the stream channel (fig. 1) (McDade and others 1990; Van Sickle and Gregory 1990). Similarly, the effectiveness of upland forests to deliver large wood to riparian areas is expected to decline at distances greater than about one tree height from the upland forested edge and depends on steepness of slope. However, timber harvest adjacent to riparian buffers eliminates large wood recruitment to the riparian area while increasing the potential for windthrow (Grizzel and Wolff 1998). In Montana, researchers also found differences in features of large wood in logged and reference streams that provide important habitat for bull trout, a federally threatened species (Hauer and others 1999). These included difference in ratios of large to small pieces of large wood, the proportion of pieces attached to the stream channel or bank, and the proportion of large wood pieces with root wads. The role of large wood is so valuable in structuring aquatic habitat that numerous efforts are underway to restore streams by adding large wood (Bisson and others 2003b; Reich and others 2003).

Table 4. Effects of fire and forest harvest on large wood (LW) inputs to streams.

Source	Time scale[a] (years)	Treatment and study type	Location	Findings
Bragg 2000	+10 to +250	Comparative simulation study of large wood inputs to streams following clear-cutting and slash removal, relative to wildfire and insect-caused mortality	Lodgepole-pine dominated, mixed-conifer, Wyoming	Overstory removal and slash burning reduced long-term large wood contributions by 50% relative to wildfire or beetle kill
Bilby and Ward 1991	+5 to +100	Retrospective sampling of near-stream areas in clearcuts, second-growth and old growth	Douglas fir dominated mixed conifer, southwest Washington	Near-stream clearcuts reduced channel large wood counts and size within 5 years of clearcut, relative to old growth
Hauer and others 1999	Not specified	Retrospective sampling of large wood in streams (3-4th order) located in unlogged wilderness, and watersheds that were logged with no buffers, and logged with buffers	Mixed conifer, Flathead Basin, northwest Montana	Marked differences between logged and reference streams in ratios of large to small pieces of wood, numbers of unattached and unattached pieces, and large wood pieces with and without root wads.
Ralph and others 1994	+3 to +40	Retrospective sampling of streams draining watersheds with unharvested old-growth forests, and intensively and moderately harvested forests	Western hemlock-Douglas fir –western red cedar forest types, western Washington	Clear reduction in size of large wood in streams, and shift in location of large wood towards channel margins in harvested basins relative to reference (old-growth) streams
Chen and others 2005	+10 to +40	Retrospective sampling of streamside areas with harvested riparian forest, burned riparian forest, and undisturbed old-growth riparian forest	Lodgepole pine dominated – mixed conifer; central Interior British Columbia	Higher volume (3X), biomass and carbon content of large wood in disturbed (wildfire or harvest) stands relative to old-growth stands

[a] Time scale relative to treatment (year of treatment = 0)

The influence of fuel treatments on large wood is a sensitive issue because of the many management actions that have reduced its abundance in stream channels. There is little ecological justification for the direct removal of large wood from riparian areas or riparian trees or snags that would create it. Prescribed fire, however, will not necessarily remove large wood from riparian areas or stream channels. Prescribed burns are typically conducted in spring or fall when fire severity is likely to be low to moderate because air temperatures are low and humidities and fuel moisture are relatively high (Knapp and others 2005). Under these conditions, large, sound boles of fallen trees do not readily ignite (especially those in and over the stream channel), although rotten pieces are consumed (Bêche and others 2005; Brown and others 2003; Stephens and Moghaddas 2005). Whereas decomposing large wood may contribute to soil formation and provide wildlife habitat in riparian areas (Chen and others 2005), only sound pieces are likely to resist breakage, promote local erosion and sediment storage, and form habitat in stream channels (Montgomery and others 2003). In addition, tree mortality caused by riparian prescribed fire is likely to contribute coarse wood in the riparian area and stream channel (Bêche and others 2005; Chen and others 2005).

Given the historical prevalence of fire in montane riparian areas (Everett and others 2003; Macdonald and others 2004), the effects of prescribed burns may emulate those of low to moderate severity wildfires that were part of the historical disturbance regime that maintained the structural and functional diversity of streams and riparian areas (Reeves and others 1995). Nevertheless, the historical interaction between fire, forest type, and large wood varied regionally (Agee 2002; Skinner 2002). It is likely that the impacts of riparian burning will also vary considerably throughout the western United States. Reports on the effects of riparian burning are few (for example, Bêche and others 2005), and we urge that these management experiments be widely shared in the literature.

Riparian Soils

Chemical, physical, and biological processes occurring within riparian soil profiles have the potential to filter, buffer, degrade, immobilize, and detoxify organic and inorganic compounds before they enter streamwater. The likely effects of upland management on down slope hydrologic and biogeochemical fluxes will impact processes that regulate nutrient, carbon, and sediment retention within riparian areas (table 5). The influence of fuel reduction treatments on compaction and productivity of upland soils are described elsewhere in this report (Chapter 9). In this section, we discuss how management of upland areas may modify riparian soil processes and contribute to their watershed effects.

The intersections of near surface hydrologic flowpaths with carbon and nutrient rich soils form "hotspots" of biogeochemical activity in riparian areas (McClain and others 2003; Wagener and others 1998). Riparian soils are frequently moist because of their lower landscape position and proximity to streams and shallow water tables. Water movement from upslope areas and hyporheic zones controls the flux of nutrients and carbon through riparian areas and regulates the soil moisture conditions that influence biogeochemical processes (Triska and others 1989). The finer textured soils found in many riparian areas have higher water holding capacity and their greater exchange capacity increases nutrient retention relative to upslope landscape positions. Especially in arid environments, increased soil moisture availability in riparian areas enhances the productivity of streamside vegetation and may support unique or more diverse plant associations as compared to upland areas (Carsey and others 2003). Root production, soil nutrient uptake and turnover, and litter production (above and belowground) also tend to be higher in streamside plant communities. In lower gradient reaches that are seasonally wet and support productive vegetation, riparian soils may be high in organic matter (Crowe and Clausnitzer 1997).

Biogeochemical processes within riparian soils regulate nitrogen transfer from terrestrial to aquatic ecosystems. Groundwater discharge represents the largest source of dissolved nitrogen delivered to forest streams (McClain and others 1998), yet plant

Table 5. Effects of prescribed burning and forest harvest treatments on soil resources and sediment movement.

Source	Time scale[a]	Treatment	Location	Comments - Findings
Covington and Sackett 1992	-1 wk to + 1 yr	Broadcast burn	Ponderosa pine, Ft. Valley Experimental Forest, near Flagstaff, Arizona	Increase in soil NH_4-N immediately after burning, followed by increase in soil NO_3
Monleon and Cromack 1996	+ 0.3, 5, 12 years	Low-intensity broadcast burn	Ponderosa pine, Central Oregon	Burning increased release of N & P from litter & reduced litter decomposition rates
Covington and others 1991	1 to 25 years	Slash pile burn	Pinyon-Juniper, Coconino NF near Flagstaff, Arizona	Increase in soil NH_4-N immediately after burning, followed by increase in soil NO_3. Each returned to preburn conditions in ~ 5 years.
Korb and others 2004	0 to 2 years	Slash pile burn	Ponderosa pine, Coconino NF near Flagstaff, Arizona	Higher soil pH, NH_4 and NO_3 and lower total C and N inside burn scars
Reuss and others 1997 Starr 2004	1 to 10 years +20 years	Clearcut	Lodgepole pine-dominated subalpine forest, Fraser Experimental Forest, central Colorado	Harvest increased soil nitrification and cation and nitrate export. Effect remains significant after 20 years
Giardina and Rhoades 2001	5 year after cutting 1 year after burning	Clearcut + slash retention; clearcut followed by surface fire	Lodgepole pine, Medicine Bow NF, S. Wyoming	Clearcuts had higher NH_4, NO_3, net mineralization, and soil moisture than uncut forest. Slash burning doubled soil N availability compared to unburned cut.
Swanson and others 1987 Brown 1983 Binkley and Brown 1993	Reviews of research and monitoring results	Clearcut	Pacific Northwest Pacific Northwest North America	Increase in suspended sediment concentrations associated with forest roads
Binkley and others 2003	1 year	Application of chipped harvest residue	Lodgepole pine-dominated subalpine forest, Fraser Experimental Forest, central Colorado	Soil NH_4, NO_3 declined beneath chips. Soil moisture increased.
Benson 1982	5 years	Application of chipped harvest residue	Lodgepole pine, Bridger Teton NF, western Wyoming	Surface runoff and soil erosion less than residue removal or undisturbed forest.

[a]Time scale relative to treatment (year of treatment = 0)

nutrient uptake and microbial transformations occurring within riparian soils can remove 90 percent of dissolved nitrate from near surface groundwater prior to its release into surface water (Gilliam 1994; Haycock and Pinay 1993; Peterjohn and Correll 1984). Attenuation of nitrate in riparian soils is attributed to a combination of plant uptake and denitrification, the microbially mediated transformation of nitrate to N_2 or N_2O gas, and subsequent loss to the atmosphere (Groffman and others 1992; Hedin and others 1998; Hill 1996). In contrast to most upland forest soils where denitrification rates are low (Groffman and others 1992), frequent saturation of riparian soils provides a redox environment that favors denitrification (Lowrance and others 1997; Peterjohn and Correll 1984; Vidon and Hill 2004). Denitrying bacteria also requires labile carbon and nitrate within the anoxic soil layer. These compounds move along groundwater flowpaths from uplands into riparian soils and denitrification releases significant amounts of nitrogen from undisturbed (Duff and Triska 1990; Hussey and others 1985) and disturbed (Davidson and Swank 1987; Rich and Myrold 2004; Waide and others 1988) watersheds.

Soil variability regulates groundwater flux and riparian biogeochemical processes. In subalpine forest watersheds, greater than 95 percent of snowmelt passes along shallow groundwater flowpaths (Troendle and Reuss 1997) and through riparian areas before entering streams. The depth and seasonal patterns of these flows determine their chemical composition and the magnitude of nutrient transformation, retention, or export (Simmons and others 1992). Soil texture and porosity control the hydraulic conductivity of near surface groundwater flow paths, exchange with labile nitrogen and carbon sources, and rates of denitrification (Hedin and others 1998; Vidon and Hill 2004). Riparian soils develop from and upon alluvial, colluvial, and aeolian parent materials and are highly variable. Fine and coarse-textured lenses and buried organic layers common to riparian soil profiles (Crowe and Clausnitzer 1997; Youngblood and others 1985) modify vertical and lateral water and nutrient movement into and through riparian areas (Groffman and others 1992; Jacinthe and others 1998; Simmons and others 1992). These sources of soil and groundwater heterogeneity form biogeochemically reactive zones imbedded within relatively inert regions. Improved ability to identify biogeochemical hotspots would help guide efforts to buffer the areas most crucial for water quality protection.

Riparian areas are vulnerable to both compaction and physical disturbance during ground harvesting operations due to areas of high moisture and low soil strength that are common within streamside zones. These concerns, along with riparian and aquatic habitat protection, provide the basis for limiting mechanical harvesting activities within streamside zones. Beyond designation of riparian buffers, land managers should consider how upland fuel reduction operations may influence nutrient and sediment retention in riparian areas and potential water quality impairment. Both vegetation removal and the actions of harvesting equipment alter site nutrient and water balances (Bormann and Likens 1979; Swank 1988). The linkages between upland management and riparian processes depend on a variety of landscape, vegetation, soil, and hydrogeologic factors that determine the flux of water, nutrients, and sediment into riparian areas, as well as specifics of the fuel management activities.

In subalpine forests of the Fraser Experimental Forest (northern Colorado), clearcutting increased snow accumulation and peak water equivalent by 36 percent and increased flow along subsurface flowpaths four-fold (Troendle and Reuss 1997). The export of nitrate from undisturbed subalpine forest hillslopes is negligible. In comparison, harvesting increases mineral nitrogen availability (table 5) (Giardina and Rhoades 2001), leaching (Fahey and Yavitt 1988; Parsons and others 1994), and groundwater flux (Reuss and others 1997; Stottlemyer and Troendle 1999). Greater subsurface water flux and nitrate concentrations may promote denitrification, if adequate labile carbon is available to fuel microbial activity (Groffman and others 1992; Simmons and others 1992). Nutrient and water uptake by riparian and residual upland vegetation will also respond to harvesting and may contribute to nutrient retention and water quality protection.

Disturbance of organic and mineral soil layers during harvesting operations can alter soil structure, infiltration, and bulk density and may lead to channelized runoff and erosion (table 5) (Binkley and Brown 1993; Brown 1983). Overland flow and sheet erosion are typically minimal in undisturbed forests, but steep slopes of many forest watersheds are susceptible to sediment transport via channelized flow (Megahan and others 1992). Loss of surface litter also increases surface runoff and decreases infiltration. Clearcutting has been shown to increase suspended sediment yield from Rocky Mountain and Pacific Northwest watersheds (Binkley and Brown 1993; Leaf 1966; Swanson and others 1987; Wondzell 2001). Similar impacts result from other ground disturbing activities such as road and fire break construction associated with fuel management activities (Wondzell 2001). The potential impact of fuel reduction treatments on the ability of riparian areas to retain sediment depends on the geomorphic setting, soil properties of the basin, and condition of the riparian vegetation.

The impact of upland prescribed burning on the capacity of riparian soils and vegetation to retain nutrients and sediment depends on fire severity and extent and distribution within a watershed (DeBano and others 1998; Fisher and Binkley 2000; Neary and others 1999). Low and moderate severity controlled burns have smaller consequences (both positive and negative) than high severity wildfires (Wondzell 2001). Fuel consumption and fireline intensity determine nutrient loss and nutrient and sediment movement following combustion. Effects can be comparable between high severity wildfire and controlled slash (Feller 1988; Giardina and others 2000) and broadcast burns (Covington and Sackett 1992; Knoepp and Swank 1993; Johnson and others 1998; Monleon and others 1997). Combustion of standing or surface fuel coupled with decreased plant uptake and fluctuating microbial activity often results in a temporary increase in soil nitrogen availability that occurs shortly after broadcast (Covington and Sackett 1992; Giardina and Rhoades 2001; Kaye and Hart 1998) and slash pile combustion (Covington and others 1991; Korb and others 2004). Elevated soil nutrient pools can lead to greater nitrate and cation leaching (Knoepp and Swank 1993; Trammell and others 2004) and in some cases, higher streamwater export (Chorover and others 1994). In uplands, high severity prescribed burns can also alter soil structure, porosity, infiltration and water repellency (Benavides-Solorio and MacDonald 2001; DeBano 2000; Robichaud 2000) and increase surface runoff and sediment movement. The effects of upland fires on the flux of nutrient and sediment into and through riparian areas may be ameliorated by residual upland or riparian vegetation and forest floor organic matter (Pannkuk and Robichaud 2003; Robichaud 2000). The processes determining the outcome of prescribed burning conducted within riparian ecosystems are likely to be similar, though we are not aware of comparable published results for streamside areas.

Mechanical chipping and mastication operations are being widely prescribed to treat hazardous fuel, yet the implications of these practices on riparian and watershed conditions are largely unknown. As compared to typical harvesting operations and unharvested stands, these fuel management prescriptions rearrange the amount, size, and orientation of surface woody materials. Similar to other upland management activities, these mechanical treatments are likely to influence soil processes and nutrient retention within riparian areas. A recent review of published findings relating to woody debris additions reported that implementation of chipping and mastication treatments varies considerably among sites depending on equipment and operational differences (Resh and others 2006). The influence of the treatments on soil properties varied as well, although some generalizations emerged. Soil carbon and moisture increased following the mechanical fuel reduction operations. Maximum soil temperature and understory vegetation declined. Woody debris additions had variable effects on soil nutrients. In some cases, soil nitrogen availability decreased as carbon rich woody material stimulated microbial nitrogen immobilization (Binkley and others 2003; Blumfield and Xu 2003; Lalande and others 1998). For example, in Colorado lodgepole pine stands, addition of wood chips reduced soil nitrogen availability by ~ 65 percent (Binkley and others 2003). There is some evidence that logging residue and chip additions may depress sediment movement (Benson 1982). The potential for upland chipping or mastication to significantly alter nutrient and sediment movement into riparian areas partly depends on the horizontal continuity and depth of woody material additions. To date, there are no completed studies that directly assess the linkages between these new mechanical fuel management strategies and riparian processes or watershed conditions.

Cumulative Watershed Effects of Fuel Management Activities on Riparian Areas

Most land management activities contribute to cumulative watershed effects, and fuel reduction treatments being conducted anywhere within a watershed could potentially influence riparian functions. Because stream-riparian corridors are located at the lowest point within drainage basins, they can act as integrators of entire watersheds and may be particularly vulnerable to effects of fuel reduction treatments conducted upslope and upstream. In the past, undesirable changes in riparian areas have resulted from the failure to recognize linkages among streams, riparian areas, and uplands. To minimize negative cumulative effects on riparian functions, integrative planning and assessment of all management activities, including fuel reduction treatments, grazing, forest harvest, and recreation, should occur at both watershed and larger landscape scales. The management of wildfires, including location of back burns, cutting lines and natural ignitions that have been allowed to burn, may have very direct and cumulative effects. Wildland fire use (managed wildfire) is increasingly common, and portions of the landscape that have been allowed to burn are likely to be much larger than areas treated with prescribed fire and mechanical fuel removal. The management history, hydrologic and sediment regimes, and the role of natural disturbance all need to be considered in planning fuel treatments (Chapter 12).

The watershed approach of assessing additive and interactive impacts on riparian areas is conceptually simple. However, the actual evaluation of cumulative watershed effects on riparian functions is technically difficult because of limited knowledge of many biological and physical processes, interactions among processes for which impacts may accumulate through space and time, and time lags in the expression of effects. Another limitation is the lack of reference conditions and limited baseline data for comparison and evaluation of environmental changes, including measures of impact severity (Reid 1998). Despite these difficulties, an analysis of cumulative watershed effects on riparian functions must evaluate potentially important impacts on valued riparian functions, downstream impacts, and impacts accumulating through space and time within the watershed using the best available analysis methods and information. Relative risks of severe wildfire versus impacts of fuel treatments need to be weighed (O'Laughlin 2005). In addition, information needs should be acknowledged and monitoring goals clearly identified (REO 1995).

Analysis methods for assessment of cumulative effects have been developed to fulfill requirements of the National Environmental Policy Act and require a non-traditional approach to information (Reid 1998) (Chapter 14). The ecosystem analysis method used on federal lands in much of the Pacific Northwest provides a model that integrates background information about ecosystem and landscape interactions that can be used for later cumulative effects assessments during project planning (FEMAT 1993; Reid 1998). In addition, landscape analysis tools, such as Landscape Management System (LMS), have been developed to assist in planning of fuel treatments (McCarter and others 1998; Peterson and others 2003). These tools display spatial patterns of forest structure and fuel across a landscape for current conditions and compare them to patterns produced by various fuel treatment scenarios. When possible, spatial delineation of riparian areas should be incorporated into landscape level planning of fuel treatments and may contribute to effects analysis as well as integrate riparian protection and management into fuel reduction programs.

Management Implications

1. **Riparian areas are spatially diverse and variously defined. Attention to ecological context within a drainage basin and the larger landscape is critical, as is the connectivity between upslope and upstream management and condition of streams and riparian areas.** Impacts of fuel treatment activities will vary depending on their locations within a watershed, the natural fire regimes, and the fire and management history of the treated basins. Stream-riparian corridors are dynamic and planning should allow for continuous change, including successional processes and natural disturbance.
2. **During planning and implementation of fuel reduction treatments, consideration of potential impacts on key riparian functions is essential to minimize local and immediate effects as well as cumulative, longer term effects.** Riparian areas provide valued ecological functions (table 1) that have been altered by land management (tables 2-5) (Hessburg and Agee 2003). Local and regional issues will dictate which riparian functions are priorities for management goals and critical for protection.
3. **Riparian buffers and BMPs may not protect all riparian functions during fuel reduction treatments.** Although BMPs and the establishment of riparian buffers have mitigated the effects of forest harvest activities on stream water temperature and

quality, current BMPs may not be effective in protecting all valued riparian functions, particularly in watersheds that have been recently burned by severe wildfire. With multiple fuel reduction treatments (including consecutive entries over time), BMP effectiveness may vary and additional conservation practices may be necessary.

4. **Objectives for fuel reduction treatments should include the return to fuel loads that support ecosystem processes and natural disturbance regimes and incorporate short- and long-term targets for the vegetation condition of uplands and riparian areas (Rieman and others 2003).** Using concepts such as natural or historical range of variability (Landres and others 1999), reference areas, and desired future condition, the planning and implementation of fuel reduction treatments may be regarded as opportunities to restore certain ecological conditions, especially in riparian areas (Arno 1996). Fire managers are frequently able to implement fairly exact prescriptions, such as reducing certain fuelbeds while retaining others. Restoration objectives, in addition to emphasis on fuel management, are encouraged. In addition, follow up monitoring for achievement of project objectives (short- and long-term) is critical to expand our knowledge of fuel management.
5. **Current knowledge on the effects of fuel reduction treatments on riparian areas is very limited. Potential environmental effects need to be assessed in a landscape context that includes relative influences of all management influences, including past fire suppression and current wildland fire use.** We have summarized potential effects of fuel reduction treatments on riparian vegetation, the provision of terrestrial and aquatic habitat (contributions to streambank stability, aquatic food webs, maintenance of stream temperature, and large wood dynamics), and the filtering and sediment retention capacity of riparian areas (tables 2-5). We emphasize that very little is actually known about either specific or cumulative impacts of specific fuel reduction treatments. Our understanding of cumulative watershed effects is largely derived from a handful of watershed scale studies of past practices (Reid 1998). Results from these studies are quite variable and confounded by local effects of other past and current management activities (Wondzell 2001). Much of the site level research cited in this chapter was conducted in the Pacific Northwest and may be difficult to extrapolate to other regions in the western United States. Explicit recognition of uncertainty is encouraged during planning and implementation of treatments, with an adaptive management approach to changing direction if the desired outcome is not achieved. We regard each individual fuel reduction project as an experiment and emphasize the need for monitoring to track the impacts of prescribed burning, tree removal, chipping and mastication, and salvage logging.
6. **Research is needed to address the impacts of fuel treatments on watershed processes, riparian functions, and aquatic resources.** Although studies are underway (http://jfsp.nifc.gov/JFSP_Project_Info.htm), we have noted numerous research needs throughout this review, as well as regions with limited data (for example, many Rocky Mountain ecosystems). Fuel reduction treatments are highly variable, and each treatment, sequence, or combination of treatments may have significantly different environmental effects. For assessment of cumulative effects, research on impacts of multiple stage projects is key, particularly in relation to other active management, including that of wildland fire.

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Chapter 11.

Biological Responses to Stressors in Aquatic Ecosystems in Western North America: Cumulative Watershed Effects of Fuel Treatments, Wildfire, and Post-Fire Remediation

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Introduction

Aquatic communities in North America evolved with disturbance regimes (for example, glaciation, erosion, tectonics, volcanism, fire) that varied in frequency, intensity, and severity (Poff 1992; Power and others 1988; Resh and others 1988). Disturbance has been recognized for its importance in the organization and maintenance of aquatic ecosystems (Reeves and others 1995), in shaping species resilience and persistence (Reice and others 1990; Yount and Niemi 1990), and structuring the evolution of aquatic organisms (Lake 2000; Malanson 1984; Schlosser 1990; Sedell and others 1990). Disturbances are regarded as having a dominant role in determining the structure of stream communities (Palmer and others 1996; Resh and others 1988).

Natural and anthropogenic disturbance regimes are classified as pulse, press, or ramp types that vary in their temporal and spatial patterns of impact intensity and duration (Lake 2000). Pulse disturbances are short-term and often intense events, whereas press types may arise sharply but reach a constant level that is maintained. Ramp disturbances arise if the intensity increases over time. In most situations, discrete disturbance regimes are unlikely to occur and the system is more likely to be characterized by a mix of disturbance types. For example, the effects of a press disturbance, such as land use change, may interact with invasions by exotic species (ramp) and this regime may be affected by pulse disturbances in the form of floods that occur unpredictably and with varying intensity. However, the life history traits of the longest-lived affected population may determine the importance of disturbance in shaping aquatic ecosystem structure and function (Benda and others 2003; Bisson and others 1997; Reeves and others 1995; 1998; Rieman and others 2003).

Catastrophic shifts in ecosystems are often precipitated by stochastic (pulse) events such as fire, but the foundation for them lies in the gradual (press-ramp), underlying loss of resilience (fragmentation, habitat loss, etc). The real impact of a press disturbance on population persistence may only be evident after a pulse disturbance and over relatively long time scales. Land use changes constitute a press disturbance that may have a persistent effect on stream community structure and function over the long-term (Harding and others 1998). Press disturbances, such as timber harvesting, require longer recovery times than pulse disturbances such as floods or droughts (Detenbeck and others 1992). When forest management practices affect the stability of in-stream habitats, frequent pulse disturbance could result in the local extirpation of refugia-dependent species (Parkyn and Collier 2004). Short-term responses in abundance or survival of organisms or the physical attributes of streams to cumulative effects are not easily detected because suitable habitat features in disturbed landscapes may not provide stable refugia during pulse disturbances. These effects may only become evident over long time scales. Thus, it may also be important to consider the biological mechanisms (for example, survival, growth, dispersal) influencing the responses at different scales (Harvey and others 2006; May and Lee 2004). Maintaining a stability domain (sensu Scheffer and others 2001) that allows ecosystems to resist disturbance and recover will present significant challenges to resource managers and might be considered in terms of our ability to conserve resilient populations, evolutionary legacies, and ecological functions.

Disturbance responses depend on the ability of aquatic organisms to find refugia in space and time in heterogeneity of their physical habitat, diversity of life history strategies, behavioral adaptations, or connectivity to refugia at various spatial scales (Angermeier 1995; Poff and Allan 1995; Rieman and others 2000). In the southeastern United States, extinction prone aquatic species are typified by a limited geographic range, restricted range of habitat sizes, and highly specialized habits (Angermeier 1995; Jelks and others 2008; Taylor and others 1996; Warren and others 2000). These associations reflected the effects of reduced habitat area and increased isolation (insularization), which are also important determinants of extinction in terrestrial systems (Angermeier 1995).

Lotic systems, like most natural ecosystems, exhibit extreme heterogeneity in environmental conditions and biotic communities at multiple spatial scales, ranging from microhabitats to whole landscapes and ecoregions (Montgomery 1999; Pringle and others 1988; Townsend 1989; Townsend and others 2003; Vannote and others 1980). At each scale, variation in environmental conditions affects stream biota (Angermeier and Winston 1998; Lake 2000; Poff 1997; Walters and others 2003), leading to high variability in abundance and community structure at multiple spatial scales (Downes and others 1993; Li and others 2001; Vinson and Hawkins 1998). Vegetation, geology, hydrology, and land use generally account for differences in community structure among river systems (Richards and others 1997; Townsend and others 1997, 2003) but variation in species composition is related mainly to local and in-stream factors (Angermeier and Winston 1998; Gregory and others 1991; Li and others 2001; Walters and others 2003).

Fire is widely recognized as one of the most important disturbance processes in the western United States (Hessburg and Agee 2003). Fire regimes in western North America exert strong influences on patterns of forest composition, structure, and successional dynamics (Agee 1991; Halpern and Spies 1995) (Chapter 2). During the past century, fire suppression has altered fire regimes in some vegetation types, and consequently, the probability of large stand replacing fires has increased in those areas. Retrospective studies indicate that fire return intervals are longer than prior to European-American settlement as fire suppression, landscape conversion, and fragmentation altered fire regimes in much of the western United States (Agee 2003; Hessburg and Agee 2003). Fire regimes in riparian areas vary by region and forest type (Chapter 2). In wetter forests, riparian fire regimes were more moderate, fire return intervals were longer, and fire intensity and severity was lower than in adjacent upland forests (Dwire and Kauffman 2003). In drier forest types, fire regimes in riparian and upland forests were more similar. Local conditions, governed by microclimate, geomorphology, soils, and elevation in riparian areas, contribute to the variability of fire regimes and fire severity in riparian areas (Dwire and Kauffman 2003; Whitlock and others 2003). Land management practices, particularly in the western United States, usually conjure landscape degradation associated with decreases in vegetation density and cover. The indirect effects of selective and extensive timber harvest, fire suppression, and grazing practices have significantly altered forest structure and fuel loads. Forests that were once mosaics of species, ages, and patterns have been simplified; the area becoming dominated by higher density, middle aged stands, thereby reducing their resistance to the effects of severe wildfire (Hessburg and Agee 2003; Rieman and Clayton 1997; Rieman and others 1997).

The National Environmental Policy Act of 1969, Clean Water Act of 1972, and Endangered Species Act of 1973 required greater consideration of the relationships among ecosystem components and increased the focus on the effects of land use activities on fish and wildlife resources, especially threatened species (Rinne and Medina 1995). Fuel reduction and fire suppression are key elements of the National Fire Plan, which provides guidance for an interagency approach to fire and fire related management (USDA 2000). In implementing the management goals of the Healthy Forest Restoration Act of 2003 (H.R. 1904), it will be important for resource managers to understand the effects of fire related management on aquatic and terrestrial ecosystems and the tradeoffs between achieving fuel reduction and minimizing the effects of those efforts on aquatic and riparian ecosystems. In this paper, we attempt to summarize the effects of wildfire and fuel treatment practices on aquatic resources, identify areas of uncertainty about the impacts of fuel reduction strategies on the terrestrial-aquatic ecotone, and highlight research needs to better inform resource management.

Disturbance, Fire, and Land Management

The physical and biological effects of wildfire and forest harvest on aquatic ecosystems are not identical. Numerous studies comparing fire and logging disturbance regimes confirm the dominant role of the watershed in tempering the aquatic responses to terrestrial disturbance and reflect both important similarities and differences in the responses to disturbance within the catchment (Carignan and others 2000; Smith and others 2003). Both fire and logging exhibit similar stressor characteristics (temporary deforestation, sediment and nutrient influx, increased stream temperatures) but have important differences in spatial extent, frequency, intensity, and severity of disturbance with respect to water quality. Abiotic and biotic elements of a watershed interact to govern the direction and magnitude of change in response to disturbance. Habitat alteration probably has the greatest impact on individual organisms and local populations that are the least mobile, and reinvasion will be most rapid by aquatic organisms with high mobility. Perturbation associated with hydrological processes is probably the primary factor influencing post-disturbance persistence of fishes, benthic macroinvertebrates, and diatoms in fluvial systems. These effects may be immediate and direct or occur indirectly over long periods of time (Dunham and others 2003; Gresswell 1999; Minshall and others 1997). Many aquatic organisms (such as salmonids) have evolved strategies to survive perturbations occurring at the frequency of wildland fires (greater than 100 years), but this does not prevent extirpations of local populations (Gresswell 1999; Rinne 1996).

Physical Effects of Fire and Forest Management Practices

Upland and riparian vegetation interacts with soils, geology, and topology to influence channel form, habitat, flow characteristics, and nutrients (Bisson and others 1987; Gregory and others 1991; Montgomery 1999). Where vegetation cover has been disturbed by fire or human activity, terrestrial habitats may experience increased surface flows that cause rapid, concentrated surface runoff, accelerated erosion, and increased stream peak flows and sediment loads (DeBano and Schmidt 1989; DeBano and others 1996). Small streams are typically more affected by catchment disturbance than larger streams and, as adjacent slopes become steeper, the likelihood of disturbance from instream effects increases (Lee and others 1997; Megahan and Ketcheson 1996). The physical effects of fire on aquatic systems depend on the severity of the fire, post-fire climate, development of soil repellency, and alteration of the hydrologic regime (Benda and others 2003; Beschta and others 1987; DeBano and others 1996; Pierson and others 2001; Tiedemann and others 1979; Wondzell and King 2003). Short-term consequences of fire include mobilization of nutrients, soil modification, increased water temperature (including direct heating), and changes in hydrology, water chemistry, and geomorphology (Bisson and others 2003; Bozek and Young 1994; Gresswell 1999; Minshall and others 1989). Timber harvest activities may accelerate surface erosion and increase sedimentation (Chamberlin and others 1991; FEMAT 1993; MacDonald and others 1991; Meehan 1991; Reid 1993; Rhodes and others 1994).

Erosion can be initiated or accelerated by timber harvest and fire management activities including the construction of fire lines, roads, and post-fire rehabilitation efforts (Landsberg and Tiedemann 2000; Robichaud and others 2000). Tractors and skidders promote soil compaction and may temporarily increase the potential for overland flow but reduce interstitial flow through the soil, which may require decades to recover (Backer and others 2004; Hartman and others 1996) (Whitson and others 2003; Williamson and Neilsen 2000) (Chapter 9). The use of heavy equipment in timber management or restoration may disturb soil integrity, increasing erosion (McIver and Starr 2000; Robichaud and others 2000). Burned Area Emergency Restoration (BAER) treatments, including felling trees and snags for erosion control, may not reduce storm runoff, erosion, or sedimentation and may cause additional disturbance (McIver and Starr 2000; Robichaud and others 2000). The cumulative effects of timber harvest and associated road building on aquatic life is exacerbated in forest catchments on steeper, more environmentally sensitive terrain (Gucinski and others 2001; Megahan and Hornbeck 2000; Platts and Megahan 1975).

Increased fine sediment delivery to streams is usually associated with timber harvesting and road construction (Platts and Megahan 1975; Eaglin and Hubert 1993). The majority of sediment from timber harvest activities is related to roads and road construction (Chamberlin and others 1991; Furniss and others 1991; Megahan and Hornbeck 2000) and associated increased erosion rates (Beschta 1978; Meehan 1991; Reid 1993; Rhodes and others 1994; Swanson and Dyrness 1975; Swanston and Swanson 1976). Serious degradation of aquatic habitat can result from poorly designed, constructed, or maintained roads (Furniss and others 1991; MacDonald and others 1991; Rhodes and others 1994). Road networks interact with stream networks and lake basins at the landscape scale and affect biological and ecological processes in stream and riparian systems. Peak flows and debris flows are influenced by the arrangement of the road network relative to the stream network (Swanson and others 1988; Thompson and Lee 2000, 2002) and they can alter the balance between the intensity of flood peaks and the stream network's resistance to change (Jones and others 2000). Timber harvest activities caused increases in lake sedimentation rate and lake productivity in three of four lakes studied in western Washington, accelerating the rate of change in the trophic status of each lake (Birch and others 1980).

Water Quality

The modification of terrestrial habitats is often accompanied by changes in water quality and quantity (Bjornn and Reiser 1991; Chamberlin and others 1991; Rhodes and others 1994; Chapter 8) and perturbation of nutrient cycles within aquatic ecosystems (Megahan and Hornbeck 2000; Spencer and others 2003). Timber harvest often temporarily increases water yield and alters the timing of the flow, usually seen as an increase in low flows. The magnitude and timing of storm and peak flows depend on the degree of soil loss, soil compaction or hydrophobicity, and the pattern and schedule of harvest. The effects, which vary depending on the prevailing climate, rate of vegetation recovery, and type of vegetation, can decrease after initial disturbance but may remain above natural levels for many years (Bolstadt and Swank 1997; Hartman and others 1996; Platts and Megahan 1975; Swanson 1981; Chapter 7). As vegetation recovers, dry season minimum flows may be reduced because of enhanced evapotranspiration (Hicks and others 1991).

Water quality can be altered by timber harvest activities (Bolstad and Swank 1997; Chamberlin and others 1991; Chapter 8). Stream temperature may rise following loss of streamside shading, disrupted subsurface flows, reduced stream flows, elevated sediments, and morphological shifts toward wider and shallower channels with fewer deep pools (Beschta and others 1987; Chamberlin and others 1991; MacDonald and others 1991; Reid 1993; Rhodes and others 1994; Swank and others 1989). Logging may alter temperature as riparian shading is reduced (Hartman and others 1996; Johnson 2004; Johnson and Jones 2000; Swank and others 1989). Increased solar input into streams may also change the seasonal thermal regimes causing shifts toward earlier and longer summers when solar input is at its highest (Hartman and others 1996; Johnson and Jones 2000; Poole and Berman 2001). Increased stream temperatures can exceed thermal preferences of many species in the PNW and may remain elevated for up to 15 years after timber harvest (Johnson and Jones 2000). Dissolved oxygen can be reduced by low stream flows, elevated temperatures, and increased fine inorganic and organic materials that have infiltrated into stream gravels (Chamberlin and others 1991; Wondzell and King 2003).

Nutrients

Stream concentrations of particulate organic matter, phosphorus, nitrogen, and ions can increase following forest fires (Bayley and others 1992; Earl and Blinn 2003; Spencer and Hauer 1991) and clearcutting (Feller and Kimmins 1984; Hartman and others 1996; Likens and others 1970; Webster and others 1990). Nutrient concentrations may increase following logging but generally return quickly to normal levels (Chamberlin and others 1991). Increases in total phosphorus and nitrogen concentrations, dissolved organic carbon (DOC), and other trace nutrients appear to be driven by higher discharge, which enhances nutrient loading from the watershed and (or) from the stream channel (Carignan and others 2000; Prepas and others 2003; Spencer and others 2003). During subsequent years, nutrient concentrations may periodically increase in fire impacted sites during spring run-off. Post-fire changes propagated through the aquatic food web suggest shifts from reliance on terrestrial and other allochthonous sources to a periphyton-based food web following canopy removal and nutrient enrichment. Chlorophyll *a* concentrations increase, presumably in response to nutrient run-off and increased light levels (Allen and others 2003). Some chemical effects on water quality diminish with time, but nutrient levels may remain unchanged or continue to increase, suggesting that longer time frames may be necessary for increased nutrient input to aquatic ecosystems to return to normal levels (Carignan and others 2000). As a result, biotic responses to alterations of water quality may take longer to manifest themselves (Brass and others 1996). Paleoecological studies of lake sediments revealed fundamental changes in lake mixing regimes following logging with associated alterations of hypolimnetic trophic structure persisting for over 100 years (Scully and others 2000). Experimental clearcut logging in some lake catchments produced no significant changes in water quality (dissolved organic carbon, chlorophyll *a*, total nitrogen increased; Ca^{2+} and Mg^{2+} decreased), though water clarity decreased (Steedman 2000; Steedman and Kushneriuk 2000). The responses, however, varied regionally. In some cases, forest harvesting was associated with increases in nutrient and chlorophyll *a* concentrations, cyanobacteria, and net primary production (Prepas and others 2001; Rask and others 1998). In other cases, differences in nutrient dynamics between logged watershed lakes and reference lakes were undetectable (Steedman 2000).

The benefits of the type of fuel management applied (logging, thinning, prescribed fire) depend on the frequency and intensity of their application. Presently, the influence of disturbance type (logging or fire), time elapsed since the disturbance, and extent of natural variation limits our ability to detect the direct effects of fire management activities. Shorter rotations associated with frequent treatments may reduce soil fertility (Chapter 9). Low intensity prescribed fires post-harvest may increase short-term productivity but soil nitrogen may be volatilized and lost to the catchment when fire is used as a fuel reduction tool. Roads contribute more sediment to streams than any other land management activity (Gucinski and others 2001; Meehan 1991), but land management activities designed to combat wildfires or restore forest conditions depend on roads. Natural and anthropogenic disturbances that reduce terrestrial litter inputs to streams alter aquatic ecosystem function, highlighting the importance of riparian ecotones in sustaining diverse aquatic food webs (England and Rosemond 2004; Wallace and others 1997). Reduced stream shading, accompanied by increased nutrient input, may contribute to algal blooms.

Habitat

The supply of large woody debris to stream channels is typically a function of the size and number of trees in riparian areas, and thus can be profoundly altered by timber harvest (Bisson and others 1987; Naiman and others 2000; Robison and Beschta 1990; Wood-Smith and Buffington 1996). Shifts in the composition and size of trees within the riparian area affect the recruitment potential and longevity of large woody debris within the stream channel. Large woody debris influences channel morphology, especially in forming pools, slow water refuges, and instream cover, and provides shade, retention of nutrients, and storage and buffering of sediment (Robison and Beschta 1990; Rhodes and others 1994). In intensively logged watersheds, the size of instream woody debris was smaller and pool depths significantly shallower than those found in a relatively undisturbed watershed (Overton and others 1993; Ralph and others 1994). All of these changes can eventually culminate in the loss of biodiversity within a watershed (Hauer and others 1999; Hawkins and others 1997).

Biotic Responses to Watershed Disturbance

Macroinvertebrates

Changes in aquatic invertebrate assemblages have been used for many decades to monitor impacts on land and in water. Land and water uses and impacts are reflected in species assemblages in streams and lakes. However, the level of detail of most monitoring is not sufficient to track species losses in aquatic invertebrates. Following watershed disturbance, the biomass of stream macroinvertebrate communities has been shown to increase in some cases (Burton and Ulrich 1994; Haggerty and others 2004; Stone and Wallace 1998) but not in others (Minshall and others 1997). Minshall (2003) described general patterns of responses of stream macroinvertebrate assemblages to fire, finding only minor or no direct effects except in extreme cases of intense heating of the stream from severe wildfire. Short-term, post-fire alterations of assemblage structure and decreases in biomass may be associated with changes in riparian vegetation. Most impacts were indirect effects on species composition and food web processes that stemmed from altered runoff and channel morphology (Minshall and others 2001). Following severe wildfire, macroinvertebrate assemblages showed low resistance to spate-induced debris flows, and high resilience but low assemblage similarity to pre-fire assemblage structure with the post-fire assemblage dominated by generalist taxa (Mihuc and Minshall 1995; Vieira and others 2004). In comparative studies of the consequences of timber harvest and fire on lake aquatic communities, responses of macroinvertebrate densities varied. Generally, invertebrate biomass was greater in lakes from burned catchments than from harvested and reference catchments. Scrimgeour and others (2001) found post-wildfire nutrient enrichment prompted significant increases in total benthic macroinvertebrate biomass in recently burned lake catchments. These analyses suggest that benthic biomasses continue to be elevated for about 15 to 20 years following fire before declining to pre-disturbance levels. Under the influence of phosphorus, nitrogen, and chlorophyll *a*, biomass response depended on percent of catchment disturbed (Scrimgeour and others 2000). In some cases, forest harvesting was associated with increases in chlorophyll *a* concentrations and primary production causing moderate increases in zooplankton density (Rask and others 1998). Biomass increased in burned watershed lakes but declined in logged catchments (Patoine and others 2000; Prepas and others 2001). Changes in a food source of such importance as aquatic invertebrates can have repercussions in many parts of the food web (Hernandez and others 2005; Spencer and others 2003).

Fish

As with other aquatic organisms, the short-term direct effects of high severity fires on fish populations may be minimal (Spina and Tormey 2000) or intense (Bozek and Young 1994). Intense fires and their subsequent effects on hydrologic regimes, erosion, debris flows, woody debris recruitment, and riparian cover can strongly influence the structure and function of aquatic systems (Rieman and others 2003; Swanson 1981). Post-fire effects may result in fish mortality (Bozek and Young 1994; Minshall and others 1997; Rieman and Clayton 1997; Rinne 1996; Spencer and others 2003), but local extirpation of fishes is patchy, and recolonization is often rapid (Bisson and others 2003; Gresswell 1999; Rieman and others 1997, 2003). Subsequent effects associated with the loss of vegetation and infiltration capacity of soils may include increased erosion, changes in the timing and amount of runoff, elevated stream temperatures, and changes in the structure of stream channels (Benda and others 2003; Rinne and Neary 1996, Swanson 1981; Wondzell and King 2003). The nature of these changes depends on the extent, continuity and severity of the fire, and on lithology, landform, and local climate (Rieman and Clayton 1997; Swanson and others 1987). Where native fish populations are naturally depauperate or have declined and become increasingly isolated because of anthropogenic activities, the effects on fish populations are more pervasive and long lasting (Propst and others 1992; Rieman and others 2000; Rinne 1996; Rinne and Minckley 1991). Ecological specialists—species with narrow habitat requirements—in highly degraded and fragmented systems are likely to be most vulnerable to disturbance (Angermeier 1995; Angermeier and Winston 1998; Dunham and others 2003; Trebitz and others 2003). The role of fire in facilitating invasions by non-indigenous aquatic organisms is unknown, but must be considered among the indirect effects. Disturbance associated with fire may make already vulnerable native populations susceptible to colonization by invasive species (Dunham and others 2003). Aquatic systems may not experience shifts in assemblage structure, but may exhibit effects at the population level that translate into long-term effects (St.-Onge and Magnan 2000; Tonn and others 2003, 2004). This illustrates the importance of post-fire monitoring and research at more relevant spatial and temporal scales to better understand the natural response regime (Gresswell 1999).

The impacts of timber harvest on fish assemblages are extensions of their effects on aquatic ecosystems (Rinne and Neary 2000; Rinne and Stefferud 1999). The extent to which hillslope and riparian soils are disturbed and mobilized to the stream channel will reduce pool habitats (McIntosh and others 1994, 2000), decrease the survival of incubating salmonid eggs (Reiser and White 1990), and/or increase turbidity (Bolstadt and Swank 1997; Lisle and Napolitano 1998). A change in timing or peak flow magnitudes may scour redds or embed them. Removal of timber from riparian areas decreases the amount of large woody debris available for recruitment into the channel, affecting pool formation, sediment storage, and cover availability (Beechie and Sibley 1997; Bilby and Ward 1991; Bisson and others 1992; Solazzi and others 2000; Reeves and others 2002), though the extent of management activity in the catchment may determine the severity of the impacts (Reeves and others 1993). Short-term increases in recruitment of large woody debris in the channel may benefit fish and amphibian populations but the benefits may be transient as the long-term supply of large trees in the riparian zone is depleted. Thinning of the riparian canopy increases solar input, stream temperatures, and aquatic productivity. Fish populations may exhibit lower abundance or productivity resulting from reduced coldwater habitats or increased stream temperatures over time spans of decades (Hornbeck and Kochenderfer 2000; VanDusen and others 2005).

Vegetation removal, whether by fire, logging, or other human activity, acts at multiple scales to affect the natural processes that control catchments and stream processes, destabilizing flow and thermal regimes, simplifying habitats through siltation, reducing water quality, and impacting biotic communities (Osborne and Wiley 1988; Rabeni and Smale 1995; Snyder and others 2003; Wang and others 1997). Patterns in fish assemblage structure are often attributed to longitudinal changes in stream attributes (Jackson and others 2001; Jones and others 1999; Lyons 1996; Rahel and Hubert 1991; Schlosser 1982) and hydrologic variability (Poff and Allan 1995). Changes in richness, density, and species composition are associated with decreased hydrologic disturbance (in other words, more stable flows), increases in pool depth and habitat diversity, and local geomorphic conditions and processes that contribute to spatial heterogeneity within the stream continuum (Poff and Allan 1995; Schlosser 1990) and may serve to mask responses to anthropogenic disturbance (McCormick and others 2000).

Changing fire regimes and the potential for larger, more destructive fires may threaten the loss of aquatic habitat diversity and lead to accelerated extinction of some vulnerable populations (Dunham and others 2003; Gresswell 1999; Minshall 2003; Rieman and Clayton 1997; Rieman and others 2003). A simple focus on managing fuel, however, may not address the role of fire or the primary threats to the persistence of many species. An approach that aims to restore resiliency to fire related disturbances in both terrestrial and aquatic ecosystems is needed. Development of predictive models relating physical attributes of streams and fish population abundance have shown promise in identifying the high priority reaches for restoration (Barnett and others 2003; Latterell and others 2003; Wu and others 2000). This suggests that similar models might be developed that would identify stream reaches that would be high priority areas for exclusion of treatments.

Amphibians and Reptiles

Amphibians are the most abundant vertebrates in many forests and have the potential to play a significant role in ecosystem dynamics (Wahbe and Bunnell 2003). Riparian and upland habitats provide important habitat for amphibians and semi-aquatic reptiles that depend on mesic ecotones to forage, aestivate, and reproduce (Burke and Gibbons 1995; Castelle and others 1994; Semlitsch 1998). Terrestrial amphibian species richness, abundance, and community composition are highly dependent on catchment forest cover (Houlahan and Findlay 2003; Wahbe and Bunnell 2003). Intensive harvest completely eliminated terrestrial salamanders or reduced them to very low numbers when mature forests were clearcut (Knapp and others 2003; Naughton and others 2000; Perison and others 1997; Petranka and others 1994) though species may persist where large amounts of woody debris are left on the forest floor (Biek and others 2002). At larger catchment scales (400 to 4,000 ha), responses of amphibian and reptile species to management practices (including clearcutting) varied considerably depending on the taxon (Loehle and others 2005; Renken and others 2004). Post-harvest declines in local amphibian populations persist for years post-harvest but abundance and species composition may remain relatively stable at larger spatial scales (Babbitt and Tanner 2000; Herbeck and Larsen 1999). Selective harvests that create small gaps may have little or no effect on amphibians (MacCracken 2005; Messere and Ducey 1998). Reductions in litter depth and coarse woody debris contribute to declines in some amphibian populations because of structural changes to forest floor habitats (DeGraaf and Rudis 1990; Houlahan and Findlay 2003; Loehle and others 2005; Mitchell and others 1997), but studies of forest structure in the Pacific Northwest found no evidence that variation in amphibian abundances was strongly influenced by the amount of coarse woody debris on the forest floor (Aubry 2000; Bunnell and others 1999). Abundance of stream dwelling amphibians is also reduced following timber harvest (Adams and Bury 2002; Bury and others 2002; Corn and Bury 1989).

The vulnerability of amphibian populations to wildfire varies by region, species' life histories, and fire regimes. The coincidence of fire and migration, reproduction, and larval periods may determine the vulnerability of amphibian populations to wildfire. Studies suggest that direct fire related mortality of adult amphibians is rare, either because of the timing of the fire or because amphibian species were able to exploit refugia from fire (for example, burrows, moist ground, ponds, streams; see papers summarized in Pilliod and others 2003). Increases in stream temperature, because of canopy loss, will affect species that exhibit strong thermal preferences (Pilliod and others 2003; Welsh and others 2001; Welsh and Lind 2002). Sedimentation in streams results in reductions in interstitial spaces that can affect reproductive success in amphibian populations, depress growth rates from lost foraging space, and expose individuals to increased predation. Fire may alter conditions along migratory corridors, from terrestrial habitats to the aquatic systems where amphibians breed. Preliminary surveys of post-fire amphibian populations in the Klamath-Siskiyou Region suggest no negative effects of wildfire on terrestrial amphibians, but stream amphibians decreased following wildfire.

The timing of fuel treatments and prescribed burns may be of critical importance to the maintenance of overwintering habitats, post-emergence dispersal, and reproduction of amphibian populations (Thompson and others 2003). In fire driven ecosystems, Russell and others (1999) suggested that in the long-term, prescribed burning could help maintain amphibians in managed forests by sustaining some of the natural processes of ground cover development. Two studies of salamanders in eastern United States plantation forests supported those predictions, although the plantation forests studied were young (25 years) and were compared with much older natural origin forest (Bennett and others 1980; Pough and others 1987). Prescribed fire and thinning to reduce fuel loads will remove large amounts of coarse woody material from forests, which reduces cover for amphibians and alters nutrient inputs to streams. Prescribed fire may increase the mortality of terrestrial amphibians by fire because prescribed burning usually occurs from fall to spring when amphibians in the Northwest are active (Bury 2004). Most reptiles are adapted to open terrain, so fire usually improves their habitat. Reptiles, which often exploit open areas for feeding, basking, and display behaviors, tend to increase in harvested areas (Moseley and others 2003; Perison and others 1997; Renken and others 2004; Shipman and others 2004).

While working to restore "healthy" forests and reduce the risk of catastrophic fires, forest managers must confront the challenge of maintaining biodiversity in western forests and incorporating the understanding of how fire affects semi-aquatic biota with the effects of fuel reduction management on wildlife in western forests (Bury 2004; Olson and others 2002). Management implications for forest lands adjacent to aquatic systems include effects on terrestrial and semi-aquatic species whose habitat requirements extend beyond the margins of wetlands, streams, and rivers (Houlahan and Findlay 2003; Semlitsch and Bodie 2003; Thompson and others 2003). Implementation of an ecosystem management strategy that reverses the current trend of having landscapes dominated by early and mid-successional forests would help restore depleted populations to levels where salamanders better fulfill their ecological roles as forest floor insectivores. Other management techniques that would benefit salamanders include leaving buffers along headwater streams (Sheridan and Olson 2003; Stoddard and Hayes 2005) and using harvesting techniques that ensure that the basic structure and function of forests remain intact following timbering operations.

For purposes of conservation and management, it is important to define core habitats used by local breeding populations surrounding aquatic habitats (Houlahan and Findlay 2003; Semlitsch and Bodie 2003). Semlitsch and Bodie (2003) reviewed studies of riparian habitat use by herpetofauna and found that the mean distances of riparian habitat use ranged from 117 m by salamanders to 368 m by frogs with no significant differences between mean minimum or mean maximum distances exploited by amphibians and reptiles. The survival of semi-aquatic vertebrate populations may rely on intact terrestrial habitats (up to 2 km from aquatic systems) during reproductive seasons, for overwintering, and for maintaining connections among populations (Findlay and Houlahan 1997; Houlahan and Findlay 2003; Roe and others 2003).

Houlahan and Findlay (2003) were unable to resolve the interaction between forest cover and road density in explaining amphibian species richness, but roads have been implicated in reducing amphibian dispersal (Gibbs 1998), increased mortality (Fahrig and others 1995), and reduced genetic diversity (Reh and Seitz 1990). Responses to habitat alterations associated with management activities suggest that these basins may be useful as reserves, especially in catchments where timber harvest may not be commercially viable (Sheridan and Olson 2003).

Fire Suppression Chemicals, Toxicity, and Mortality in Aquatic Organisms

Toxicity effects of fire suppression chemicals on aquatic organisms have been poorly studied (see reviews by Kalabokidis 2000; Gimenez and others 2004). Most fire suppression compounds are based on ammonium salts and toxicity is inferred to result from ammonium ion concentrations (Hamilton and others 1996). Fish, particularly salmonids, and some invertebrates appeared to be vulnerable to the application of these compounds. Boulton and others (2003) observed no short-term effects of fire suppressant chemicals on water chemistry and macroinvertebrates in streams following wildfire, but Buhl and Hamilton (1998, 2000) reported both laboratory toxicity and fish kills associated with fire control compounds.

Fuel Management Activities, Best Management Practices, and Cumulative Watershed Effects

Post-Fire Remediation and Logging

In response to the increased risk of runoff and erosion, land managers and technical specialists sometimes apply erosion control efforts (Burned Area Emergency Rehabilitation or BAER treatments) to mitigate the effects of fire (Neary and others 2000). A recent review of BAER practices indicated inadequate scientific evaluation of the effectiveness of the treatments (Robichaud and others 2000). Some practices contribute to loss of soil integrity, impede the persistence or recovery of native species, disrupt riparian functions, and impair water quality (Karr and others 2004; McIver and Starr 2000). Postfire logging and road building, undertaken on steep slopes with sensitive soils, exacerbate erosion associated with changes in soil and vegetation structure that decrease infiltration and increase overland flow (DeBano and others 1996; McIver and Starr 2000). Too little is known about the cumulative effects of site specific responses at the watershed scale, knowledge that is essential if forest management is to be linked to aquatic ecosystem integrity (Chapter 14). Research at the watershed level that integrates terrestrial and aquatic components is needed to inform management about the risks and opportunities available in the post-fire landscape. Research and resource management agencies should cooperate to develop the basic tools of experimental design that can rapidly evaluate post-fire treatments and reduce the uncertainty of its long-term effects. Salvage harvesting policies could be prepared before major disturbances occur that would guide the timing and intensity of salvage harvesting.

Natural disturbances are key ecosystem processes that help maintain biodiversity, and productivity and support ecosystem restoration by recreating some of the structural complexity and heterogeneity lost through intense management of natural resources (Beschta and others 2004; Bisson and others 2003; DellaSala and others 2004). Salvage harvesting activities undermine many of the ecosystem benefits of major disturbances by removing large quantities of "biological legacies" (for example, snags and downed trees) that are critical habitat for species and important for the recovery of terrestrial and aquatic systems (Benda and others 2003; Lindenmayer and others 2004; May and Gresswell 2003; Swanson 1981; Van Nieuwstadt and others 2001). Effects from postfire logging in riparian areas can persist for many decades because of the loss of dead trees that would normally become incorporated into stream channels and forest floors over several decades or more (Beschta and others 2004; May & Gresswell 2003). Similarly, logging large trees from upslope areas prone to landslides would also reduce, over time, the recruitment of large wood to riparian and aquatic ecosystems. Fire and subsequent hydrologic events can contribute wood and coarse sediment necessary to create and maintain productive in-stream habitats (Bisson and others 2003; Reeves and others 1995) and produce important heterogeneity in channel structure (Benda and others 2003). Natural disturbances interacting with complex terrain has been linked to a changing mosaic of habitat conditions in both terrestrial and aquatic systems (Bisson and others 2003; Miller and others 2003; Naiman and others 2000; Reeves and others 1995). Even with mitigation of fuel, large disturbances are inevitable. Only frequency and magnitude of individual events can be changed (Istanbulluoglu and others 2004). This variation of conditions in space and time may be the key to evolution and maintenance of biological diversity, and ultimately, the resilience and productivity of many aquatic populations and communities (Bisson and others 2003; Dunham and Rieman 1999; Dunham and others 2003; Poff and Ward 1990).

In forested watersheds, valued riparian functions such as stream shading, bank stabilization, and large wood recruitment are largely protected by best management practices, particularly the establishment of riparian buffers since the influence of most riparian features decreases with distance from the stream channel (Beechie and others, 2000; Van Sickle and Gregory 1990). However, the indirect effects of upland fuel treatments on riparian habitats are poorly understood (Chapter 10). Forest harvest strongly affected the riparian microclimate and the riparian environment became more similar to upland conditions (Brosofske and others 1997), but the effects do not necessarily contribute to increases in stream temperature (Poole and Bermann 2001). The effect of fuel reduction activities on drivers of stream temperature needs to be incorporated into management and restoration planning.

Hydrological, geomorphological, and ecological processes in streams and rivers are affected by the recruitment of large wood from riparian and upland portions of watersheds (Bilby and Bisson 1998; Gregory and others 2003; Lienkaemper and Swanson 1987; Montgomery and others 2003). Management plans for fuel reduction that could affect the recruitment of large wood to stream systems need to consider the importance of wood delivery, retention, and transport processes in stream habitats.

Discussion

The widespread decline in forest health has been linked to timber harvest and fire suppression (Bisson and others 2003; Brown and others 2004; Hessburg and Agee 2003). Restoration and management of healthy forest ecosystems must start from the realization that they cannot be fireproofed (Agee 1997). Given the goals of ecosystem restoration and reduction of the risk of severe wildfire, a comprehensive program of pre-fire management, post-treatment monitoring, and adaptive management will be necessary to restore western forests. Effective fuel management will recognize differences among fire regimes, forest and landscape types, and departures from natural regimes, and incorporate the differences among systems (Hessberg and others 1999a, b). Plans must address the importance of spatial pattern of treatments in changing fire behavior at the landscape scale and the strategic placement of treatment areas within constraints imposed by land ownership, the occurrence of endangered species, and the protection of riparian buffers. Resource management strategies that operate under the assumption that forest health can be improved simply by managing vegetation through silvicultural treatments risk damage to key components of aquatic ecosystems. Repeated fire and fuel management programs require knowledge of the cumulative effects of those treatments on aquatic ecosystems.

Restoration efforts based on reestablishing conditions within the range that existed in some former sustainable state with an appropriate fire and forest management program requires knowledge of historic and current conditions and conditions in reference areas (Angermeier 1997; Bisson and others 2003; DellaSala and others 2004; Quigley and others 2001; Winston and Angermeier 1995). We lack information about key characteristics of indicators of forest condition that supports management for conservation of biodiversity (Lindenmayer and others 2000; McRae and others 2001). Potential indicators that move beyond single species management focus on the composition, complexity, connectivity, and heterogeneity of forest structure, the same features that characterize the resistance and resilience of the aquatic network (Rieman and others 2003). Past conditions, including understanding the effects of the frequency, magnitude, and extent of disturbance regimes and the conditions they create over time, can be used to provide a context for managing ecosystems (Benda and others 2003; Cissel and others 1999; Swetnam and others 1999). Among the limitations of applying the "natural disturbance regime" to ecosystem management is that the extent of anthropogenic change in a region may have diminished the ability of species to respond to the disturbance (Swanson and others 1987, 1994). Silvicultural and prescribed fire management tools can lead to the goal of restoring stand structure and composition, but such restoration may have mixed implications for restoring other aspects of failing ecosystems such as fishes and their habitats. The interrelationships of terrestrial and aquatic disturbance regimes and succession processes must be understood if effective land management strategies that do not degrade aquatic resources are to be employed (Hessburg and Agee 2003; Lee and others 1997; MacDonald 2000; Macdonald and others 2004; Rieman and others 2003).

Natural variability of land use, natural vegetation, species diversity, and interactions with inherent disturbance processes needs to be defined at relevant spatial and temporal scales that are climatically, topographically, and biogeographically consistent to provide an appropriate geographic scale (Hann and others 1997). Quantifying variability requires knowledge of both the statistical properties of the condition indicator and an appreciation of the specific attributes of the indicator being described. Describing the "range" of natural variability may be inadequate as the sole descriptor because the occurrence of extreme events (pulse disturbances) may be critical in shaping ecosystem characteristics. Incorporating natural variability into management plans as testable hypotheses about the mechanisms of ecosystem change will facilitate development of future fuel treatment plans. By understanding the variability of natural regimes, using the "natural" state as a template, and modeling the departure of physical systems from that distribution of conditions, we will improve our ability to quantify cumulative effects of treatments on aquatic systems. Successful adoption of such strategies will require explicit statements about the uncertainties involved. Because restoring forest health will require repeated applications of treatments, we require better understanding of the cumulative effects of repeated disturbances.

Reducing Uncertainty: Knowledge Gaps and Research Opportunities

Effective management of riparian areas would minimize cumulative watershed effects on riparian functions by preserving their dynamic connections to upland areas and stream corridors (Reid 1998). Understanding the effects of timber harvest, fuel reduction, and fire requires integration of information about the spatial extent of management activities, temporal aspects of natural disturbance regimes (for example, fire return or landslide frequencies), and historical human disturbance (Franklin and Agee 2003). Anticipating the cumulative watershed effects of fuel treatments on riparian and aquatic ecosystems will be more difficult because of the limited knowledge of the impacts of spatially dispersed, temporally intensive, and repeated treatments in the watershed. As such, effective experimental design of fuel treatment prescriptions and post-impact monitoring will be necessary to increase the available information about the cumulative effects of management activities in sensitive areas.

Uncertainty exists over the approaches for incorporating fire and other disturbance processes into the management of systems from which it has been excluded. Treatments need to be applied as experiments that provide new sources of knowledge that will inform future management decisions (Robichaud and others 2000). Restoration of forest landscapes that protect aquatic resources requires strategies that identify and protect areas with high ecological integrity and connect them at the catchment level to increase the resistance and resilience of ecosystems (Gresswell 1999; Lee and others 1997; Reiman and others 2000). Status (condition) and trends (changes in condition over time) may be tracked at multiple levels of biological organization (from the individual level through population and metapopulation dynamics, assemblage structure, or ecosystem processes). Responses across spatial scales demonstrate the importance of designing sampling strategies and analyses capable of discerning differences from local to regional scales (Angermeier and Winston 1998; Frissell and Bayles 1996; Gregory and others 1991; Li and others 2001; Poff and Allan 1995). Study designs that measure abiotic and biotic factors at multiple spatial scales provide important information for the monitoring and assessment of stream ecosystems (Kershner and others 2004; Larsen and others 2004). Replicating habitats sampled within a stream (as is commonly done in an upstream-downstream paired design comparing reference versus impacted sites) may not reveal the impact, but only portray the natural background variability of stream communities. Both spatially extensive designs with little sampling over time and temporally extensive designs with little or no spatial sampling may be biased in terms of their view of the relative importance of local and landscape factors (Wiley and others 1997). Therefore, study designs with temporal and spatial replication that account for assemblage variability at a range of spatial scales may improve the likelihood of detecting responses of stream ecosystems to wildland fire, fuel treatments, and restoration efforts. Studies designed to follow recovery from disturbance over long time scales (decades or more) would be necessary if we are to overcome limitations imposed by data collected at time scales consistent with more frequent (seasonal or annual) disturbance events. Detailed and temporally and spatially intensive monitoring may be necessary to resolve the variance components of data collected in lotic systems (McCormick and Peck 2000; Minshall and others 2001). Alternatively, paired watershed studies intended to monitor post-fire or post-treatment effects could be designed using the comparative approaches inherent in treatment versus control and before versus after disturbance studies (Before-After, Control-Impact or BACI designs; Barbour and others 1996; Smith and others 2003; Underwood 1992; Wiens and Parker 1995).

To be an effective alternative to cumulative effects studies, adaptive management requires a commitment to regular monitoring and analysis for subsequent management decisions (MacDonald 2000). Forest management and restoration planning require data from regional assessments of forest condition that incorporate spatial and temporal patterns of vegetation, habitats, fuel, and potential fire behavior. Further planning should consider the relations between these conditions and a host of issues surrounding terrestrial habitats and associated aquatic ecosystem conditions. Before management alternatives can be selected and implemented, they must be adequately evaluated for their effects on many terrestrial and aquatic ecosystem components. However, because post-harvest effects often require decades to dissipate, the long-term experimental study of cumulative effects is very costly. In addition, high variance confounds attempt to provide integrated assessments of the effects of timber harvest on aquatic habitats and biota. There is a need for development of a modeling capability for these processes that addresses the large variations in habitat conditions (Benda and others 2004; Davies and others 2000; Keane and others 2002; Poole 2002;).

The spatially discontinuous nature of river networks can inform restoration efforts by providing a better understanding of variability at different spatial scales (Poole 2002). Prioritization of sites for treatment may be based on physical models of fire behavior and response regimes as they relate to key habitats for aquatic populations at risk (Rieman and others 2000; Brown and others 2004). Once identified, such high priority stream reaches could be reconnected via restored corridors at the landscape level. Large scale catchment features successfully predict local scale habitat (Burnett and others 2003; Davies and others 2000) or process controls (Istanbulluoglu and others 2002, 2004; Miller and others 2003; Wondzell and Howell 2004). Independent constraints on climate and geology strongly influence lower level processes in catchment hierarchies and the assemblage responses to the abiotic regime (Burnett and others 2003; Davies and others 2000). The refinement of these models to address spatial and temporal variation in landscape processes supports the emerging capability for modeling catchment scale habitat influences on animal populations and characterizes the species specific potential of streams to provide habitat for fish (Burnett and others 2003). The development of risk assessment models for population and assemblage responses to changes in catchment land use will require detailed information (Keane and others 2002). Further research will contribute to modeling capabilities that assimilate historical conditions and predict future conditions under different management scenarios by reducing the uncertainty associated with the need to estimate model parameters (Keane and others 2002; Roloff and others 2005).

Conclusions

Forest resource managers are confronted with two related management challenges. Aquatic resources, particularly imperiled salmonid populations, require protection and restoration. At the same time, the imperative to reduce the risk of catastrophic wildfires requires ecosystem based management that accounts for the inherent linkage between aquatic and terrestrial systems and involves processes that are fraught with uncertainties (Bisson and others 2003). To ensure the resilience of terrestrial and aquatic ecosystems, it is necessary to consider the restoration of terrestrial and aquatic ecosystems simultaneously. A challenge for research will be to integrate the emerging understanding of changing vegetation and fire with that of watershed and aquatic ecological processes to understand the full implications of the changes made. A challenge for management will be to restore ecological processes that support both resilient terrestrial and resilient aquatic communities. Research and management need to coordinate their efforts to arrive at a common conceptual model that expresses the uncertainties associated with particular courses of management actions and perceived risk to aquatic resources.

The Healthy Forests Restoration Act of 2003 (HFRA; H.R. 1904) increases the emphasis on fuel reduction in forest planning. However, the main impact of HFRA is largely procedural. It is not a strategy for wildland fire risk management. The prevailing conditions in today's forests developed over a century of fire suppression and cannot be dealt with effectively except by long-term adaptive management (Agee and Skinner 2005; Hessburg and Agee 2003). While restoration may be an appropriate course of action, particularly to protect valued aquatic resources, fire disturbance of forests is inevitable and even desirable. Restoration efforts must incorporate natural variability, which precludes a one size fits all approach to fuel management and post-fire activities (Landres and others 1999). The procedural mandates of HFRA will necessitate action before such research can inform the decision making process. Adaptive management, undertaken with controls (including both unlogged and unburned catchments) and replication, can provide extremely valuable information to resource managers.

Successful fuel management will depend on a strategy that incorporates natural variability in patterns of disturbance and its effects on aquatic resources. Fire management is adaptive and requires a long-term commitment to monitoring. Data will be necessary to inform the decision making process, either to reduce uncertainty in decision support tools or evaluate the results of a management option through effectiveness monitoring. Efforts to restore forest condition and maintain the connectivity of terrestrial habitats with their aquatic ecosystems will require that we inventory what we know, analyze variability in our existing data, express the uncertainty associated with analyses and associated predictions, articulate clear goals, design treatments as experiments to test specific hypotheses, monitor treatment outcomes, and apply the results in future planning processes.

We can study the natural vegetation and fire patterns of areas designated for treatment and other areas like it and apply knowledge of those patterns to their management. Recent bioregional assessment projects (for example, Interior Columbia River Basin Ecosystem Management Project and Sierra Nevada Ecosystem Project) provided more information about existing forest and rangeland conditions and the state of ecosystems and their inhabitants than ever before. Incorporation and integration of this information to address interactions between landscape spatial and temporal patterns of vegetation, habitats, fuel, and potential fire behavior will provide a basis for adaptive management that recognizes the functional attributes of the terrestrial-riparian-aquatic interface. Data analyses that characterize variability at different spatial scales will support adaptive management and planning. Modeling and simulation tools capable of incorporating multidimensional data and examining the effects of management activities on key facets of ecosystems are needed. Ideally, they would also serve as tools for planning and evaluating the effects of alternative courses of action on aquatic ecosystems. Modeling and predicting potential impacts of forest harvest operations and wildfire on water quantity and quality are critical tools for forest managers. To make these predictions, the impacts of harvest operations and wildfire on model input parameters must first be quantified with measurements. No one knows exactly how to restore healthy forests that sustain viable populations of native species in functional habitat networks across space and through time. Acknowledging that the inherent complexity and dynamism of ecosystems contributes to uncertainties about outcomes of management practices will be necessary if we are to learn from the effects of those practices and incorporate them into future management planning. Recognition that risks to aquatic ecosystems may be additive, multiplicative, or synergistic across space and time will not preclude mitigation of high fuel loads in forests. Experimental, adaptive approaches to fuel management would unite researchers and resource managers in planning, implementing, and monitoring treatments.

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Cumulative Watershed Effects of Fuel Management in the Western United States

Chapter 12.

Landscape Scale Effects of Fuel Management or Fire on Water Resources: The Future of Cumulative Effects Analysis?

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Introduction

Wildfire may pose serious threats to both direct (for example, heating, dissolved toxic gases) and indirect (for instance, post-fire floods, erosion, changing habitat), aquatic ecosystems (Dunham and others 2003; Gresswell 1999). There is, however, increasing recognition that major flood, erosion, and mass wasting events after fires can also be important to the formation of complex habitats that are beneficial in the long-term (Bisson and others 2003; Reeves and others 1995).

There is growing interest in restoration of terrestrial vegetation communities that have shifted in composition, pattern, and continuity, producing forests that are more flammable and more contagious for fire in some parts of the country (Hessburg and Agee 2003). Some have noted the potential benefit of restoration to watershed and aquatic ecosystem values because the reduction in fuel will mitigate the severity of future fires (Graham and others 2004; USDA 2000). Restructuring forests might also provide benefits through the restoration of forest-riparian functions that contribute to the maintenance of structurally complex and resilient aquatic habitats (Rieman and others 2000, 2005). At present, these benefits are largely hypothetical and others have noted that management has not always been benign (Bisson and others 2003; DellaSala and Frost 2001; Graham and others 2004). The effects of management can also be fundamentally different than severe fire, which tends to be more chronic than periodic in nature (Istanbulluoglu and others 2004; Reeves and others 1995).

An important question stems from this uncertainty: "Which poses a greater risk, wildfire or the management intended to reduce its effects?" (Bisson and others 2003; Rieman and others 2003). The answer likely depends on context, and the more contentious question is how we objectively evaluate the differences in risk between fire and management for a particular area or project. This question has typically been addressed in debate or analysis based on the apparent risks associated with the local and short-term effects of fire and the management intended to mitigate those effects (O'Laughlin 2005). Ecologically important differences between the two may only be apparent as we consider how they might play out over longer time scales (10^1 to 10^2 years) and larger spatial scales (10^3 to 10^5 ha). In short, differences in cumulative effects may be recognized at these scales.

How the relative cumulative effects of fire and management alternatives are analyzed is important. Analyses that compare only the absolute sediment or thermal loadings accumulated through a watershed will be incomplete and potentially misleading. The literature on cumulative effects has already noted that non-linear and synergistic effects of activities may be important as well (Dunne and others 2001; MacDonald 2000; Reid 1993; Sonntag and others 1987). In particular, we argue that synergistic effects related to synchrony over several watersheds or sub-populations are an important cumulative effect consideration for comparing the effects of fire versus fuel management.

Cumulative Watershed Effects

By strict definition, cumulative effects are expected to manifest only as a result of multiple management decisions or projects (CEQ 1997). A more general definition describes cumulative effects as those that occur over larger spatial scales and longer time scales and through potentially non-linear interactions of multiple processes (Sonntag and others 1987). Concern for cumulative effects, resulting largely from the National Environmental Policy Act of 1969, has challenged the limits of existing scientific theory in many disciplines, and literature has tried to frame an efficient and effective cumulative effects assessment process (Dunne and others 2001; MacDonald 2000; Reid 1993; Sonntag and others 1987) (Chapter 12). Because watersheds are a natural accumulator of potential pollutants, such as sediment, nutrients, or thermal energy, they have been an area of focused development for analysis procedures (Dunne and others 2001; Reid 1993).

An individual project with ground disturbing activities typically displays watershed effects near the stream's headwaters. Downstream, these effects are diluted (Bisson and others 1992). However, when multiple projects are considered over a short time period, potential risks to downstream areas may be more apparent because of the accumulated effects (Bisson and others 1992; Reeves 1993). Best Management Practices were developed to mitigate the effects of any project; however, even the relatively minor effects of many projects, particularly during a short time period, might still substantially change downstream habitats. Consequently, most of the concern with respect to forestry in watersheds is with cumulative effects.

This is a well defined problem with a substantial amount of literature providing guidance for the assessment of cumulative watershed effects (CWE) of forest management activities. A strong underlying theme is the downstream transport and accumulation of watershed materials (water, nutrients, sediment, thermal energy) from multiple land use actions (Dunne and others 2001; MacDonald 2000; Reid 1993). The topological context provided by a watershed makes this a natural consideration. In addition, synergies between upstream and downstream actions are possible, for example, how the changes in gradient and hydraulic geometry along a stream interact with changes in sediment and wood supplies and how the downstream reaches can integrate effects from many projects in ways that may be more complex than the simple addition and dilution of materials.

In contrast to the integration effects over the *spatial* domain of a watershed, quantitative cumulative effects analyses for forestry often consider the distribution of effects across *time* (Cline and others 1984; Dunne and others 2001; MacDonald 2000; Megahan 1974; Reid 1993; Washington Forest Practices Board 1995), generally with annual time steps. One reason for this approach has been to examine patterns of recovery after anthropogenic disturbances. This approach is in line with the Total Maximum Daily Load (TMDL) concept used to regulate cumulative pollutant loads from multiple point and non-point source activities (such as silviculture or agriculture) under the Clean Water Act. One of the key mitigations for point loading activities is to meter out pollutants generated in bulk to allow flow and biological processes to dilute, assimilate, or transform the pollutant. TMDLs are regulatory limits to pollutant loadings that are set using state standards and the expected flow at a given point that will dilute the pollutant. This approach is easily applied for silvicultural chemical use, but prescribed allowable loadings for sediment are more commonly based on an average over an unspecified multi-annual time scale than on a daily time scale. TMDLs become, in essence, a prescribed limit to CWEs. The goal of many CWE analyses then becomes manipulation of various treatments and mitigations to control the distribution of pollutants over time at each required point of compliance.

A comparable analysis of the distribution of cumulative effects in *space* is seldom considered. The growing body of work in landscape ecology argues that this is an important issue. In essence, the spatial pattern, structure, and quality of habitats may have a profound influence on the resilience, persistence, and diversity of aquatic populations and communities (Naiman and others 1992). From a cumulative effects perspective, the question becomes not just how much degradation within a watershed could cause significant declines in important biological indicators (for instance, abundance or productivity of sensitive species), but how many stream segments within a watershed or how many watersheds within a larger river basin might be degraded simultaneously (Benda and others 1998; Reeves and others 1995). An example comes from the Oregon Coast Range, where substantial work has documented that the downstream accumulation of impacts has degraded habitats and decreased diversity of populations in streams with logging (Bisson and others 1992; Reeves 1993). It has been estimated that 86 percent of 5th code watersheds in the Oregon Coast Range have seen reductions in forest cover between 1936 and 1996 (Wimberly and Ohmann 2004). This is both a remarkable achievement of human engineering and labor and a potentially dangerous ecological situation for what was once a large interconnected collection of populations of fishes and other organisms. It seems reasonable to suggest that analysis of cumulative effects within any of the remaining 14 percent of the coast range watersheds consider the importance of their role within the larger context.

Cumulative effects analysis then may consist of two contrasting approaches—a *serial* (downstream) cumulative effects technique derived from multiple impacts along a single flow path and/or on *parallel* cumulative effects technique derived from multiple impacts among individual flow paths that do not necessarily flow into one another but may share a common ecological context (fig. 1). The serial analysis considers the downstream transport or accumulation of many watershed products, for example, water, sediment, nutrients, and energy. It also considers non-linear effects from serial contributions with changes in transport capacity and the interactions of multiple constituents. Parallel analysis captures effects to watershed products that move both up and downstream (for instance, genetic material, biological populations), and are vulnerable to processes that cross boundaries of multiple watersheds. Both have defining scales in time and space. With respect to the interpretation of the significance of effects, the serial analysis can be thought of as an integrated approach in space, while a parallel approach will be more spatially explicit. In the serial analysis, we may accumulate spatially distributed inputs to look at effects at a point. In the parallel analysis, the idea is to consider the spatial relationships between affected habitats and sub-population or population units. The context of fire and fuel management is particularly relevant for the contrast between parallel and serial analyses, precisely because of the large spatial scale that fires and proposed fuel management projects may occupy.

The cumulative effect literature has emphasized the point of considering non-linear, non-additive, or synergistic interactions of multiple actions or processes (Dunne and others 2001; MacDonald 2000; Reid 1993; Sonntag and others 1987). The parallel analysis is one example of this idea that focuses on geographic interactions. The reason we highlight the distinction using the serial-parallel classification is that this particular geographic interaction is a strong example of the synergistic effect; could apply to the conservation issues for many sensitive aquatic species populations (for example, salmon and trout); may have profound consequences for management and regulation; has limited exposure in existing CWE literature; and, as a consequence, has had little exposure with watershed management professionals. The general scope of the idea has been recognized for decades as being a cornerstone of cumulative effects analysis for terrestrial vertebrates (Collinge 1996; Debinski and Holt 2000; Fahrig and Merriam 1994; Sonntag and others 1987; Wiens 1976; Zavala and Burkey 1997). However, even

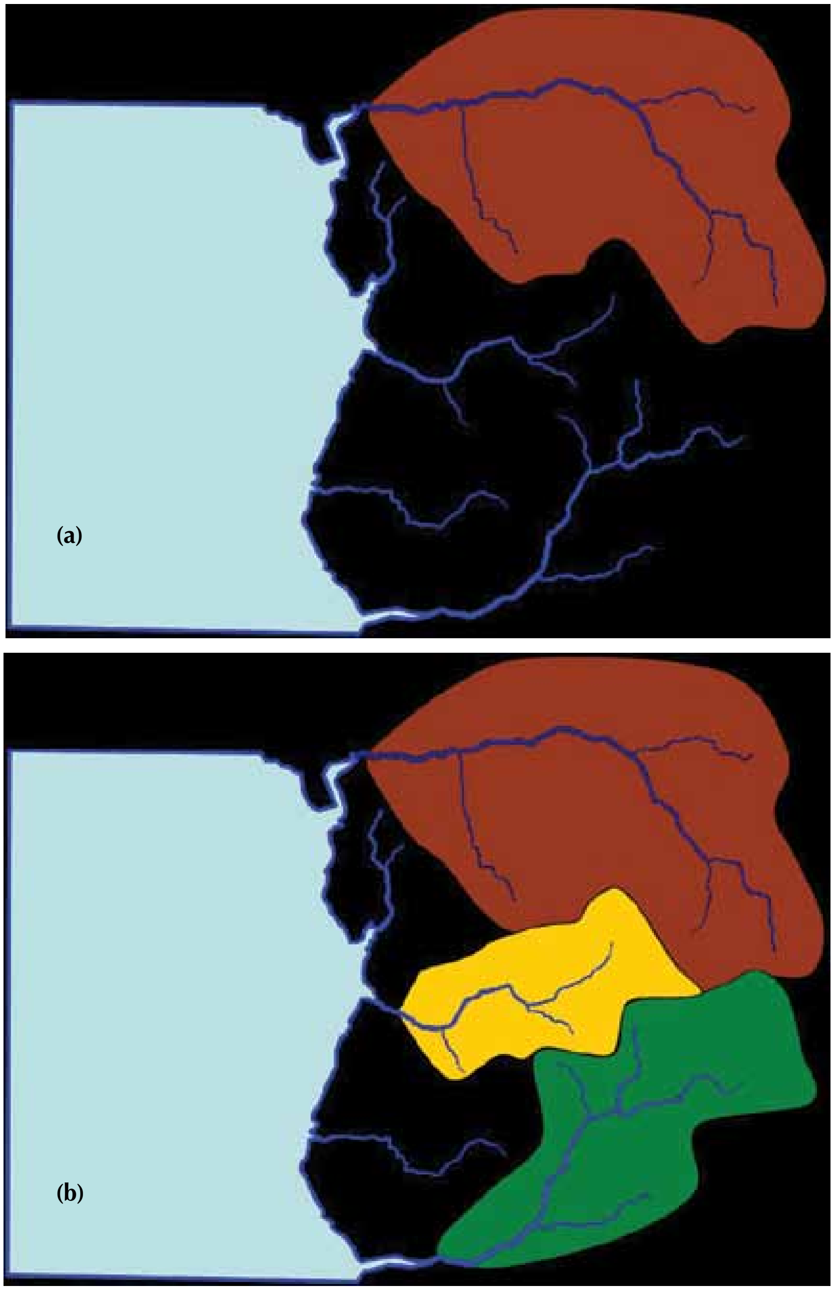

Figure 1. In serial cumulative watershed effects analysis, the effects focus on the accumulated contributions above a certain point, for example the northern watershed (brown) in (a). In a parallel cumulative effects analysis for the same basin, one may need to consider effects to populations in neighboring basins (green and yellow) linked only through a separate waterbody (light blue) to which they are tributary "in parallel," such as a large river, lake, or ocean. It may also be helpful to look within the original basin at stream segments (varying segments of white to blue) that may be affected independently by extreme events. We suggest that both types of analysis may be useful with respect to fire and fuels management.

broad, general discussions of cumulative *watershed* effects still emphasize the downstream accumulation of water and sediment as the driving factor in *cumulative* effects analysis (Dunne and others 2001; MacDonald 2000; Reid 1993).

When addressing geographic interactions, synchrony or asynchrony of impacts in separate populations is the rationale for parallel analysis. This approach tends to require larger spatial scales than have traditionally been used for downstream accumulation types of analysis. Recent advances in the understanding of scaling in physical and ecological processes provide a foundation for quantitative analysis of effects at larger scales and in more complex landscapes than was practical in the past. In light of this growing understanding and the need for solutions to problems posed by fire and fuel management decisions, we believe it is important to revisit the utility of parallel cumulative watershed analyses of ecologically and operationally relevant effects that are spatial scale and pattern dependent.

There are attendant choices in temporal scale to be considered as well. Different resources are affected over different time scales. For example, an analysis of concerns about a reservoir filling with sediment could consider information averaged (or summed) over a few decades, while analysis of ecological concerns must consider shorter time scales consistent with species seasonal habitat use and life cycles (for example, months to years). Both biological and physical processes have important seasonal variations that may create non-linear interactions. For instance, in climates with summer convective storms, production of fine sediment from roads is introduced to streams when flows are too low to transport the sediments away. Because time scales are important ecologically, some have noted the need to not just consider the magnitude of sediment loading, temperature, or peak flow changes, but the temporal distributions as well (Poole and others 2004). Important aspects of time are frequency or spacing of disturbances, duration and recovery times, and the overall temporal extent. A key concept in the discussion of disturbance and population response is that changes in the "predictability" or the timing and frequency of disturbances can have a profound influence on native species that evolved under one regime and are now faced with something novel in their evolutionary experience (Poff and others 1997). The issues with time and scale are inherent in either serial or parallel analyses, but with the latter, the concept of temporal synchrony in disturbance among multiple analysis units represents an important dimension for parallel cumulative effects analysis.

Ecological Limitations of Serial CWE Analysis

Serial analysis is most useful for describing processes where thresholds may be exceeded or long-term total loads are important. Reaches that trap substantial sediment and wood, for example, reservoirs, lakes, estuaries, and spawning habitat, are points where changes to total sediment load may be an important quantity to assess for that site. There are similar circumstances where total loads of nutrients such as phosphorous or nitrogen would be important to ecosystem processes (Reckhow 1999). Historically, there has been a strong reliance on serial cumulative effects for forest management, particularly for rigorous process based analyses (Cline and others 1984; examples summarized in Reid 1993; Washington Forest Practices Board 1995).

Modeling of sediment loads has been applied in a general way to the problem of fire versus fuel management. Elliot and Miller (2002) provide an example comparing the total loads of sediment derived from surface erosion of roads that might accompany active fuel management (for instance, mechanical thinning) over multiple years versus a fire event that erodes and recovers. They assumed that one disturbance replaced the other. Istanbulluoglu and others (2004) provide another example contrasting the effects of timber harvest and the effects to fire, also assuming that the first could be used to replace the latter. The two studies showed very different results. Elliot and Miller (2002) found that management produced substantially less sediment overall and Istanbulluoglu and others (2004) showed relatively similar values between treatments. The primary differences between the models were the slope of the land considered (moderate gradient in the first and steep in second) and consequently, the physical processes modeled (surface erosion versus mass wasting). The results are consistent with the situations and temporal scales assumed in each analysis, and the results of Elliot and Miller (2002) reflect the fact that surface erosion from roads is commonly a small part of the total or long-term sediment yield in moderate to high relief landscapes (Luce and others 2005).

An important lesson from Istanbulluoglu and others (2004) was that while the total load integrated over time was about the same, under a management scenario, the individual landslide events were smaller and the frequency was higher, changing from once every few centuries to once every few decades. From an ecological perspective, changes on this order could have profound effects on the succession of vegetation and the ultimate structure of the channel and availability of habitats. We might anticipate a similar result in the temporal distribution of sediments with a pulsed introduction following a fire compared to a smaller but more chronic supply associated with roads. The duration, persistence, or frequency of impacts is often more ecologically relevant than the magnitude of individual events, so brief high loading events after fire may be less damaging than persistent minor loads from roads (Reeves and others 1995; Rieman and others 2003; Yount and Niemi 1990). Although Elliot and Miller (2002) showed that total yields may differ substantially, they provide no information that can resolve the differences that influence ecological processes through time.

While there are ways we can improve serial CWE analysis, such as being temporally explicit or using regime based (stochastic) standards that better reflect the natural history of any basin (Poole and others 2004), there are still limitations to a spatially integrated approach. First, standards or analyses based on natural disturbance regimes are problematic because we may have limited understanding of what the "natural" regime is and the degree of departure that is biologically important (Reckhow and others 2001), and second, the spatial details may be important (Luce and others 2001).

These shortcomings are not just academic in scope but they lead to particular problems from an ecological perspective. First, attempts to meet a specific limit to change any single segment or watershed leads to policies spreading impacts over larger areas and longer durations. If loading for a particular stream needs to be kept below a particular standard, then disturbance in the basin must be limited within a given time frame, in other words, acres or harvest or miles of new road per decade. This means that planning to optimize activities on the landscape to produce timber or restore vegetation patterns must move activities from basin to basin, metering out the potential for impacts at the prescribed level on a continuous basis. Under such a scenario, most watersheds would eventually be expected to become compromised to some degree (Reeves and others 1995; Rieman and McIntyre 1993). Although each would meet the minimum criteria defined by typical CWE, few would retain the full productivity characteristic of more intact systems.

Second, attempts to identify an optimum or threshold condition of disruption leads to poor preparation for major stochastic disturbances. Essentially, major disturbances are treated as an exception in serial CWE analyses. Large floods, fires, and debris flows are often well beyond the scope of what can be manipulated by human intervention, which means that they are not generally considered within a regulatory framework. However, they do provide important ecological context within the system being regulated, and they often provide the greatest proportion of the load averaged over time (Istanbulluoglu and others 2004; Kirchner and others 2001). When working with a constant standard, a single major disturbance can fill an allocation for a long period, which is often not considered a reasonable burden for human interests to bear. Unfortunately, a stochastic standard (regime based or range of natural variability) could not provide a framework either, since one would need to know the context of that event within the distribution of events to know whether the standard had been met. Consequently, major disturbance is treated externally to the planning, monitoring, and regulatory (in other words, land management) processes. Without clear recognition of these events in the planning process, means to mitigate their effects are limited and reactive. Fortunately, there are management strategies and practices that can reduce risk from catastrophic events with forethought, so it is possible to design ecosystems that are more or less resilient to them. These strategies must consider natural adaptations that allowed species persistence through major disturbances (for instance, use of refugia, migration and dispersion, variable life histories) and the spatial scope of natural disturbance and management (Dunham and others 2003; Rieman and others 2003).

Parallel CWE Analysis to Extend Ecological Relevance

Fish and other aquatic species have survived for millennia with disturbances of varying scale and magnitude, some much greater than anything we have seen or created through management. A recent example is the aquatic ecosystem recovery from Mt. St. Helens' eruption (Dale and others 2005). Evidence from paleoclimatic studies suggests that fires have been severe and even more extensive in the past (Meyer and Pierce 2003; Whitlock and others 2003). Sedimentation data reinforces this, where drainages in excess of 100 km^2 show evidence of occasional large disturbances as the major source of long-term sediment yields (Kirchner and others 2001). Furthermore, as noted earlier, there is growing evidence that productive aquatic habitats can benefit or even depend on fire related disturbance (Benda and others 2003; Reeves and others 1995). Certainly, the evolution of aquatic species in the western United States has been influenced by a violent past, and the species and species assemblages have likely evolved in response to fire and related disturbances.

Emerging work in population biology and landscape ecology provides some insight into how species survive in disturbance prone environments. The expression of *migration* and the *spatial structure* of fish populations or networks of populations appear key (Rieman and Clayton 1997; Rieman and Dunham 2000). Migration means that individuals move among different habitats, generally in response to the availability of resources necessary to complete their life cycles. Because different habitats are distributed in space, migration is often variable in timing and extent, and the spatial extent and duration of catastrophic events in streams is limited (Miller and others 2003); not all members of the population are vulnerable to the same disturbances at the same time. Other species, such as amphibians and macroinvertebrates, may have life histories with terrestrial components, taking some part of the population out of harm's way by entirely removing them from the stream (Pilliod and others 2003). Spatial structure implies that species may exist in a network of habitats linked through dispersal. If fish are lost to disturbance in one habitat, for example, it may be recolonized through dispersal from others. From an engineering process control point of view, fish populations are using spatial and temporal complementation and redundancy to mitigate the risks associated with any particular strategy. Both mechanisms require a spatially extensive and interconnected network of habitats.

Forest management interferes with these survival processes by

1. fragmenting habitat with physical or thermal barriers;
2. encroaching on habitat, reducing the quality, number, and size of habitats composing the network; and
3. increasing the chances that spatially distinct habitats may be degraded simultaneously.

While serial CWE does not address these issues, parallel analysis can.

If the spatial structure and quality of habitats has an important influence on species persistence, then an analysis of CWE in the context of spatial structure and its variability seems important. The *range of natural variability* has been proposed as a foundation to characterize variability, and an important part of the quantification is in geographic pattern and spatial structure (Swanson and others 1997). Although the concept is commonly applied at point or reach scales, the range of variability at a given point can often be anything from severely disturbed to pristine or simplified to complex, which is uninformative. A more informative approach would be to quantify the spatial distribution of habitat conditions in a population of streams or watersheds (Benda and Dunne 1997). The goal in the context of CWE would become the maintenance of the total amount, grain size, and spacing of conditions that is consistent with the evolutionary past, or at least the distribution of conditions that will allow native species and communities to persist in the future. While actual disturbances could not be managed, the spatial distribution of risk might be. Such an approach simultaneously reduces risks to populations while allowing short-term increases in risks to segments of those populations for the long-term benefit of both terrestrial and aquatic ecosystems. With appropriate analytical support, managers might consider the frequency distribution of conditions in a population of watersheds (Benda and others 1998). For example, simulations of historical disturbance indicated that at any point in time, no more than 40 percent of the watersheds in the Oregon Coast Range were in a condition of reduced productivity or complexity resulting from recent natural disturbance (Reeves and Duncan 2009). This level of disruption could become an ecologically defensible standard for spatially explicit cumulative effects. That is, no more than 40 percent of the watersheds in a larger basin could be in a degraded condition at any point in time. Ultimately, natural patterns of forest succession, disturbance, and watershed recovery would dictate the amount of human related disturbance that any basin could support (Swanson and others 1997).

Many landscapes have been strongly influenced by spatially extensive disruption and habitat fragmentation (Hessburg and Agee 2003). Management activities that contribute to the restoration of processes that will ultimately lead to a more resilient landscape and connected network could be important. Rieman and others (2000) suggested that there may be parts of the landscape where terrestrial restoration would not be in conflict with aquatic restoration goals even if it did contribute to short-term, local degradation of habitats. In particular, they identified that most of the forest in need of structural restoration was at relatively low elevations in mixed severity ponderosa pine forests. Commonly these are also areas along main stem corridors that have experienced silvicultural manipulations and fire suppression for several decades. These areas are less likely to have strong populations of sensitive species, which have gradually become isolated in smaller and higher elevation tributaries, using main stems for migration when not blocked. While efforts to restore forests along these main stem corridors would likely increase sediment loads and the likelihood of landslides and debris flows from steep facing drainages, those loadings and events would be of little immediate ecological consequence since few important populations remain. Even where important populations remain, such disturbances could benefit populations if not spatially extensive within a particular habitat patch (Benda and others 2003; Rice and others 2001). If these projects break up fuel continuity, reducing the spatial footprint of individual fires and related disturbances, and leverage the restoration and reconnection of stream networks that could become productive elements of a larger spatial network in the future, the long-term benefits could still be important ecologically.

Challenges to Implementation

While there are some clear benefits to parallel CWE analysis, it represents additional effort and a shift in the way of thinking. Acceptance of the additional effort by land managers will be needed. In exchange, decisions would be more firmly grounded in impacts to ecology, which often increase flexibility for landscape restoration projects. Local and temporary disruption to watersheds or streams with little current ecological value or vulnerability as a tradeoff for potential long-term benefit would challenge the current regulatory framework. Ultimately, however, we believe such an approach is more centered on the overall goal of conserving species and their potential for resilience and adaptation in changing environments, not just protection of current habitat conditions.

Both managers and regulators will require sound science that demonstrates that the geographic relationship of a group of actions is key to the significance of their impact on aquatic communities, even across multiple watersheds. They will also need evidence that both aquatic and terrestrial ecosystems would benefit from using spatial pattern information to make decisions, even with the risk of increased loading. Some of that science must explore spatial and temporal scales of disturbance to understand the environment in which current ecosystems evolved. At issue is a need to understand the bounds of the physical processes that we would like to influence and emulate (Landres and others 1999), in other words, what are the limits of disturbance at larger spatial scales and longer time scales? Finally, science needs to provide improved and more efficient means to inventory aquatic habitat and population conditions over large areas. Parallel analysis steps out of traditional monitoring of reaches or watershed outputs and into a distributed view of the aquatic landscape. Terrestrial ecologists have been able to use aerial photography and satellite imagery for some time for their inventories, and similar technology is needed for this type of approach. Many of the scientific challenges are being addressed by a range of studies at this time. Agencies will receive the greatest benefit from this research within the conceptual framework of combined serial and parallel analysis.

Conclusions

Considering spatial patterns in CWE analysis presents some advantages over the spatially lumped approach that has traditionally been used. Aquatic biology has begun to incorporate a landscape perspective to better predict population and community dynamics and persistence in the face of major disturbances. The physical sciences supporting the relevant analyses of disturbance will also require flexibility in spatial and temporal scaling to match ecological process scales, in other words, to explore the spatial pattern and timing of landslide and debris flow events across a network of streams under natural wildfire regimes compared to a range of managed regimes.

While this portends more effort by management and regulatory agencies, the development of the science has led to automation of GIS tasks for spatially distributing impact analysis (Prasad and others 2005). Advances in estimating the local impacts of projects and serial cumulative effects as discussed in this book are also directly applicable within a parallel CWE approach.

The subsequent analyses will have a closer tie to ecological outcomes, making them more useful and more defensible. In particular, spatially distributed analyses support planning that is based on the natural range of variability concept. Although the transition from an existing landscape with spatially extensive homogenization and degradation of habitat to one where natural disturbances play a more active role is challenging, it is likely that the parallel approach can highlight priorities for restoration activities. At the same time, it could also highlight where the short-term risks posed by restoration activities might be unacceptably high without advance preparations.

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Cumulative Watershed Effects of Fuel Management in the Western United States

CHAPTER 13.

Tools for Analysis

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Introduction

This chapter presents a synthesis of current computer modeling tools that are, or could be, adopted for use in evaluating the cumulative watershed effects of fuel management. The chapter focuses on runoff, soil erosion, and slope stability predictive tools. Readers should refer to chapters on soil erosion and stability for more detailed information on the physical processes involved.

All cumulative watershed effects (CWE) tools are models of natural processes. A "model" is a mathematical or qualitative representation of nature. It includes an understanding of the analysis area including the identification of the important features and processes, such as topography, soil properties, and vegetation and climate, as well as their interactions. Models provide answers to the question, "What watershed changes are anticipated as a result of proposed fuel management activities?"

Getting Started

The first step in modeling cumulative watershed effects (CWE) is to define the problems that need to be analyzed. Chapter 14 outlines the qualitative and quantitative questions to be answered in a watershed analysis. Briefly, the manager must define:

- values of concern (water resource protection, human welfare, wildlife issues, etc.);
- predictions needed (peak runoff rates, water yield, upland erosion rates, stream sediment delivery rates, etc.);
- the scale of the analysis (for example project, watershed, landscape);
- the environmental constraints (for example, vegetation, climate, topography);
- the temporal context (single event, average annual, return period analysis);
- requirements of local, state, or federal regulatory agencies; and
- organizational constraints (due dates, available resources, computer capabilities, level of analytical and computer skills).

Once the problem is clearly defined, the watershed manager should review current data availability, such as soil surveys, GIS layers, air photo libraries, weather records, past watershed disturbance history, and stream flow records for conditions similar to those of the watersheds of concern. If it has not already been done as part of other analyses, the manager should conduct a field survey and develop a set of field notes. Typically, slope lengths and gradients, vegetation conditions, and relevant soil properties (texture, depth, evidence of water logging or erosion, water repellency, etc.) are noted. During the field survey, the manager should note current conditions and evidence of past activities contributing to these conditions, such as compaction or evidence of stunted or unusually lush vegetation, and if appropriate, evidence of past mass movement.

Impacts of concern

Generally, two public values dominate cumulative watershed effects analyses: impacts on aquatic ecosystems (Chapter 11), and impacts on human resources (for example, water supplies, structures in flood plains or on alluvial fans, recreation resources).

In some cases, the values at risk or level of public impact may influence the selection of a more or a less sophisticated modeling approach. For example, a high visibility watershed on the edge of a major city may require greater analysis than would a remote watershed adjacent to a wilderness area because the potential risk to offsite values is much greater.

What needs to be predicted

Before choosing a modeling tool, the manager first must identify the specific predictions that are necessary. Typical predictions include annual water yields, peak runoff rates, and related attributes such as runoff duration and time to peak, upland erosion rates, and watershed sediment delivery rates. Predictions may be for average values or probabilities of exceeding a given value. It is desirable, but not always possible, to use the same model to predict both runoff and erosion. Currently, few tools predict both. The WATSED suite of models predicts both, but the erosion and sediment delivery are not linked to the runoff. The WEPP model predicts both runoff and erosion, and the more recent versions of WEPP (Version 6.2 or later) include lateral groundwater flow in watershed runoff (Covert and others 2005; Dun and others 2006). The SWAT and AGNPS models predict both runoff and erosion, but do not link the two on the hillslope. The runoff is used to route eroded sediment through the stream system with these models.

Table 1. Scales of analysis and dominant processes.

Scale:	Small	Medium	Large
Area	< 100 ha (250 acres)	100–500 ha (250 acres–2 sq mi)	Over 500 ha (Over 2 sq mi)
Dominant Runoff Processes	Surface runoff from rainfall excess or saturated overland flow	Surface and shallow lateral flow	Shallow lateral flow and groundwater processes
Dominant Sediment Processes	Rill and interrill erosion; landslides	Rill, ephemeral gully, and gully erosion; landslides and debris flows	Channel erosion and transport processes, large deep-seated landslides
Dominant Disturbances	Wildfire; prescribed fire or road surface erosion	Wildfire, stream crossing failure, large runoff events	Large runoff events

Attributes of Tools

Scales and frameworks for analysis

Within the context of this chapter, watershed impacts of management can be evaluated on a project or hillslope scale (5 to 40 ha) or a small (under 100 ha), medium (100 to 500 ha), or large (over 500 ha) watershed scale (table 1).

For hillslope scale, onsite surveys coupled with soils and contour maps generally provide adequate information for analysis. Analyses are typically carried out with hillslope tools, hillslope by hillslope, and results are compiled in summary tables. Air photos may be particularly beneficial for mass wasting and road erosion analysis at this scale. For mass wasting, photos before and after significant mass wasting events are often compared and the features of failure sites are linked to other site conditions, such as slope steepness, upslope area, and disturbance history.

For larger area analysis, geographic information systems (GIS) generally are the most effective means to compile, integrate, and synthesize data required to describe the watershed and to run sediment, water, and stability modeling tools. Although models vary in their data needs, managers often access publicly available datasets in their GIS (for example, USGS digital elevations, NRCS soils data, and climate files), generate derivative datasets from elevation models (such as gradient and aspect data), and delineate watersheds with integrated drainage channels networks. Watershed delineation is especially valuable when first starting a project or for smaller watershed modeling projects where the limited scale of the analysis does not warrant use of the more detailed and labor-intensive modeling systems. Larger watersheds are frequently divided into small watersheds or hillslope polygons known as hydrologic response units (HRUs) to aid in describing specific areas within a watershed where a given management activity is targeted, such as a thinning operation, or a significant disturbance has occurred, such as a wildfire. One of the challenges in modeling larger watersheds is in linking the HRU runoff and sediment processes to the larger scale (Beighley and others 2005).

Sources of sediment

There are seven typical sources of sediment associated with fuel management (table 2): surface erosion from undisturbed and disturbed forest hillsides, runoff and erosion from forest road networks, sediment delivered from mass wasting processes, and sediment from channel bed and bank erosion. Hillside disturbances tend to be ephemeral, lasting 1 to 3 years before the hillslope is recovered, whereas roads can be a chronic source of sediment, generating sediment every year. Landslides generally generate sediment only during prolonged wet spells when forest hillslopes become saturated or when there are unusually high runoff-initiated debris flows in upland swales. Hillside and road erosion processes are described in Chapter 5 and channel and mass wasting processes in Chapter 6.

Hillside sources of sediment associated with fuel management are further complicated by wildfire effects. High severity wildfires tend to generate much more sediment than do lower severity prescribed fires or wildfires. The impacts of fuel management on fire severity and frequency are discussed in Chapter 3. In order to fully evaluate

Table 2. Typical sources of sediment in a watershed analysis.

Source	Frequency of occurrence	Relative erosion amount
Hillslopes following wildfire	20 to 200 years	100
Landslides	5 to 10 years	5
Hillslopes following prescribed fire	5 to 20 years	10
Hillsides following thinning	10 to 40 years	1
Undisturbed hillslopes	Yearly	0.1
Road networks	Yearly	2-5
Stream channels	5 to 10 years	5-90

the watershed impacts of fuel treatment activities, the manager also must consider the potential for erosion from wildfire. Treatment erosion rates are generally much lower, but will likely occur more frequently than wildfire events. Hence, the manager will need to carry out a series of analyses comparing erosion from wildfire at the current forest condition to erosion from fuel treatment and erosion from a wildfire following fuel treatment (table 2).

Sediment production from both roads and hillslopes is largely dependent on the weather. Wet years will generate more runoff and sediment. If there is a wet year following a hillside treatment, there is a risk of a high level of erosion, whereas there will likely be little to no erosion following fuel treatment in an average to dry year. Roads tend to generate sediment even in dry years, but at a much lower rate.

Routing of sediment through a stream system is also highly weather dependent, with little sediment being routed most years, and most sediment being routed during years with major runoff or flood events. Thus, the stream channel serves as a temporary sediment bank, storing sediment from a disturbance for several years to decades as sediment from upland areas is gradually routed through the system.

Many of the nation's forests are in areas with steep relief and major differences in climate. Higher elevations tend to have higher precipitation, but a greater amount of that precipitation tends to be snow. Snow melt rates generally are much lower than rainfall rates, and so tend to generate less surface runoff and erosion. Snow hydrology processes also are influenced by the presence of trees, with mature forests holding significant amounts of snow in the canopy and thinned openings and fringes of open areas adjacent to forests being areas of snow accumulation. Large open areas may experience less snow accumulation during times of high wind speeds as the wind scours snow from these openings. Open areas with less wind scour could also be areas of high snow accumulation due to less canopy interception. Thus the severity of wildfire, ephemeral nature of hillside disturbances, chronic nature of road sediments, impacts of elevation on climate, effects of forests and wind on snow hydrology, and infrequent routing of stream sediments are major challenges to modeling the watershed impacts of fuel management activities. The ability of various tools to model these attributes will be evaluated in the remainder of this chapter.

Hydrologic considerations

In addition to consideration of the importance of scale and sources of sediment, there are a number of hydrologic considerations to consider when selecting cumulative watershed effects tools. The first is the nature of the climate. The hydrology may be driven predominantly by thunderstorms in the southwestern or eastern United States, frontal storms along the west coast, snowmelt in the higher elevations of the western mountains, or rain-on-snow events in the Interior Northwest. If climates are dominated by snowmelt, hydrologic tools developed for rainfall dominated climates will not function well. Among these tools are the Universal Soil Loss Equation and NRCS Curve Number technologies.

Hydrologic analyses of large area flood events may include the development of runoff hydrographs. Specialist tools are available for such analyses, but are generally best done by other agencies, such as the Corp of Engineers, or consultants who have had experience with these tools.

Another hydrologic consideration may be the seasonal or long-term distribution of runoff. For example, if there is concern about erosion following a chemical brush control operation in July, the manager may wish to estimate the runoff and erosion risk during a single month and will need a model that has such a capability. Long-term changes in runoff characteristics may be of interest if a long-term fuel management plan is envisioned. Such long-term changes may have impacts on changing channel morphology if annual peak flows are increased significantly.

Environmental constraints

Some of the currently available predictive models were developed by agricultural researchers focusing on agricultural conditions. This is particularly true for the USLE-based models and the NRCS Curve Number technology. Some of these tools have been further developed for forest conditions, such as the USLE for southeastern forests (Dissmeyer and Foster 1981, 1985). Some tools may not be stable for some common forest conditions, such as steep slopes.

If the model is empirically based, managers need to be careful if adopting a model that was developed for different climates, vegetations, or geologic conditions. For example, the WATSED technologies were originally developed with data from central Idaho. Methods have since been developed to adapt this tool for other areas in the Northern Rockies, but not for forests elsewhere. The Washington Forest Practices (1997) has extended some of the WATSED technology for application in Washington State.

Local constraints

In some cases, state or local regulations may dictate the model that is selected. For example, at one time, the USLE and RUSLE models were considered inappropriate for rangeland conditions (NCBA 2003). Numerous city and county codes require that peak runoff rates be predicted by a specific method, generally the Rational method or Curve Number method, or for some specific design storm. There may be guidelines developed by the Forest Service or other agencies for runoff curves based on past experience or observations that can be applied locally, although the manager may wish to adjust the values based on site characteristics. In these cases, the manager will need to provide credible information to support any adjustments made to such guidelines.

Organizational constraints

As new models or new applications of existing models become available, the manager will likely want to compare the new tools to existing methods or observations to be confident that the model is providing reasonable predictions. Some of these new models may still be under development, in which case the managers may need to work closely with the developers in research organizations or universities to make sure that the model is correctly applied and the results properly interpreted. As newer models begin to receive more widespread application and acceptance, such collaboration is less critical.

One of the first considerations may be the skills, training, and experience of the manager. Has he/she developed the skills on a given tool or is some training necessary? Is such training available?

Depending on the manager's level of computer expertise, the model interface may be crucial. Models requiring high levels of GIS skills may be acceptable to some specialists, whereas others may have limited GIS modeling experience. Some tools are now available online, but the assumptions associated with the simplified interfaces may not provide the manager with the flexibility needed for some analyses. Some managers may not have a high speed Internet connection, which may limit the use of these models.

Additionally, the computer hardware available to the manager may be inadequate for some models, lacking the memory or speed required for the tool. Older computers may not have the software necessary to run some models, such as recently developed GIS or spreadsheet-based models.

Other considerations

There are a number of other attributes that a manager may wish to consider when selecting a tool. Is the model under consideration widely used and accepted by the academic, legal, and environmental community? For example, one of the reasons for a legal decision against the Forest Service was that the model used for the watershed analysis did not predict any variability associated with the estimated means. Another consideration is the availability of the model.

Obtaining the model may be a problem. Is it online or does it require installation from a CD? If it is on a CD, does the manager have the latest version? A few models require registration to proceed with immediate download, and less commonly, some tool developers require that the user first register and then wait to be sent a download access password. One tool reviewed, SedMODL2, charges a fee to cover the cost of burning and shipping software on a CD. Once downloaded, many applications are installed through common installation wizards with special computer configuration requirements. Some installations may require the manager to get administrative permission to install the software. Some models that fall into the research domain, such as DHSVM, require the user to compile code and install special system emulators. These installations may require assistance from computer support specialists.

To use a tool, a database must be built. This can be one of the biggest challenges in operating a given tool. Are data readily available to run the model from either a Forest Service database or a public source? For example, some of the more sophisticated hydrologic tools require detailed information on soil depths and properties and properties of bedrock, which may not be readily available. In some cases, a database may need to be reformatted or certain fields extracted to obtain the necessary information to run the model. If online databases are used, it is the user's responsibility to ensure that the data are correct. For example, one online data set contained information from a faulty sensor at a SNOTEL site that was only discovered when the model was not performing as expected.

Many models include file builders to aid in preparing the data for the model. The input file builder may be as straight forward as choosing from selections on drop-down lists, as with the FSWEPP online interface. Many tools, especially those designed as distributed, process-based systems, generally running in a GIS, may require large amounts of data from multiple data sources. There is a trend to build interfaces that guide the user to electronic libraries of public data to simplify data acquisition. This approach is especially valuable where users are building models for watersheds for which location-specific data are incomplete or not available. Alternatively, wizards that interactively guide users through each step of the model building process (for example, DHSVM) provide a framework to ensure that all components are in place for a model run.

The outputs from a given tool should provide the information the manager requires. Some models developed for non-forested conditions may not give the information that the user desires or in a satisfactory format. Output files containing more information than a user requires may need additional analyses to reformat or synthesize results.

The documentation is an extremely important part of any tool. In some cases, documentation is readily available online and includes many illustrations and examples. In other cases, documentation is inadequate, and managers should lobby model developers to provide improved, appropriate documentation if the software meets the manager's needs.

The availability of technical assistance is often important, particularly with more complex models. Some technical assistance may be built into the model, with help screens incorporated into the interface. In some cases, local or regional experts may be available to provide such assistance. In others, specialists in other agencies, such as the NRCS, Bureau of Reclamation, Army Corp of Engineers, university specialists, federal researchers, or consultants, may be available to provide the necessary assistance.

It is unlikely that any watershed tool will meet all of the manager's needs. Managers will have to select from a series of tools that best meet their needs and organization's abilities. Also, not every CWE analysis requires or merits the complexity possible with some of the currently available tools. New tools with new features and capabilities continue to be developed, and a career as a watershed manager will be one of continued learning and evaluation.

Categories of Models

Cumulative effects models can be categorized into lumped or distributed, conceptual or physically based, or deterministic or stochastic.

Lumped vs. Distributed Models

Some models assume that an entire watershed is behaving as a single unit and assume that all of the inputs can be lumped into a single set of variables to describe the entire watershed (fig. 1). The outputs are generated as runoff and/or sediment yield at the watershed outlet. The Rational Peak runoff method and the Curve Number runoff and peak flow models are common examples of lumped models (Ward and Elliot 1995).

Distributed models allow the user to vary model inputs and site characteristics in space. There can also be interactions between cells, modeling the "runon-runoff" processes common on disturbed forest hillslopes. Hot spots for sediment sources can be identified to focus management, as can environmentally benign areas where management can be more flexible. The GeoWEPP tool is an example of a distributed model using hillslope polygons and the DHSVM model is an example of a distributed model using grid cells. Many models use combinations of lumping and distributing. For example, soils may be distributed by grid, hillslope topography by polygon, and climate may be lumped for the entire area. One common approach to distributed modeling is to use hydrologic response units (HRUs) as the smallest unit of discretization. An HRU may be a small watershed or a hillslope polygon, depending on the focus of the analysis (Beighley and others 2005).

Some form of distributed modeling is essential for estimating sediment detachment and delivery. The data needs, however, may be great, and there may not be adequate information available to describe variations in soil properties including soil depth and surface residue cover. Distributed models are also difficult to calibrate as there are many cells that can be adjusted in order to obtain a reasonable prediction. The benefit of distributed modeling is that more management options can be evaluated. For example, a manager can compare the watershed impacts of thinning only the upper part of hillslopes instead of the entire hillslope or the watershed effects of altering the width of an undisturbed forested buffer along stream channels. Management affects on north-facing slopes can be compared to those on south-facing slopes. In larger watersheds, it may be possible to synchronize or desynchronize hydrographs to obtain desired runoff characteristics. Lumped models would not be as versatile at modeling such spatial variability.

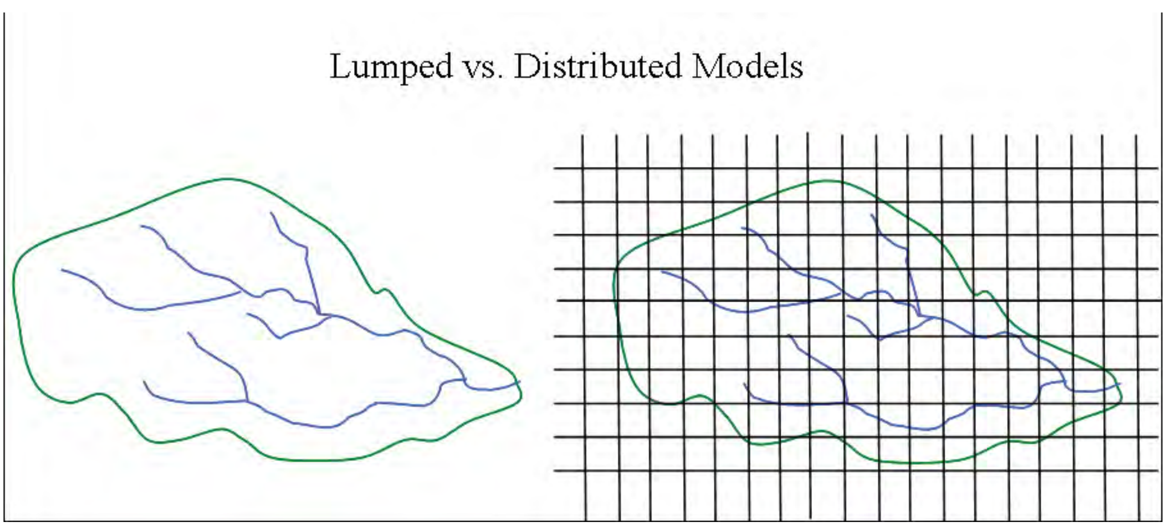

Figure 1. Lumped vs. distributed model. The "lumped" watershed on the left has a single value to describe soil, vegetation, and climate conditions in the entire watershed, whereas the "distributed" watershed on the right may have different values for each grid cell, or in some cases, individual hillslope polygons.

A current effort in distributed modeling is to incorporate road networks and their impacts into watershed analysis. This technology has the potential to include the impacts of roads as sources of sediment and runoff and also to evaluate the effects of roads on fish passage (RMRS 2007).

Conceptual vs. Physically Based Models

Conceptual models are based on the physical processes that drive the watershed responses. Physically based models contain mathematical equations and relationships that describe watershed processes. Few models can be labeled as purely conceptual or purely physical, but rather range along a spectrum of complexity from purely conceptual to purely physical. The more conceptual or empirical models require less data, but are less flexible in the application. The more physically based models can often be applied to a greater variety of circumstances, but in order to do this, they will be more computationally intensive and require larger input data sets.

There is often a heavy reliance on empirical data in conceptual models. They tend to be lumped or only partially distributed. Because of their reliance on observed data, they should not be extrapolated to conditions beyond those that were used in their development. Examples of conceptual models are the Universal Soil Loss Equation (USLE) and the Curve Number technology.

Physically based processes may include:

- vegetation growth and senescence;
- impacts of plant community on evapotranspiration, rainfall and/or snow interception, and soil stability;
- soil water balance and subsurface water movement;
- sediment detachment, transport, and deposition by raindrop splash, shallow overland flow; concentrated rill flow, gullying, and channel processes; and
- mass failure and debris flow processes.

Input parameters for physically based models are generally variables that can be measured or derived from measurements of physical or biological processes, such as topography, runoff rates, biomass amounts, and surface cover. Physically based models are generally more academically acceptable, and can be generally applied to areas other than where the original data used for model development were collected. Data needs, however, may exceed what is readily available in areas beyond the sites where the models were originally developed. Examples of physically based models include the Water Erosion Prediction Project (WEPP) and the DHSVM model.

Deterministic vs. Stochastic Models

A deterministic model will always give the same outputs for the same set of input variables. A stochastic model generally has at least one probabilistic input and will give a result that describes the risk or likelihood of a given prediction. Examples of stochastic models include the climate generator for the WEPP model or the return period analyses associated with peak runoff events from the Curve Number technology.

Lump-Based Runoff Tools

There are two runoff prediction technologies that are generally lump-based: the Rational Peak Flow prediction, and the NRCS Curve Number runoff volume and peak flow prediction (Ward and Elliot 1995). These technologies are frequently incorporated into other higher level hydrologic models and a description of each will be given.

Rational Peak Flow

The Rational Peak Flow prediction method is (Schwab and others 1993):

$$q = 0.0028\, C\, i\, A \tag{1}$$

where q = design peak runoff rate (m^3/s)
C = runoff coefficient
i = rainfall intensity for the design return period and for a duration equal to the "time of concentration" of the watershed (mm/h)
A = area of watershed (ha)

The runoff coefficient C is a function of vegetation and rainfall intensity (Schwab and others 1993) and for forests, ranges from 6 to 20 (table 3). Users should check with local NRCS or state agency users to determine local values. Some typical values are presented in table 3. The time of concentration for a given watershed is often estimated by the Kirpich equation (Schwab and others 1993):

$$T_c = \frac{L^{0.77}}{3077 s^{0.385}} \tag{2}$$

where T_c = time of concentration (h)
L = distance from watershed divide to watershed outlet (m)
s = watershed gradient (m/m)

When applying the Rational method, the time of concentration is frequently a relatively small number, typically less than 1 hour. To estimate the peak intensity for the desired duration storm, the NOAA Precipitation-Frequency Atlas is often consulted (Bonnin and others 2003). There are Internet sites available that aid the user in determining the intensities of these shorter duration storms for many states (http://hdsc.nws.noaa.gov/hdsc/pfds/). The Rational method is best suited to watersheds under about 2 mi^2. For larger watersheds, other runoff methods are recommended. One of the reasons the runoff coefficient is so small for forest watersheds is that much of the runoff is shallow subsurface lateral flow or groundwater, rather than surface runoff, so peak rates are less than would occur on agricultural sites of similar area.

Table 3. Typical values for Runoff Coefficient *C* in Rational Equation and descriptions of hydrologic soil groups (Engel and others 2009).

	Hydrologic Soil Group			
Land Use, crop and management	**A**	**B**	**C**	**D**
Cultivated with crop rotations				
Row crops, poor management	55	65	70	75
Row crops, conservation mgmt	50	55	65	70
Small grains, poor mgmt	35	40	45	50
Small grains, conservation mgmt	20	22	25	30
Meadow	30	35	40	45
Pasture, permanent with moderate grazing	10	20	25	30
Woods, permanent, mature, no grazing	06	13	16	20
Urban residential				
30 percent of area impervious	30	40	45	50
70 percent of area impervious	50	60	70	80

Hydrologic Soil Group Descriptions:
A -- Well-drained sand and gravel; high permeability.
B -- Moderate to well-drained; moderately fine to moderately coarse texture; moderate permeability.
C -- Poor to moderately well drained; moderately fine to fine texture; slow permeability.
D -- Poorly drained, clay soils with high swelling potential, permanent high water table, claypan, or shallow soils over nearly impervious layer(s).

Curve Number Runoff Volume and Peak Flow

The Curve Number runoff technology can be used to predict both runoff volume and peak runoff rate (Fangmeier and others 2006). The Curve Number technology was initially developed from a network of small watersheds covering the entire United States. Most of these watersheds were in agricultural areas, so data for rangelands and forests was limited. The USDA Natural Resource Conservation Service (NRCS) played a leading role in the development of the research technology.

Estimating total storm runoff with curve number

The first step in both total runoff and peak flow is to estimate the total runoff depth *Q*. The Curve Number method uses two foundation equations to estimate *Q*:

$$Q = \frac{(I - 0.2S)^2}{I + 0.8S} \tag{3}$$

and

$$S = U\left[\left(\frac{1000}{CN}\right) - 10\right] \tag{4}$$

where Q = runoff depth (mm or inch)
I = storm depth (mm or inch)
S = maximum potential difference between rainfall and runoff, sometimes referred to as surface storage (mm or inch)
U = unit conversion (25.4 mm for metric, 1 inch for English units)
CN = NRCS Curve Number for soil and cover condition (see table 4)

In watersheds with mixed cover, an area weighted average is generally employed to estimate the curve number (Fangmeier and others 2006).

Table 4. Some typical Curve Number values for forested conditions (Goodrich and others 2005).

Cover type	Ground cover (%)	Hydrologic Soil Group (see table 3 for descriptions) A	B	C	D
Bare	0	77	86	91	94
Fallow	5	76	85	90	93
Shrubland	25	63	77	85	88
Grassland/Herbacious	25	49	69	79	84
Undisturbed Forests					
Deciduous & Mixed	50	55	55	75	80
Evergreen	50	45	66	77	83
Forest, Low severity fire					
Deciduous & Mixed	43	59	60	78	82
Evergreen	43	49	71	80	85
Shrubland	21	65	79	86	89
Moderate severity fire					
Deciduous & Mixed	34	65	65	80	85
Evergreen	34	55	76	82	88
Shrubland	17	68	82	88	90
High severity fire					
Deciduous & Mixed	25	70	71	83	87
Evergreen	25	60	82	85	90
Shrubland	12	73	88	91	91

Estimating peak runoff rates with curve number

From the runoff amount Q, peak runoff can be estimated using the methodology developed by the NRCS (2002). This manual method has since been incorporated into numerous public and private software programs, several of which are discussed later in this chapter. One method that can be readily adapted to local conditions for applying the Curve Number technology (Schwab and others 1996) is to first estimate the time of concentration with the empirical relationship:

$$T_c = \frac{L^{0.8}\left(\frac{1000}{CN} - 9\right)^{0.7}}{C\, s^{0.5}} \tag{5}$$

where Tc = time of concentration (hours)

L = length of watershed (m or ft)
CN = NRCS Curve Number
C = constant 441 for metric, 1,140 for English units)
s = average watershed gradient (m/m or ft/ft)

Other methods commonly used to estimate time of concentration generally require the user to estimate the runoff velocity overland and in channels, and the lengths and slopes of the overland area and the channel. Numerous public and proprietary software programs assist in this calculation. Once the time of concentration is estimated, the NRCS has developed a series of curves to estimate peak runoff rate as a function of total storm runoff. There are numerous software programs that have incorporated these curves into the software itself. One relationship between time of concentration, peak runoff rate, and total runoff is (Schwab and others 1996):

$$\log(q) = 2.51 - 0.7 \log(Tc) - 0.15 (\log(Tc))^2 + 0.071 (\log(Tc))^3 \tag{6}$$

where q is peak runoff rate in cubic feet per second per square mile of watershed area per inch of storm runoff. For metric units, multiply this number by 0.0043 to get cubic meters per second per square km of watershed area per mm of storm runoff.

Limitations of the Curve Number method

Recent observations of runoff volumes and rates from forests before and after wildfire have shown that the runoff volume appears to change little; the time of concentration is generally in the magnitude of days for undisturbed forests and minutes to hours for forests following wildfire (Canfield and others 2005). One analysis suggested that the Curve Number method was not appropriate for forest or rangeland watersheds, and a better estimate of runoff is as a fraction of precipitation based on field observations (Springer and Hawkins 2005). For thinned or prescribed fire conditions, the time of concentration is likely to be longer than would normally be estimated for overland flow for non-forest conditions as most of the runoff is from subsurface lateral flow. In the past, watershed managers often reduced the Curve Number to reduce peak flow rate estimates from forests compared to non-forested areas rather than increase the time of concentration values. This area warrants further research.

USLE-Based Tools

In the period from 1945 until 1965, a method of estimating soil erosion based on statistical analyses of field plot data from small plots located in many states was developed, which resulted in the Universal Soil Loss Equation (USLE) (Wischmeier and Smith 1978). The USLE was parameterized for some forest hillslope conditions for intensive forest management practices in the southeastern United States (Dissmeyer and Foster 1981). A revised version of the USLE (RUSLE) was later developed as a computer application (Renard and others 1997). When predicting erosion, RUSLE allows a more detailed consideration of management practices, rangeland, seasonal variation in soil properties, and topography than does the USLE. In 2005, RUSLE2 was released as an application for the Windows operating system (Foster and Toy 2003). The basic form of the USLE/RUSLE models is:

$$A = R\,K\,L\,S\,C\,P \tag{7}$$

where
A = average annual soil loss (tonnes/ha/year or tons/acre/y)
R = rainfall and runoff erosivity factor for a geographic location
K = soil erodibility factor
L = slope length factor
S = slope steepness factor
C = cover management factor
P = conservation practice factor

The A, R, and K factors are different for English and metric units. To avoid confusion, this chapter will use all English units. Metric units for these three variables can be found in Fangmeier and others (2006). The other factors have no units. The L and S factors are used in some other models, including the WATSED cumulative effects model (USFS 1990).

USLE Factors

R factor

The R factor is based on the rainfall intensity and energy for a given location (Renard and others 1997). Figure 2 is the "isoerodent" map for the western United States in English units, providing an estimate of the R factor. The west coast and eastern United States maps, as well as other methods for estimating R, can be found in Renard and others (1997). The R factor is best suited for climates where runoff and erosion are dominated by large storms, not snow melt, whereas much of the area in figure 2 is dominated by snow process, making these areas problematic for USLE applications.

K factor

The K factor is generally estimated from soil properties. The equation developed for agricultural soils is:

$$K = \frac{\left[2.1 \times 10^{-4}(12 - OM)M^{1.14} + 3.25(s-2) + 2.5(p-3)\right]}{100} \tag{8}$$

where
K = soil erodibility (English units)
OM = organic matter (percent)
M = particle size fraction in soil between 0.001 and 0.1 mm, or percent silt plus percent fine sand (percent)
s = subsoil structure class: 1 – very fine granular
2 – fine granular
3 – med or coarse granular
4 – blocky, platy or massive
p = permeability class: 1 – rapid
2 – moderate to rapid
3 – moderate
4 – slow to moderate
5 – slow
5 – very slow

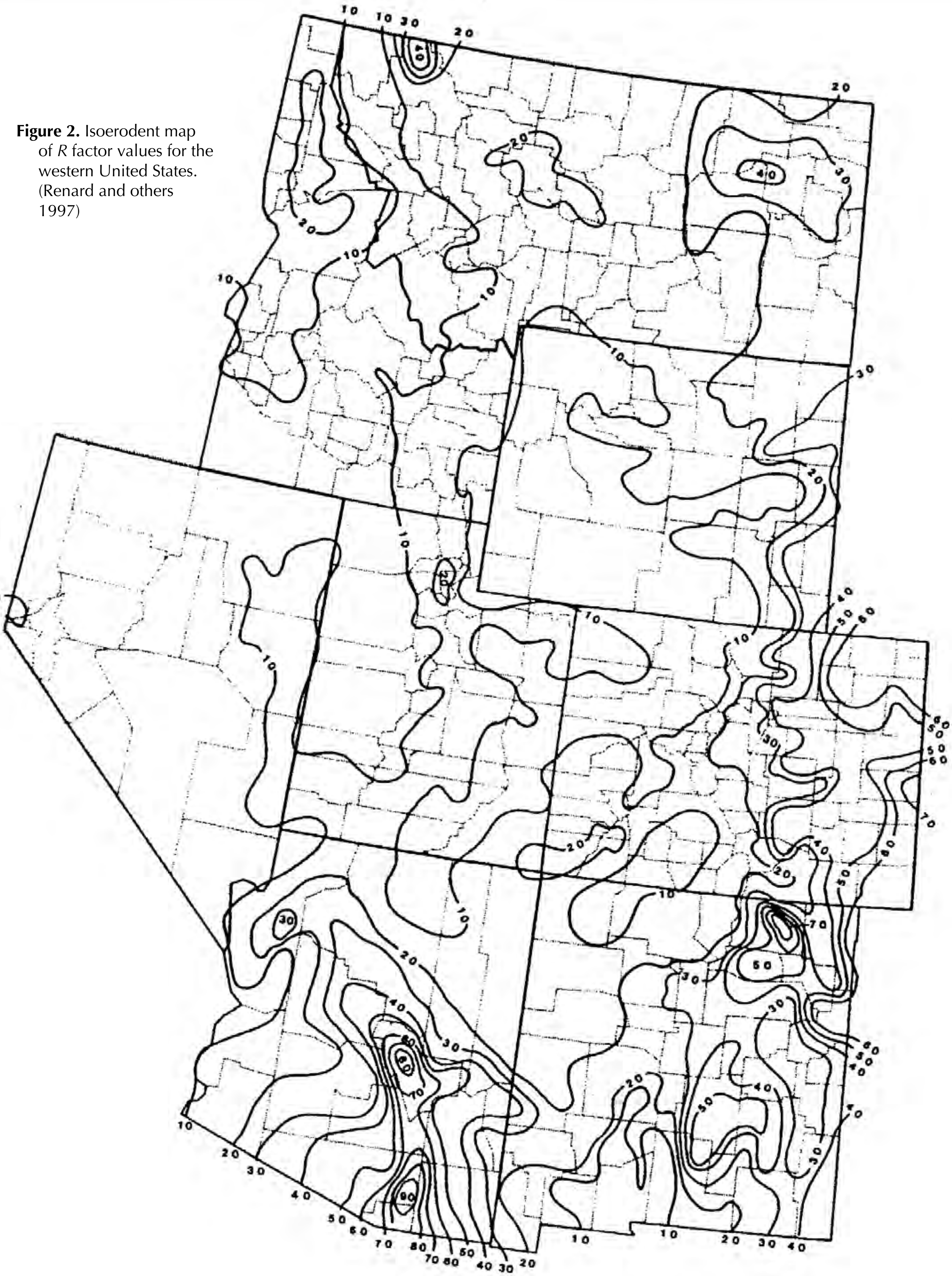

Figure 2. Isoerodent map of *R* factor values for the western United States. (Renard and others 1997)

In forest environments, soil erodibility has been found to be a function of not only soil properties, as assumed in the USLE technologies, but the vegetation condition. Vegetation condition in the USLE technologies is addressed in the *C* factor. Because of this interaction, users should be cautious when applying the USLE to forest conditions to ensure that compatible *C* and *K* factors are used. It is not advisable to obtain a *K* factor estimate from one source and a *C* factor estimate from another.

L and S factors

The topographic factors, *L* and *S*, adjust the predicted erosion rates to give greater erosion rates on longer and/or steeper slopes when compared to the USLE "standard" slope steepness of 9 percent and length of 72.6 ft (22 m). In many mountainous conditions, the lengths and steepness values are greater than intended for the *L* and *S* factors. These factors address the increasing rill erosion rates as more runoff accumulates with longer slopes and the hydraulic shear in runoff increases on steeper slopes. The methods for estimating *L* and *S* factors were modified in RUSLE, and it is advisable to use the RUSLE/RUSLE2 method for estimating these topographic factors rather than the USLE method.

The RUSLE *L* factor can be calculated as:

$$L = \left(\frac{l}{72.6}\right)^{b} \tag{9}$$

where L = slope length factor
l = slope length in ft
b = dimensionless exponent

For conditions where rill and interrill erosion are about equal on a 9 percent, 72.6-ft long slope

$$b = \frac{\sin\theta}{\sin\theta + 0.269(\sin\theta)^{0.8} + 0.05} \tag{10}$$

where θ = field slope angle = $\tan^{-1}(S)$
S = slope steepness (ft/ft)

For most conditions where rill erosion is greater than interrill erosion (such as soils with a large silt or fine sand content), *b* should be increased up to 75 percent. Where rill erosion is less than interrill erosion (on short slopes), *b* should be decreased as much as 50 percent. RUSLE2 makes this calculation internally.

The *S* factor depends on the length and steepness category of the slope. For slopes less than 15 ft long:

$$S = 3.0\ (\sin\theta)0.8 + 0.56 \tag{11a}$$

For slopes greater than 15 ft long and steepness less than 9 percent

$$S = 10.8 \sin\theta + 0.03 \tag{11b}$$

For slopes greater than 15 ft long and steepness greater than or equal to 9 percent:

$$S = 16.8 \sin\theta - 0.50 \tag{11c}$$

For the USLE, the slope length is measured from the point where soil erosion begins (usually near the top of the ridge) to the outlet channel or a point downslope where deposition begins. RUSLE also considers non-uniform concave or convex slopes. RUSLE2 considers the entire hillslope, including areas of deposition.

C factor

The *C* factor is the cover management factor, sometimes referred to as the cropping factor. This factor was originally developed to allow users to specify the cover condition for every 2-week period in a rotation. A rotation may last for several years, and an extended calculation is necessary (Ward and Elliot 1995). The RUSLE technologies frequently add subfactors to the estimate of *C*, further complicating this critical calculation. This methodology was developed in order to consider the surface condition during any 2-week period in relation to the climate during that same period. In forest and rangeland conditions; however, it is much more appropriate to think of the term as cover management and consider *C* as constant for the entire year. Runoff and soil erosion are dominated by ground cover, and when using USLE technology, it is important to ensure that the correct term is selected. Wischmeier and Smith (1978) developed a set of *C* factors for forest and rangeland conditions that were a function of canopy height and cover and ground cover. For forest management activities in the southeastern United States, Dissmeyer and Foster (1981) expanded the Wischmeier set to include site preparation tillage practices common in the Southeast. Because tillage is not generally associated with fuel management, but ground cover disturbance is, the Wischmeier and Smith (1978) *C* factors are presented in table 5 for forested conditions and table 6 for burning. The RUSLE1 technology can calculate a *C* factor internally from a fixed cover condition, and a similar "permanent vegetation" option is available in RUSLE2.

Table 5. USLE C factors for forest conditions (Wischmeier and Smith 1978).

Percent of area covered by canopy of trees and undergrowth	Percent of area covered by duff at least 50 mm (2 in.)	C Factor
100 – 75	100 – 90	0.0001 – 0.001
70 – 45	85 – 70	0.002 – 0.004
40 – 20	70 – 40	0.003 – 0.009

Table 6. USLE C factors for burning (Wischmeier and Smith 1978).

Ground cover (percent)	Soil condition	
	Excellent to good	Fair to poor
10	0.23 – 0.24	0.26 – 0.36
20	0.19	0.21 – 0.27
40	0.14	0.15 – 0.17
60	0.08 – 0.09	0.10 – 0.11
80	0.04 – 0.05	0.05 – 0.06

Recent Variations on the USLE

In the RUSLE2 technology, sediment delivery across buffers is predicted as a function of runoff estimated by the Curve Number method. The *C* factor in RUSLE is based on a weighted average of vegetation cover throughout a growing cycle and takes into account prior land use, canopy cover, surface cover, surface roughness, and soil water content.

Some variations of the USLE have been developed to make erosion estimates for individual storms. This may be done by considering the *R* factor for an individual storm or with the Modified USLE (MUSLE) technology described below.

The factors in RUSLE have generally been developed from, and validated by, research studies on tilled agricultural soils. Some rangeland research with RUSLE developed *C* factors based on surface cover, which give reasonable erosion predictions for rangeland hillslopes (Elliot 2001). Applications of RUSLE to disturbed forest hillsides and roads have been limited.

MUSLE

The modified USLE (MUSLE) replaces the *R* factor with the product of rainfall amount and runoff amount to predict soil erosion for a single storm. Most applications of the MUSLE technology use Curve Number to estimate the runoff. The other USLE factors (*K*, *L*, *S*, *C*, and *P*) remain unchanged for MUSLE applications. Because MUSLE is relatively easy to program, it has been incorporated into numerous soil erosion models in recent years. Its limitations are similar to those of the two technologies that drive it.

AGNPS, AnnAGNPS: Agricultural Non-Point Source

The AGNPS erosion model is a distributed parameter tool for moderate- to small-sized agricultural watersheds (Bingner and others 2007; Suttles and others 2003). AGNPS and its newer iteration, AnnAGNPS, are well established, actively supported, production-ready tools. AGNPS models a single event, simulating a pulse of sediment from an individual storm. AnnAGNPS extends the modeling into continuous, annual outputs. The system is driven by three core technologies: erosion modeled by RUSLE, hydrology by the NRCS Curve Number method, and sediment/contaminant transport using CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems). Additional AGNPS modules include a channel network evolution model (CCHEID) and stream corridor model (CONCEPTS), components that emphasize the condition of water within stream channels. Snowmelt is not modeled. Both versions run within GIS shells and are challenging to apply. AGNPS is fundamentally an agricultural tool with limited utility in forested mountainous environments.

SWAT: Soil and Water Assessment Tool

The SWAT modeling tool predicts the impact of land management practices upon water, sediment, and chemical yields in large, complex watersheds (Arnold and others 1998; Di Luzio and others 2002, 2004). SWAT is a distributed model with linked modules using both process and empirical logic. Key climate and vegetation simulations are process driven while the core hydrology and sediment modeling includes use of an enhanced NRCS Curve Number approach to estimate runoff volume and USLE and MUSLE technologies to estimate soil erosion. SWAT has a long development history, strong institutional commitment, and a substantial publishing record based upon worldwide application. Comprehensive documentation and ease of access, download, and installation permit users to efficiently load and execute the SWAT tool. Although originally developed for agricultural applications, the developers have demonstrated a commitment to improving mountain hydrology, including implementing advances in topographically driven snowmelt processes.

SWAT is a user-friendly, if complex, modeling system that could be a valuable tool for CWE analysis. The existing system requires addition of more refined forest practice definitions. One of the big challenges when applying SWAT to forested watersheds is that SWAT currently estimates all runoff with Curve Number technology. Generally, users will calibrate the Curve Number for SWAT for a given watershed from observations from a nearby watershed. As most forest runoff is dominated by lateral flow, and many by snowmelt hydrology, this will be a major limitation to applying SWAT for steep forested watersheds.

Rule-Based Tools

R1-WATSED and Derivatives—Water and Sediment Yields

The USDA Forest Service, Region 1 WATSED program is a cumulative watershed effects tool designed to model watershed response to multiple management activities and disturbances over time (USFS 1990). It estimates and tracks changes in annual water and sediment yields, mean monthly flows, delivery of total sediment to a defined stream reach, and relative annual changes to sediment delivery within the stream network. WATSED models vegetation and hydrologic recovery, past and assumed future activities, background erosion from hillslopes, and surface and mass erosion from activities including roads.

WATSED is an empirical model driven by locally derived and calibrated coefficients and recovery response curves. The model assigns lumped parameters to each land area modeled (landtype units within watersheds) using a linked table structure. Currently there is no GIS interface, but prototypes are under development. Changes to vegetation cover are modeled and expressed as Equivalent Clearcut Areas (ECAs). The ECA concept was developed in the 1970s as a method for estimating the change in runoff amounts and peak flows associated with a forest practice. It assumes, for example, that 2 acres of partially cut forest will have the same affect on water yield as 1 acre of clearcut forest. As a forest regrows, the ECA is reduced (Ager and Clifton 2005). WATSED has a Windows interface to guide the user through all stages of the model application. WATSED spawned several derivatives, each customized for conditions within a given national forest, including LoloSED, NezSED, and BoiSED for Lolo, Nez Perce, and Boise National Forests, respectively.

State variants of WATSED

Several states, Washington and Idaho in particular, have developed state-specific lookup tables to apply to the WATSED technology. Examples are Washington Forest Practices and Idaho Cumulative Watershed Effects tools. A computer application of the Washington Forest Practices, called WARSEM, has been developed for Washington State.

SEDMODL2

SEDMODL2 is a GIS-based road erosion and sediment delivery model designed to identify road segments with a high potential for delivering sediment to streams (Dubé and McCalmon 2004). The model is based on empirical relationships developed by the Washington State Forest Practices manual. SEDMODL2 estimates annual background sediment and sediment production by individual road segments, locates road/stream intersections, and estimates delivery of road sediment to streams. Developers provide core climate data for several Western States and optional base geology data for Washington, Oregon, and Idaho. For most of the variables that define each road segment, users may choose default values or define attributes with locally available data. SEDMODL2 has an interactive Windows interface and well-written documentation.

Delta-Q and FOREST (FORest Erosion Simulation Tools)

Delta-Q and FOREST are complementary Cumulative Watershed Effects tools. These tools are intended to provide managers with estimates of relative cumulative changes in forested watershed responses due to multiple management activities over time (MacDonald and others 2004). Development of these tools is in part driven by the intent to move beyond the basic Equivalent Clearcut Area (ECA) approach. The general structure uses GIS-based, two-dimensional spatial representations as an organizing shell to calculate cumulative impacts and watershed recovery across multiple treatment areas. Delta-Q uses an empirical approach driven by curves developed from 26 paired watershed datasets to model changes to water yields from disturbed forested areas. FOREST predicts changes to sediment regimes from hillslopes and roads in three integrated tools that calculate sediment production, delivery, and eventually, in-stream routing. Hillslope calculations are based on user-provided data defining hillslope erosion response and recovery rates. Hillslope delivery to streams uses database files derived using the WEPP model. The database files are included in the FOREST distribution package and instructions are included so that the user can also use WEPP to customize a file for his or her own region. Sediment changes from roads are calculated using one of three user-chosen methods: one of two empirical equations or via look-up tables provided with the core model or provided by the user. The look-up tables may incorporate local knowledge or may be developed using an outside program such as WEPP:Road. Road delivery to streams is based on an empirical relationship between percent connectivity and mean annual precipitation.

Delta-Q and FOREST provide easy to use, straight-forward, and simple approaches to assessing relative cumulative change. The user must adjust basic response settings for local conditions. The empirical approach of Delta-Q could limit applicability where the area of concern is distant from one of the 26 experimental sites and there is insufficient basis to determine which experimental site most closely matches the area of concern. In FOREST, model calculations for activity areas and roads are not linked. The spatial interface provides a convenient and efficient means to assess possible changes.

WARMF (Watershed Analysis Risk Management Framework)

WARMF is a lumped parameter GIS model that uses USLE-based erosion prediction and Curve Number technology for surface runoff. WARMF includes groundwater flow and estimates nutrient and bacterial loads. Because of its lumped nature, it does not lend itself to project scale management. It also has the limitations of the USLE and Curve Number technologies for forest conditions. It is one of the few models available that addresses nutrients and bacterial loads (http://www.epa.gov/ATHENS/wwqtsc/html/warmf.html).

Physically Based Tools

Physically based models predict runoff and/or soil erosion from equations that generally describe the processes that are occurring, such as infiltration, runoff and subsurface water flow and soil detachment, transport, and deposition.

WEPP

The Water Erosion Prediction Project (WEPP) model is an interagency physically based hydrology and soil erosion model (Flanagan and Livingston 1995). It can be run for individual hillslopes or for a small watershed (up to about 2 mi^2). The WEPP technology includes a Windows interface, a GIS interface, several online interfaces, and a stand alone executable program with text file input and output files that can be incorporated into other applications. Most applications of WEPP include databases of soils, climates, and vegetation descriptors.

CLIGEN

The CLIGEN weather generator and database of statistics from more than 2,600 weather stations is part of the WEPP technology. Within the WEPP Windows interface is a feature to allow users to alter the statistics for the climate they have selected to match a specific site. Additional climate data can be downloaded from an online web site (http://forest.moscowfsl.wsu.edu/fswepp/) that has a 4-km (2.5-mile) grid of monthly precipitation values for the continental United States This site also allows a user to modify temperature and precipitation data if local records are available.

WEPP Hillslope version

The Hillslope version of WEPP predicts runoff, onsite erosion, and offsite sediment delivery for a hillslope. Interfaces have been developed to run this version from either Windows or over the Internet.

WEPP Windows. The WEPP Windows hillslope interface predicts erosion from single hillslopes or from lists of hillslopes in project mode. Users may alter any of the hundreds of input variables required to run WEPP. This is a highly flexible interface, but users seldom have all of the inputs necessary to build input files. Templates are included in the interface for a wide range of forest conditions, including those needed for cumulative watershed effects of fuel management.

Internet Suite of Tools. Both the Forest Service and the Agricultural Research Service have developed Internet interfaces for the WEPP model (table 7). The Forest Service suite of interfaces (FS WEPP) is available for forested hillslope applications (Elliot 2004), and the ARS interfaces have a greater emphasis on agricultural or rangeland conditions.

WEPP FuMe is specifically designed to support fuel management activities. It carries out 12 runs from a single set of inputs. These runs include nine hillslope scenarios: undisturbed forest; low, moderate, and high impact thinning; low, moderate, and high impact prescribed fire; and low and high intensity wildfire. Additionally, there are three road runs for low, moderate, and high levels of traffic. The output page provides tables and a narrative of the results of those runs in the context of fuel management activities to assist the user in synthesizing the results.

WEPP Watershed Models

WEPP Windows contains a watershed option. WEPP Watershed combines hillslopes, channels, and instream structures such as check dams, which it calls "impoundments." The current version predicts peak flow using a variation of the rational equation, which limits the size of a watershed to less than 2 mi^2 if users desire to consider predictions of peak runoff rates.

At the WEPP hillslope scale, only surface runoff is considered, whereas both surface runoff and subsurface lateral flow are considered when modeling watersheds. More than 90 percent of the runoff from many steep forested watersheds is from shallow lateral flow or groundwater flow (Conroy and others 2006; Covert and others 2005; Dun and others 2006; Zhang 2006).

Table 7. Online WEPP interfaces.

Name	Location	Features
WEPP:Road	http://forest.moscowfsl.wsu.edu/fswepp/	Road erosion and sediment delivery from individual segments
WEPP:Road Batch	http://forest.moscowfsl.wsu.edu/fswepp/	Multiple road segments for a given soil and climate
Disturbed WEPP hillslopes	http://forest.moscowfsl.wsu.edu/fswepp/	Disturbed forest and rangeland
WEPP FuME	http://forest.moscowfsl.wsu.edu/fswepp/	Multiple WEPP runs to support fuel management planning
ERMiT	http://forest.moscowfsl.wsu.edu/fswepp/	Post wildfire sediment delivery and mitigation analysis
WEPP Web Interface	http://milford.nserl.purdue.edu/	Agriculture and rangeland erosion, detailed graphics
WEPP CAT	http://typhoon.tucson.ars.ag.gov/weppcat/index.php	Ag, and range Climate Assessment Tool including buffers

WEPP Watershed Windows Interface. The WEPP Watershed Windows interface is difficult to use but it allows the user to alter all of the variables necessary to run a watershed analysis, including properties of channels and impoundments. Building a watershed within WEPP Windows is an arduous task for large landscapes. It is best suited to modeling engineered sites, such as ski slopes, parking lots, or agricultural terraces, where dimensions and grades of hillslope planes and channels are generally well defined. For natural watersheds, users should use the GeoWEPP technology to build watersheds.

GeoWEPP. GeoWEPP is a GIS wizard that builds files to run WEPP Watershed on either ArcView or Arc 9.x platforms. Topographic analysis tools build the slope files for both hillslope polygons and channels from a Digital Elevation Model (DEM). The same soil, climate, and management databases that are used by WEPP Windows are used by GeoWEPP. Generally, input files are created, edited, and tested in WEPP Windows so they can be accessed by GeoWEPP. GeoWEPP can also read text files that describe the soil or vegetation in each grid cell, which may enhance its application for interpreting GIS maps of fuel management activities within a watershed.

DHSVM: Distributed Hydrology, Soil, and Vegetation Model

DHSVM is a distributed, physically based hydrologic tool that prepares the data with the aid of a GIS and then runs the model using command line codes in a Unix interface. Recently, it has incorporated sediment detachment and transport as a function of surface runoff. It was developed at the University of Washington in conjunction with the USDA Forest Service, Pacific Northwest Research Station. DHSVM was specifically designed to address complex hydrologic interactions and variability due to climate and topography. It was originally developed to assess changes in flow resulting from logging at relatively large scales. DHSVM (Version 3) models hydrologic processes within vegetation; surface and subsurface flow; management activities; road networks incorporated into hillslope and stream channel connections; saturation induced mass wasting and redistribution; hillslope erosion driven by saturation excess runoff and rainfall and leaf drip detachment; road surface erosion including integration with road side ditches and culverts; sediment delivery to stream networks; routing through channels; water discharge accounting for contributions from overland and subsurface flows; and sediment deposition, storage, and transport within stream channels based upon channel geometry, water flux, and particle sizes of delivered debris.

DHSVM is primarily a research tool relying on an understanding of GIS and UNIX command line code. A data assembly wizard assists with preparing input information. With further development, DHSVM could provide support for a comprehensive CWE analysis. Access further information on-line at: http://www.hydro.washington.edu/Lettenmaier/Models/DHSVM/.

SMR: Soil Moisture Routing Model

SMR is a physically based distributed hydrologic model that uses simple "map calculation" commands within grid-based GIS software packages (in other words Arc/INFO, GRASS) to represent the hydrology of a landscape. The model, originally developed at Cornell University (Brooks and others 2007; Frankenberger and others 1999; Johnson and others 2003), was designed as a simple management tool to simulate spatially distributed soil water, surface runoff, subsurface lateral flow, and streamflow using publicly available data and requiring minimal calibration. Since the program uses commands inherent in nearly all available GIS software packages, the source code is a very simple batch file or script file (only a few pages long) that is easy to read and modify. Although the model does not include many of the complex algorithms in the DHSVM model, such as variability in aerodynamic and canopy resistance within multiple layers of the canopy and corrections for atmospheric stability to calculate evapotranspiration, the fundamental hydrologic mechanisms used to route subsurface lateral flow and generate saturation-excess runoff are very similar to those in the DHSVM model. Despite these simplifications, the model has been shown to provide good agreement with distributed soil moisture, perched water tables, and snow water equivalent measurements as well as stream flow and spatial surface runoff patterns. The model has been used in agricultural dominated watersheds to identify critical management zones associated with nutrient and pathogen transport. In forestry, SMR applications have helped to identify landslide susceptibility (Gorsevski and others 2006b) and quantify the effects of climate change on regional water supply and streamflow (Mehta and others 2004). Further information is available at: http://soilandwater.bee.cornell.edu/Research/smdr/index.html.

Summary of Watershed Tools

Table 8 provides a summary of currently available tools that have been applied to watershed analysis. All of these tools can be run within a GIS framework, some more easily than others. Most tools are based on the empirical USLE/Curve Number technologies, with the exception of WEPP and DHSVM. The main differences among the empirical models are the spatial detail allowed in the hillslope description and how the sediment is routed through the stream system. There are also differences in the availability of databases for forest cumulative watershed effects and other support that is available for the model. Currently, WEPP has the best database for modeling cumulative watershed effects. The empirical technologies have somewhat easier interfaces to use but do not correctly model the dominant forest hydrologic and erosion processes.

Table 8. Summary of currently available watershed modeling tools.

Tool name	Predicts	Empirical / process	Status
AGNPS	Runoff and erosion	Emp	Production
Delta-Q / FOREST	Runoff and erosion	Emp	Production/Beta
DHSVM and SMR	Runoff	Process	Research
Rational	Runoff	Emp	Production
NRCS Curve Number	Runoff	Emp	Production
SedMODL2	Runoff and erosion from roads	Emp	Production
SWAT	Runoff and erosion	Emp	Production
WARMF	Runoff and erosion	Emp	Production
WATSED	Runoff and erosion	Emp	Production
WEPP	Runoff and erosion	Process	Production

Other Modeling Tools Available

These tools are considered separately because, to date, they are commercial experimental research tools and, while they are developed and popular among the water and sediment modeling community, their intended uses may vary significantly from CWE applications. With many of these models, additional training or consultant support is needed before applying the model.

NetMap

NetMap is an integrated suite of numerical models and analysis tools created for three purposes: (1) to develop regional scale terrain databases in support of watershed science and resource management, (2) to automate numerous kinds of watershed analyses keying on environmental variability for diversifying resource management options, and (3) to improve tools and skills for interpreting watershed-level controls on aquatic systems, including natural disturbance. Hillslope attributes, such as erosion potential, sediment supply, road density, forest age, and fire risk, are aggregated down to the channel habitat scale (20 to 200 m), allowing unique overlap analyses, and they are accumulated downstream in networks revealing patterns across multiple scales. Watershed attributes are aggregated up to subbasin scales (approximately 10,000 ha), allowing comparative analyses across large watersheds and landscapes. Approximately 25 automated tools address erosion risk, habitat indices, channel classification, habitat core areas, habitat diversity, and sediment and wood supply, among others. Search functions target overlaps between specific hillslope and channel conditions and between roads and landslide or debris flow potential. To facilitate its use, NetMap contains hyperlinked users' manuals and reference materials, including a library of 50 watershed parameters. NetMap provides decision support for forestry, restoration, monitoring, conservation, and regulation (Benda and others 2007). NetMap approaches watershed analysis by stream segment reach rather than from DEM grids as in AGNPS and DHSVM or from hillslope polygons as in SWAT and WEPP. Its original intent was to aid in evaluating impacts of land management on aquatic ecosystems, but it also shows considerable potential to aid in cumulative watershed effects for other watershed services. The 2007 version does not predict absolute amounts of hillslope erosion, but rather estimates the erosion risk from hillslopes associated with each channel segment. NetMap is not a public domain package, so to apply it, the managers need to obtain the program and the necessary watershed information from the developers (http://www.earthsystems.net/).

Hydrologic Engineering Centers (HEC) Tools

The HEC tools were developed by the Army Corps of Engineers. HEC-HMS, Hydrologic Modeling System, simulates rainfall-runoff responses and flow accumulation and routing through watersheds. HEC-RAS, River Analysis System, models open channel flow through watershed scale stream and river networks. Future versions of HEC-RAS are expected to model sediment and contaminant transport. HEC-ResSim models flow regimes in regulated river systems. In general, the HEC-series tools are thoroughly documented with well-written user manuals. For completed HEC modules, ArcView 3.x interfaces provide step-by-step assistance with model build, execution, data analysis, and graphic display. Through a partnership with The Nature Conservancy, HEC-EFM, Ecological Functions Model, proposes to model changes in flow characteristics during the year so that users can evaluate the ecological responses to alternative flow regimes. For more information on HEC tools access go to http://www.hec.usace.army.mil.

GRAIP: Geomorphic Road Analysis and Inventory Package

GRAIP is a data collection and analysis process and a set of tools for evaluating the impacts of roads on forested watersheds (RMRS 2007). GRAIP combines a road inventory methodology with a GIS analysis tool set to predict sediment production and delivery, risk of mass wasting from gullies and landslides, and ease of fish passage where roads cross streams. For further information on the status of this tool go to: http://www.fs.fed.us/GRAIP/index.shtml.

ArcHydro: GIS for Water Resources

ArcHydro was designed as an organizing framework with which to generate integrated watershed systems linked to relational databases that then port data to and from watershed modeling tools (ESRI 2002; Maidment and Djokic 2000). It functions as a toolbar plugin in ESRI Arc 8 to 9.x. Required terrain data are extracted from a DEM. Channel segments of a network system are linked to drainage areas through node architecture. Nodes associate all watershed metrics into a personal geodatabase structure. XML programming language then links the geodatabase to external modeling systems. ArcHydro was developed by the Center for Research in Water Resources (CRWR) of the University of Texas at Austin and ESRI. Software and documentation are available at http://www.crwr.utexas.edu/giswr/.

Methods of Slope Stability Analysis

Introduction

Landslides may be considered at a variety of spatial scales during analyses of the watershed effects of fuel management. At one extreme, the general hazard of slope failure is studied over a large, relatively homogeneous portion of a landscape. In this case, the analysis applies to the entire polygon and predicts only general susceptibility to landslides rather than the likelihood of a particular mass failure. At the other extreme, the stability of a particular portion of a single hillslope may be investigated and analyzed intensely. The techniques of stability analysis change considerably over this range of scales. Traditionally, stability analyses concern evaluating the likelihood that a slide or slides will happen. However, regardless of the spatial scale of investigation, there are several other important considerations beyond simple landslide occurrence that can affect the choice of stability analysis method. Is it necessary to also assess the size, specific location, timing, frequency, velocity, or travel distance of slides? Do predictions need to be made in absolute or relative terms?

The following is a brief review of a complex subject, and all those unfamiliar with this subject are strongly encouraged to seek technical assistance from experts within and outside the agency before attempting to analyze the mass stability of hillslopes.

Analysis of Individual Landslides or Single Hillslopes

This subject is a well-established core component of the field of geotechnical engineering. In general terms, an analysis is made by calculating both the available static forces resisting sliding and the static forces causing sliding. The ratio of resisting to sliding forces describes the "Factor of Safety" (FOS) against sliding, and theoretically, a slope will not fail if the FOS is greater than 1, in other words, the shear strength available is greater than the strength required to just barely maintain stability. This method is sometimes called a limit equilibrium slope stability analysis as it concerns conditions in the slope when the balance of forces is just at the limit of equilibrium.

The most simple limit equilibrium formulation, and one that works well on many mountainous hillslopes, is the "infinite slope" analysis (fig. 3). Here, the assumption is made that the failure along which sliding occurs, the failure surface, is planar and generally parallel to the ground surface. This failure surface is often understood to be at the soil/bedrock contact, as rock mechanical shear strength is normally much greater than that of soil. It is further assumed that the slope is infinitely long, hence the "infinite slope" nomenclature, and homogeneous. This means that any single small element of

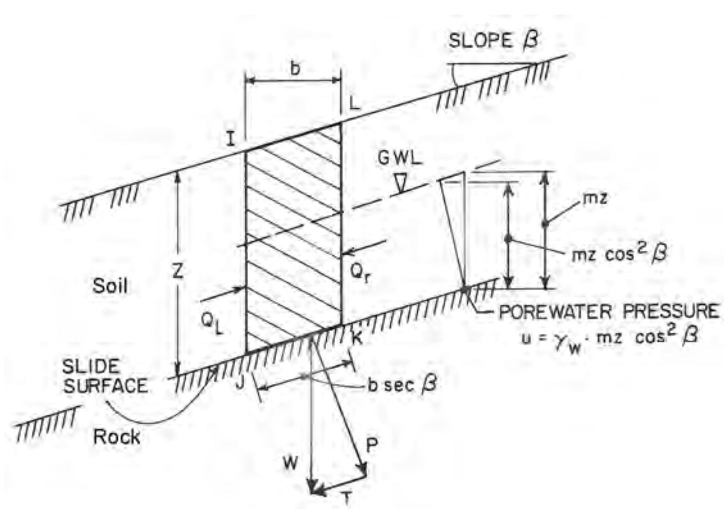

Figure 3. Forces involved in a limit equilibrium infinite slope stability analysis.

the slope can be analyzed and the stability of that element will be the same as that of the whole slope. It also means that the force acting on the upslope face of the element will be exactly balanced by the force acting on the downslope face. A further very important assumption is that forces acting parallel to the topographic contours (cross-slope forces) can be ignored. So, the analysis is purely two-dimensional and involves only the balance of forces along the failure surface on the bottom of the analyzed slope element that is normally oriented directly down the line of steepest descent on the slope (fig. 3). If a further simplifying assumption is made that the groundwater is unconfined and flowing parallel to the slope, the FOS can be calculated as:

$$FOS = \frac{c_s + c_r + (\gamma z \cos^2 \beta - \gamma_w m_z \cos^2 \beta)\tan \phi}{\gamma z \sin \beta \cos \beta} \tag{12}$$

where c_s = soil cohesion
c_r = root cohesion
γ = soil unit weight = ρg

where ρ = soil bulk density and g = gravitational acceleration
γ_w = water unit weight
m_z = vertical distance from the failure surface to the groundwater surface
z = vertical soil depth
β = slope angle
ø = angle of shearing resistance of soil (soil "friction angle")

Of these variables, the FOS is most sensitive to c_s, c_r, m_z, z, and β, and effort should be concentrated on their values while reasonable assumptions can often be made about the others without introducing undue uncertainty. Equation 12 is the basis for almost all mechanistic techniques of slope stability analysis that are currently used at any scale of investigation of mountainous hillslopes.

Fuel management potentially affects two parameters in Equation 12: c_r and m_z. The first is due to changes in the three-dimensional spacing, size, and strength of roots in a hillslope. The second is a result of changes in the groundwater conditions that may arise from variations in the water use by vegetation on the slope. These effects are discussed in Chapter 6. In soils that are high in silt, the "apparent" soil cohesion c_s can be high because of internal water tension at low water content. If fuel management results in reduced evapotranspiration, then it is possible that pore water pressure could increase and the apparent cohesion decrease, leading to a reduced FOS.

Very simple analyses like that above can be quickly computed on a hand-held calculator. However, computerized analyses also have been developed to evaluate many more complicated slope conditions, including those with non-planar failure surfaces, confined groundwater situations with excess pore water pressure, variable soil properties, pseudo-static forces from earthquakes, and surcharge loads from concentrated masses on the ground surface. When the assumptions of an infinite slope are relaxed, automated searches can be made for the minimum FOS for any size of landslide on a slope profile. These techniques are discussed in great detail in many references such as Graham (1984), Nash (1987), and Duncan (1996).

Equation 12 can be used to predict in absolute terms the FOS of an individual slide at a particular location. As we will see later, it can also be implemented in a GIS format to map the spatial variability in FOS, either in an absolute or relative sense. If this analysis predicts a landslide at a particular location, it can also be used to evaluate the timing and frequency of such sliding, although this is not easy. Examination of Equation 12 shows that the only parameter that naturally varies strongly in time is the pore water pressure (through the variable m_z). This parameter normally fluctuates in response to temporal patterns of precipitation, snow-melt, or rain-on-snow. Changes in m_z in response to variations in moisture input to the hillslope can either be established through empirical correlations or by groundwater modeling. If the temporal pattern of other parameters, such as the root cohesion, is known, then again the timing and frequency of sliding can be objectively evaluated by repeating the analysis of Equation 12 for those changed conditions.

In its two-dimensional form, the simple limit equilibrium stability analysis does not predict landslide size: the analyzed element can be of any length along the slope profile. Slide size is quite important as it strongly influences other results, such as the slide travel distance and the amount of debris likely to be carried into a stream. If the infinite slope analysis is extended to three dimensions, then size can be estimated. Currently, the great majority of stability analyses, even computer-based techniques, are two-dimensional, although there is now a moderate amount of research to develop three-dimensional analysis techniques. This is an issue of particular importance to fuel management, as the reinforcement from tree roots is truly three-dimensional (cross slope and up/down slope as well as vertical). A change in a forest canopy that increases or reduces root reinforcement logically should affect not only the location, but the size of landslides.

If the infinite slope equation is cast in terms of a balance of total forces, it is possible to predict the lateral dimensions of a slide according to:

$$\frac{\gamma z \sin\beta \cos\beta - c_b - (\gamma z - \gamma_w m_z)\cos^2\beta \tan\phi}{z((w+q)c_l + 2\overline{\sigma'_{ho}} \tan\phi)} = \frac{1}{w} \qquad (13)$$

where

c_b	=	soil + root cohesion on the base of the slide
c_l	=	soil + root cohesion on the sides and head of the slide
w	=	the width of the slide (cross slope width)
q	=	the ratio of landslide width/downslope length
$\overline{\sigma'_{ho}}$	=	depth-averaged frictional resistance on the sides of the slides
	=	$= \frac{K_o}{2}(\gamma z - \gamma_w m_z)\cos\phi$
K_o	=	at rest earth pressure coefficient = 1-sin ϕ

Equation 13 is derived from Equation 16 in Casadei and Dietrich (2003), which includes the lateral soil friction as well as lateral soil and root cohesion.

While predictions of the FOS of some specific location or the width of a slide at a site are relatively straight forward, estimates of slide velocity and travel distance are difficult. This is an area of very active research, but the prevailing advice at the present time is to assume that shallow slides that mobilize into debris flows will move at rates on the order of m/s to tens of m/s. In the Oregon Coast Ranges, it has been empirically found that shallow land slides (soil thickness 1 m or less) (Dietrich and others 2007) tend to stop when the slope angle in the runout zone declines to less than 3.5^o or when a tributary transporting a debris flow intersects a receiving channel at an angle greater than 70^o (Benda and Cundy 1990).

Thus far, the discussion has focused on planar failures. In some cases, rotational failures can occur. These are more common on uneven terrain or where the surface is underlain by uneven bedrock leading to pockets of elevated soil water. Roads frequently are subject to rotational failures. These types of failures are more difficult to analyze, and generally require iterative solutions. Computer programs such as XSTABL (Sharma 1994) have been developed to assist for these conditions.

Earth flows may be another source of sediment movement, accelerated by increased soil water contents as previously discussed. Their analysis, however, is best carried out by geotech specialist as there are no readily available tools for such soil displacement.

Slide Hazard Assessment Over Broad Areas

Landslide inventories

It is commonly observed that landslides often happen in places where they have occurred in the past. Thus, a relatively simple inventory of past landslides may have some predictive power about future events. Slide inventories over broad areas are normally conducted by stereoscopic examination of air photos with some limited ground verification of results. These inventories often simply identify slide locations, but occasionally they are expanded to also map landslide types, sizes, and runout distances. If air photos of multiple ages are available, the time of occurrence and recent state of activity can sometimes be estimated. Inventories can be quantified by calculating the spatial variation in landslide density to produce what is called a landslide isopleth map (Wright and others 1974). The advantage of photo-based inventories is that they are relatively inexpensive and can survey large areas rather quickly. It is very difficult, however, to determine the size of landslides from air photos and often little is learned of the site conditions that caused sliding. Furthermore, if current or future slope conditions are outside those represented by the photographic record of landslides, erroneous predictions of existing or future hazard may be produced. Still, this is an extremely useful technique and normally a slide inventory map is developed, even if other analysis techniques are employed. In many cases, the inventory data are used to calibrate, and validate, the other predictive methods (McClelland and others 1999).

GIS multi-factor overlay approaches

An extension of the simple inventory is to map the variables that could reasonably affect slope stability, in other words, the parameters in Equation 12. In a GIS environment, these parameter maps are overlaid and examined for combinations of parameters that exist at mapped landslide sites. By identifying the same combinations of parameters at other locations, a map of <u>future</u> landslide susceptibility or hazard is produced that is based on correlations of slope, aspect, vegetation, geologic materials, and geologic structure with <u>past</u> landslides. In more sophisticated models, the correlations are evaluated statistically using discriminate analyses, logistic regression, or Bayesian belief models (Carrara and others 1991; Dai and Lee 2001; Gorsevski and others 2006a; Gritzner and others 2001)

These GIS-based analyses are relatively simple to conduct if suitable factor maps are available for the pertinent parameters. Again, the assumption is that future conditions will be within the range represented by the existing record of landslides. This supposition may be in error if the regional climate changes, or if management, fires, or other changes in land use occur that change the vegetation community and thus the root cohesion. GIS analyses do not generally predict landslide size, timing, frequency, velocity or runout distance.

Deterministic engineering style analyses

In this approach, mechanistic analyses of the type shown in Equation 12 are performed over large areas represented in a GIS. The result is a distributed, physically based model that can predict local slope stability in absolute or relative terms. The great advantages are that the technique is objective and can predict location, timing, and frequency of slides. It is also possible to conduct sensitivity analyses and predict landscape response to changes in environmental conditions from natural and human causes. These conditions may be completely outside the range of those represented in the history of observable landslides in an area. For example, it is possible to investigate the effects of forest thinning or complete removal on slope stability. At present, almost all models of this type employ the two-dimensional infinite slope analysis (Equation 12) and cannot evaluate landslide size (Gorsevski and others 2006b; Montgomery and Dietrich's SHALSTAB model (1994); Ward and others 1982).

A distinct disadvantage of this approach is that the full list of parameters in Equation 12 must be characterized over large areas. Some, such as soil depth and pore water pressure, are very difficult to predict. Pore water pressure, in particular, is a problem because it varies in four dimensions—three spatial and a temporal dimension. Pack and others (1998) attempted to resolve the problems of parameter uncertainty by describing the probability distribution function (assuming a uniform probability density function) of some parameters, rather than using single valued parameters. In their model Stability Mapping Index (SINMAP), they then calculate a probability of failure rather than an FOS. Gorsevski and others (2006b) used a stochastic weather generator and a soil water balance model in a GIS approach to predict areas of instability. Wu and Sidle (1995) noted that rainfall is a stochastic parameter in their Distributed Shallow Landslide Analysis Model (dSLAM). This model uses either an event-based rainfall record or theoretical distributions from Monte Carlo simulations to predict the pore water pressure, and thus computes either FOS or probability of failure. dSLAM also incorporates the time rate of decay of root strength after tree death and the rate of site vegetation regrowth, although still only in a two-dimensional model. In their Level one Stability Analysis model (LISA), Hammond and others (1992) also employed a Monte Carlo analysis with probability density functions for all parameters in Equation 12. Rather than calculating the probability of failure directly on the pixels in a DEM, LISA first stratifies the landscape into homogeneous units and then evaluates the probability of failure for each stratum using a Monte Carlo scheme. This is a subtle but important difference, and in the LISA model, a high probability of failure for a stratum gives no information about where in a polygon a particular combination of conditions might exist that would lead to a higher probability of failure. The assumption is that the polygon is homogeneous and the probability of failure is equal throughout.

As discussed before, all distributed mechanistic models require calibration and validation against local landslide information in a slide inventory map.

Analysis of Slope Stability Along Roads

Depending on the terrain that is traversed and the style of construction, roads can have a variety of effects on slope stability. Cut and fill slopes are normally steeper than local undisturbed terrain and are inherently less stable. Cut slopes also frequently intercept shallow groundwater and can concentrate this water in places that will cause landslides. Normally, groundwater concentrations will occur where road fill is thickest—across the corridors of small unchanneled valleys and hollows. Disruption of groundwater flow by the overlying fill can cause elevated pore water pressures and fillslope failure. Mountain roads are sometimes damaged when they cross landslides that are so large they are relatively unaffected by the road, yet the road is a victim when the slides move.

The stability of a road corridor can be considered at a variety of spatial scales, ranging again from individual slides to the general mass stability of an entire road. Individual slides are analyzed using methods such as those introduced previously. At broader scales of investigation, mapping techniques have been developed that can be described as qualitative engineering geomorphology analyses. This method of "reading the landscape" from a geomorphic perspective over time scales well beyond human experience is described in a series of papers published mostly in the Quarterly Journal of Engineering Geology. Perhaps the first of these is the pioneering work by Brunsden and others (1975). In this approach, landforms are mapped in detail using a combination of air photo interpretation and field work, and then process domains, including those of landslides, are interpreted from the morphologic maps. By doing this, a qualitative assessment of local or regional landslide hazard can be produced. Future remote sensing will likely incorporate LiDAR capabilities into evaluating road stability (Kwak and others 2005).

Quantitative deterministic analyses, most often based on the infinite slope equation, can also be done over entire road corridors. A recent example is that of Borga and others (2004) who used the SHALSTAB model with an adaptation of the groundwater component to accommodate the interception and rerouting of groundwater by a road network. A similar approach is used by Prasad and others (2005) who employ a modified version of the SINMAP probabilistic stability analysis as part of the Road Sediment Analysis Model (RSAM).

Model Calibration and Validation

Calibration is the process of determining input variables so that the model generates satisfactory predictions. Validation is using a calibrated model on a different site or data set to see whether reasonable values are still predicted (Conroy and others 2006; Elliot and Foltz 2001; Elliot and others 1991). Many models require some form of calibration as part of the application. For example, many of the WEPP vegetation files require calibration for local weather conditions to ensure that predicted amounts of canopy and ground cover are correct. The SWAT model is generally calibrated for runoff for current conditions before evaluating alternative management activities. Many research papers have been published on model validation, but it is always a good practice to compare predicted runoff and erosion rates and amounts with values observed in the area. Are the predicted values reasonable compared to monitoring or research studies that have been done in the past for similar conditions? If not, the user may need to consider some additional calibration or add some qualifying comments in the report associated with the modeling activity. If a given tool does not appear to be performing in a satisfactory manner, it may be useful to contact the group supporting that model to ensure the model is being used correctly and to determine if the model is appropriate for the given conditions.

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Cumulative Watershed Effects of Fuel Management in the Western United States

Chapter 14.

Understanding and Evaluating Cumulative Watershed Impacts

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Introduction

Considerable effort is devoted to evaluating cumulative watershed impacts during planning for forestry activities on federally managed lands. With the recently increased emphasis on reducing forest fuels, cumulative impact analyses must now evaluate a new suite of activities over a broader scale than had generally been considered in the past. It is useful to review the concept of cumulative impacts and the variety of methods used for impact analysis to identify approaches capable of addressing the new challenges.

This paper first outlines the history and definition of cumulative watershed impacts and discusses the types of interactions that complicate their evaluation. Methods used for cumulative impact evaluation are then described. Finally, the types of errors found in evaluations are identified, and the relevance of such problems is assessed by identifying flaws found to be significant by federal courts. A variety of earlier publications can provide additional insight and detail concerning cumulative impact analysis (CEQ 1997; MacDonald 2000; Reid 1993, 1998).

History and Definition

Cumulative impacts are nothing new. The importance of cumulative watershed impacts was widely recognized by the mid-1800s when a French economist described the increasing incidence of debris flows in the Alps:

> "The elements of destruction are increasing in violence...The devastation advances in geometrical progression as the higher slopes are bared of their wood, and 'the ruin from above,' to use the words of a peasant, 'helps to hasten the desolation below'" (Blanqui 1843, Address to the French Academy of Moral and Political Science, quoted by Marsh 1864).

George Marsh observed the problems caused by altered runoff and erosion in Europe and the Middle East and noted that analogous conditions were developing in America:

> "The rivers which rise in [deforested uplands of the Adirondacks], flow with diminished currents in dry seasons, and with augmented volumes of water after heavy rains. They bring down much larger quantities of sediment,

> and the increasing obstructions to the navigation of the Hudson, which are extending themselves down the channel in proportion as the fields are encroaching upon the forest, give good grounds for the fear of serious injury to the commerce of the important towns on the upper waters of that river, unless measures are taken to prevent the expansion of 'improvements' which have already been carried beyond the demands of a wise economy" (Marsh 1864).

The U.S. Congress passed the Forest Reserve Act in 1891 largely in response to the increasing concern over downstream impacts arising from rapid upland deforestation. This Act authorized the President to set aside selected forestlands, and these reserves eventually became the first national forests. The focus on watershed concerns was evident in the 1911 Weeks Law, which authorized purchase of lands for national forests if they were in the watersheds of navigable streams. In a sense, the Forest Service exists because of early recognition of cumulative watershed impacts.

The term "cumulative impacts" did not become widely used until the Council on Environmental Quality (CEQ) produced guidelines for implementing the National Environmental Policy Act (NEPA) of 1969. This particular issue arose because NEPA planning focused on specific projects or programs. The CEQ realized that a broader perspective on problems could be overlooked under these conditions—examination of a project in isolation would not reveal the impact levels that would actually be experienced. The CEQ thus specified that the overall, or "cumulative," impact level also must be evaluated, and defined the cumulative impact as:

> "...the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time" (40 CFR § 1508.7).

The term "cumulative effect" is widely used as a synonym for "cumulative impact," and the CEQ defines "effect" to be the same as "impact" in the context of NEPA (40 C.F.R. 1508.8). The CEQ did not specifically define "watershed impact," but common usage indicates that this refers to any impact that involves water flowing through a landscape, either because water-related resources are impacted or because a change in watershed processes generates the impact. Cumulative watershed impacts are just one variety of cumulative impact, but watershed impacts often are important influences on other types of cumulative impacts. Analyses of cumulative impacts on terrestrial wildlife, for example, often need to consider the cumulative influences of watershed impacts on riparian habitat.

The notable point about the definition is that there is nothing sophisticated or mysterious about it. "Cumulative" is being used simply as an adjective with thesaurus synonyms of "aggregate," "collective," "accruing," and "combined," rather than as part of a name for a new type of impact. The net impact that a resource experiences usually represents the combined effects of multiple influences, so evaluation of the cause and severity of any impact ordinarily involves assessment of multiple influences and so is, in effect, an analysis of the cumulative impact.

Interactions Between Impact-Generating Influences

The overall impact on a resource—the "cumulative" impact—differs from an "individual impact" if the total impact is affected by more than one anthropogenic influence. Influences can interact in several ways, contributing to impacts that can accumulate through time, space, or both. Impacts can accumulate as a result of repetition of an activity at a site (example: multi-cycle logging in the transitional snow zone periodically increases rain-on-snow peakflows, increasing the cumulative cost of excess downstream flood damage) or from progression of the activity to new sites (example: additional logging increases the spatial scale of the increased rain-on-snow flooding, also increasing the cumulative cost of flood damage), and they can accumulate from different coexisting activities or from different activities occurring sequentially (example: logging, housing developments, and fire all increase rain-on-snow flood peaks). Impact severity can increase either because one type of change accumulates (example: the cumulative cost of increased flood damage accrues through time) or because different influences affect the same entity (example: flood damage increases because of both increased flood peaks and increased channel aggradation). If influences from a single activity impact a resource progressively through time (CEQ 1997) or by way of multiple mechanisms (example: logging can contribute to both increased peakflows and increased aggradation, both leading to increased flood damage), the overall impact can also be evaluated as a cumulative impact.

The most widely recognized subdivision of cumulative impacts is into “additive” and “synergistic” impacts. Impact severity for additive impacts can generally be inferred by summing the effects of the component impacts. For example, reductions in municipal water supply from reservoir aggradation can be estimated by summing sediment input rates from the variety of sediment sources in a watershed. In contrast, synergistic impacts require additional knowledge if outcomes are to be predicted. For example, increased stream temperature reduces salmonid survival, as does decreased pool habitat. However, if salmonids cope with increasing temperatures by taking refuge in deep, cool pools (Nielsen and others 1994), the combined effect of increased temperatures and infilled pools would have a much greater impact than could be predicted by considering each impact separately.

Cumulative increases in impact severity can occur if an existing adverse condition increases in magnitude, duration, or frequency. Although these kinds of changes usually are associated with one another, it may be useful to consider them separately because their influences on an impacted resource may differ. Increased turbidity magnitude, for example, usually accompanies an increase in duration of high turbidity periods and an increased frequency of high turbidity spikes. If magnitudes are high enough, turbidity itself can become lethal to salmonids. At lower turbidity levels, protracted duration can result in lengthy periods during which salmonids cannot see and capture prey, while an increased frequency of moderate turbidity peaks may lead to cumulative gill damage (Newcombe and Jensen 1996). Each type of impact holds different implications for the impacted resource and requires a different approach to analysis.

Impacts can also occur because a new condition is created by an activity, as when a new chemical or species is introduced. In other cases, impact severity may increase because the sensitivity of the resource is altered. Warmer stream temperatures, for example, might increase salmonid metabolic rates, thereby increasing the salmonids’ susceptibility to impacts from heightened dry-season turbidity caused by recreational off-highway vehicle use, mining, and riparian grazing.

Impacts also occur if the mechanisms that moderated impact severity during natural shifts in condition are no longer operative. In the past, for example, coho salmon may have been able to escape high turbidity events in the mainstem by taking refuge in intermittent clear-water tributaries and seasonal wetlands on floodplains (Peterson 1982). That coping strategy is defeated if road culverts block access to the tributaries or development fills the wetlands. The fish then become susceptible to impacts from conditions that once could have been tolerated or avoided. At a larger scale, regional depletion of populations can reduce a species’ resilience and resistance both to anthropogenic changes and to natural disturbances such as major storms and fires.

Impacts like those affected by changes in turbidity represent the combination of multiple influences—that is, are “cumulative”—in several senses. First, an increase in turbidity is likely to be caused by multiple activities that occurred at different times in the watershed, and it is the cumulative influence of these activities that leads to the turbidity levels now seen. Second, the impact to the resource accumulates through time as altered conditions persist. Third, the impacted resource often exhibits the effects of multiple impact mechanisms associated with the changed condition. As a hypothetical example, chronically high turbidity levels might leave fish weakened to the point that they succumb to infection induced by gill abrasion during a high turbidity spike. And fourth, altered turbidity is usually just one of many impact mechanisms that combine to produce the overall impact on the resource of interest.

Interaction between responses can also lead to moderation of impact levels. For example, water releases from the base of a reservoir might reduce downstream temperatures that had been artificially raised by loss of riparian shade. Mitigation strategies are based in part on inducing responses that can compensate for, or reduce. undesired responses from other management activities.

Linkages Between Land Use Activities and Impacts

The impact mechanisms discussed above are “proximal” mechanisms—the changes experienced by the resource of concern that result in impact. In the case of watershed impacts, these proximal mechanisms often take effect at some distance downstream from the activities that ultimately caused them, and they are often delayed in time. Developing an understanding of the ultimate causes for a particular impact can be challenging both because of the inherent complexity of process interactions and because of the spatial and temporal lags those interactions create.

Off-site impacts can occur only if there is a physical transfer of material or energy from the activity site to the impact site. The flow of water provides the transfer medium for watershed impacts. Most land use activities occur on hillsides (figure 1), and most activities modify characteristics such as vegetation, topography, and soil conditions. Changes to these characteristics, in turn, influence the magnitude and timing of inputs of water, sediment, heat, chemicals, nutrients, and wood to streams. If inputs to the stream system are modified, the downstream transport of these watershed products is also modified, and downstream conditions change in response to the shift in transport rate and capacity. Furthermore, changes in the input and transport of one watershed product usually influence the input and transport of others. Decreased wood inputs, for example, may result in decreased channel roughness, causing increased flow velocities, which can increase bed and bank erosion and thereby increase sediment loads. Or, an increase in

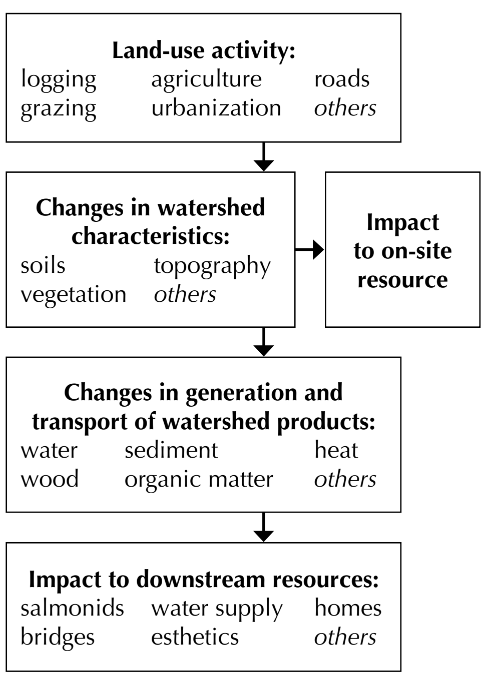

Figure 1. Relation between hillside land use activities, on-site, and off-site environmental impacts.

sediment input can aggrade a channel, causing it to shift course and undermine banks, thereby increasing wood inputs. The cascade of influences that takes effect after an upstream change depends partly on the location and nature of the change and partly on its timing relative both to other changes and to large events such as storms and droughts.

The spatial decoupling of downstream impacts from the upslope activities that triggered them obscures the circumstantial linkages between cause and effect. The locations at which an impact becomes evident depend on the distributions of

1. activities that contribute to the impact,
2. locales likely to respond to the changes, and
3. resources that might be impacted by the changes.

Different sites often respond to the same kind of change in different ways. An increased input of fine sediment, for example, may not affect channel form along steep boulder reaches but may contribute to aggradation downstream in a low-gradient reach or lake.

Temporal decoupling is also common and obscures evidence of causal linkages. Time lags between cause and response can arise for several reasons.

1. A long period of accumulation may be required before impacts become significant enough to arouse concern (example: gradual declines in a species' abundance may not be evident for a long time).
2. Different steps in the cascade of influences require different lengths of time to recover, so impacts may continue to accumulate downstream long after on-site impacts have recovered (example: an initial hydrologic change may destabilize channel banks, which continue to produce sediment even after the hydrologic regime is restored).
3. An impact may not become evident until the occurrence of an environmental trigger, such as a major drought or storm (example: slopes may appear to have remained stable until a landslide-generating storm occurs).
4. The change may not occur until the accumulated effect is great enough to trigger another suite of processes (example: accelerated sedimentation in a wetland may not be of concern until it has progressed to the point that a particular plant species begins to die out).

Interpretation of linkages may also be challenging because different kinds of changes can lead to the same response, so the cause for an impact often cannot be inferred from observation of the impact alone. The distribution and manifestation of future watershed impacts thus depends on the geography, history, and ecology of an area, as well as on the nature of contemplated land use activities. For these reasons, and because of the potential importance of synergistic interactions, causes and effects of changes must be evaluated in the context of specific locales and resources of concern.

Assessing and Managing Cumulative Watershed Impacts

Cumulative impact assessment became a challenge for federal land management agencies with the adoption of the CEQ's regulations for implementing NEPA in 1978 (40 C.F.R. 1500). The importance of environmental impacts was widely appreciated before then, but there was no mandate to consider a project as complicit in causing an impact if that project did not by itself create a significant impact. The new regulations indicated that if an impact is already significant and a new project adds to the impact, that project contributes to a significant impact. Such influences would now need to be disclosed and evaluated for projects subject to regulation by NEPA. Subsequently, various states passed legislation that requires similar evaluations for state and private projects not subject to NEPA oversight.

Before the need for cumulative impact analysis, forest management had tended to focus on the forest stand level and was based on identifying an appropriate action for the given stand condition. Hydrologists and geomorphologists were necessary for meeting forestry related goals only insofar as those goals concerned infrastructure. After implementation of NEPA, however, analysis of off-site cumulative impacts required hydrological and geomorphological expertise, but foresters, hydrologists, and geomorphologists within forest management agencies did not have a history of working together. Methods for cumulative impact analysis had to be developed that would enable the interdisciplinary work now needed and could be carried out routinely by existing personnel.

First Generation: Tactical Manuals

The Forest Service provided a decentralized response to the need for cumulative impact analysis, with approaches being designed as needed in different Regions. Most prominent among the early approaches were the Equivalent Clearcut Area method developed for Idaho and Montana, the R1/R4 Sediment-Fish Model used in the Idaho Batholith, and the Equivalent Roaded Acres method from California. This first generation of analysis methods consisted of detailed procedures that lead the user through specific calculations. Each approach effectively provides an accounting system for activity level, making it possible to manage cumulative impacts by holding a watershed's activity level below that associated with impact generation. The methods differ in assumptions about which impact is most important and in the strengths of their technical foundations.

Equivalent Clearcut Area

The Equivalent Clearcut Area approach to cumulative impact analysis was developed in the 1970s for application in northern Idaho and Montana (Galbraith 1975; USFS 1974). In this area, the impact of most concern was identified to be channel destabilization due to increased peakflows resulting from decreased evapotranspiration after logging. Altered evapotranspiration is associated directly with changes in canopy cover, so the net influence of an activity—including logging, road construction, burning, and other forest management activities—can be assessed by determining its effect on canopy cover by calculating the "equivalent clearcut area" (ECA) the activity represents. The ECA for the proposed project is then added to that already present in the watershed. If the resulting total is above the threshold considered acceptable for the area by experts, activities are deferred until the ECA recovers through time or activities are modified to produce a smaller increase in ECA.

A major advantage to the ECA approach is that it is relatively simple to implement. The analysis requires no specialized expertise and can be carried out from the office. Furthermore, anyone carrying out the analysis with the same input data will get the same answer.

Disadvantages, however, are significant. First, the method is specific to a particular type of impact. If other impacts are of concern in an area, other analyses must be done. It might be argued that if the most sensitive impact is screened for, others will be handled implicitly. However, if other impacts are caused by unrelated mechanisms, there is no reason that screening on the basis of one mechanism would produce results relevant to problems caused by another mechanism. For example, screening on the basis of increased peakflows would not identify areas sustaining impacts from acid mine drainage. Second, recovery is calculated according to that estimated for hydrologic response and does not account for the recovery rate of impacted channels. If channel morphology or fish populations require years to recover after hydrologic changes are reversed, impacts can continue to accrue from new activities taking place after hydrologic recovery. Testing of the method suggests that it may underestimate the magnitude of peakflow changes following logging (Belt 1980; King 1989).

R1/R4 Sediment-Fish Model

In the Idaho Batholith area, declining salmonid populations were a major concern, and research suggested that deposition of logging-related fine sediment in streams was contributing to salmonid decline. Considerable research was devoted to evaluating the link between specific forest management activities and erosion rates, between sediment inputs and sedimentation (Cline and others 1981), and between sedimentation and salmonid survival (Stowell and others 1983). The result is a series of models that can be used to estimate impacts on salmonid survival from the distribution of forestry related activities through time in a watershed. Recovery rates for surface erosion are incorporated, so inputs from old activities are lower than for recent ones.

This approach also is easily implemented, once the initial work is done to quantify the necessary relationships. However, the model depends strongly on the considerable research done to evaluate linkages between forestry and salmonid survival in the Idaho Batholith, so the model cannot be transported to a new area without substantial research. As was also the case with the ECA method, the approach is targeted for a particular impact mechanism and type of impact—analysis of other impacts would require different approaches.

The R1/R4 model has more recently been combined with a water balance model to construct the WATSED model (USDA 1992), now used widely in the area to estimate the combined effects of changes in runoff and sediment associated with logging.

Equivalent roaded acres

The approach for evaluating cumulative impacts developed by USFS Region 5 in California is similar to the ECA method. The focal concern in California had been identified as an increase in peakflows, and research in Oregon suggested that peakflows increase if more than 12 percent of a watershed surface is compacted. On this basis, the area compacted by each activity in a watershed is summed to determine whether the total "equivalent roaded acres" (ERA) is greater than the threshold considered appropriate for an area (for example, Haskins 1987). Here, too, recovery is assessed on the basis of recovery from compaction rather than recovery of the impacted resources. It was quickly recognized that impacts other than those from increased peakflow had to be considered. Descriptions of the method were soon revised to indicate that the calculation resulted in an index of the activity level in a watershed (USFS 1988). Thresholds of concern and weighting factors for various activities are intended to be modified to reflect conditions in each area, such as rock type and topography, and can be calibrated by measuring the activity levels associated with observed impacts. More detailed analysis is triggered if the calculated ERA approaches a threshold of concern for the watershed.

Advantages of this approach are its reproducibility and relative simplicity in calculation. The method can also be calibrated to apply to specific areas and impact types, as long as the thresholds for impact can be associated with particular activity levels.

The ERA approach shares most of the disadvantages of the ECA approach, and it also carries the implied responsibility for the user to ensure that it is appropriately calibrated for each application. As was also the case for the ECA and R1-R4 methods, there is no assurance that a single screening tool can adequately address multiple, unrelated impact mechanisms—an initial impact analysis would be needed to identify the impacts of concern and assess their associations with ERAs. The ERA approach has also not been independently tested. One study suggests that increased ERAs within 200 m of a stream are associated with impaired aquatic invertebrate communities (McGurk and Fong 1995), but it is not clear whether ERAs are a more efficient index of buffer-strip effectiveness than other metrics.

Second Generation: Strategy Guides

Use of standardized procedures such as the ERA method is very different from the approach that had been adopted by most other spheres of activity. For non-forestry applications, there did not seem to be a perceived need for a single "cumulative impact analysis tool" to address all cumulative impacts. Instead, it was recognized that any activity can influence many types of impacts at a variety of spatial and temporal scales. Each impact needs to be evaluated by analytical methods pertinent to that specific impact. A highway construction project, for example, might require analysis of cumulative changes in long-term traffic patterns at a regional scale, of cumulative peakflow changes at a small-watershed scale, and of the short-term effects of construction on cumulative siltation in a reservoir far downstream. This problem-based strategy is simply an extension of approaches to impact evaluation that have been used in the past when particular impacts were of concern. Several forestry applications have adopted the problem-based strategy, and the CEQ has provided further guidance on appropriate analysis strategies.

CDF checklist

By the early 1990s, forestry related applications began to incorporate a broader view of cumulative impacts, as exemplified by the approach eventually adopted by the California Department of Forestry and Fire Protection (CDF). Because NEPA does not apply to non-Federal jurisdictional issues, the California legislature in 1970 passed the California Environmental Quality Act (CEQA), which applies requirements similar to those of NEPA to activities under state jurisdiction. At first, CDF responded to the need to address cumulative impacts for state and private forestry by applying "best management practices" (BMPs), asserting that if direct impacts are held to a minimum, the cumulative impact will be insignificant. This approach was found by the courts to be invalid as it did not reflect the legal definition of cumulative impact (Environmental Protection Information Center v. Johnson, 170 Cal. App. 3d 604, 1st Dist., 1985). (Note: published legal opinions are here cited by volume number [170 for this case], the abbreviated name of the source [California Appellate Reporter], the series number [Third], the page number on which the opinion begins [604], the court providing the opinion, and the year of the decision; unpublished district-level opinions are cited by case number, court, and year).

In the early 1990s, CDF instituted a checklist (CDF 2009 revised) to guide Registered Professional Foresters (RPFs) through the logic trail needed to support a professional opinion of whether or not cumulative impacts were likely from a Timber Harvest Plan (THP). This kind of open-ended approach has the advantage of flexibility, potentially allowing use of appropriate analytical methods for the specific types of impacts relevant for an area.

In practice, however, the documentation required is minimal, consisting primarily of a checklist and a brief explanation of the conclusion. Although the procedure has withstood legal challenges, various commentators have questioned its credibility and effectiveness. Dunne and others (2001), for example, noted that members of their review committee had "...been told explicitly by some RPFs that, in preparing a THP, they would never conclude that a CWE [cumulative watershed effect] is likely because of the unnecessary regulatory burden that such an admission would bring."

CEQ guidance for cumulative impact evaluation

By the late 1990s it had become clear that an appreciable proportion of the cumulative impact assessments prepared for NEPA documents were inadequate (Burris and Canter 1997). In 1997, the CEQ published a manual (CEQ 1997) that outlines an appropriate approach to cumulative impact evaluation under NEPA. The recommended steps are similar to those used for more general environmental impact assessments (table 1), encompassing identification of the impacts of concern and their spatial and temporal scales of expression; description of the baseline conditions, proximal impact mechanisms and tolerance to change; determination of the influence of the project on the impact mechanisms; and description of the impact level expected after completion of the project.

The same document summarizes principles of cumulative impact analysis (table 2). Several of these points are particularly important. First, analysis is carried out from the perspective of the impacted resource. Each significant impact must be evaluated in its own right since each will be influenced by different mechanisms, appear in different places, and have different recovery rates. Second, the boundaries of the analysis area are determined by the locations of the entities that might be influenced by the activity

Table 1. Steps in cumulative effects analysis to be addressed during the components of environmental impact assessment (after CEQ 1997).

Assessment component: Scoping

1. Identify the significant cumulative effects issues associated with the proposed action and define the assessment goals.
2. Establish the geographic scope for the analysis.
3. Establish the time frame for the analysis.
4. Identify other actions affecting the resources, ecosystems, and human communities of concern.

Assessment component: Describe affected environment

5. Characterize the entities of concern in terms of their response to change and capacity to withstand stresses.
6. Characterize stresses affecting the entities of concern and their relation to regulatory thresholds
7. Define a baseline condition for the entities of concern.

Assessment component: Determine environmental consequences

8. Identify important cause-and-effect relations between human activities and issues of concern.
9. Determine the magnitude and significance of cumulative effects.
10. Modify or add alternatives to avoid, minimize, or mitigate significant cumulative effects.

Assessment component: Implementation

11. Monitor the cumulative effects of the selected alternative and adapt management.

Table 2. Principles of cumulative effects analysis (after CEQ 1997).

1. Cumulative effects are caused by the aggregate of past, present, and reasonably foreseeable future actions.
2. Cumulative effects are the total effect, including both direct and indirect effects, on a given resource, ecosystem, and human community of all actions taken, no matter who (federal, nonfederal, or private) has taken the actions.
3. Cumulative effects need to be analyzed in terms of the specific resource, ecosystem, and human community being affected.
4. It is not practical to analyze the cumulative effects of an action on the universe; the list of environmental effects must focus on those that are truly meaningful. (*The boundaries for evaluating cumulative effects should be expanded to the point at which the resource is no longer affected significantly or the effects are no longer of interest to affected parties*)
5. Cumulative effects on a given resource, ecosystem, and human community are rarely aligned with political or administrative boundaries. (*Cumulative effects analysis on natural systems must use natural ecological boundaries and analysis of human communities must use actual sociocultural boundaries to ensure including all effects*)
6. Cumulative effects may result from the accumulation of similar effects or the synergistic interaction of different effects.
7. Cumulative effects may last for many years beyond the life of the action that caused them.
8. Each affected resource, ecosystem, and human community must be analyzed in terms of the capacity to accommodate additional effects, based on its own time and space parameters.

in question, so each issue is likely to require analysis over a different area. This need to define the analysis scope from the point of view of the impact also holds for temporal boundaries, because rates of impact expression and recovery will differ for each impact considered.

Clearly, these analytical issues are not compatible with strict adherence to standardized impact analysis methods such as the ECA, ERA, and R1/R4 approaches, and those approaches are now often supplemented by additional information concerning specific issues of concern.

The Eastwide strategy

Forest Service Regions in the eastern United States developed an analysis approach that incorporates much of the CEQ guidance (Tetra Tech 2002). The Eastwide approach follows the structure suggested by MacDonald (2000), which is based largely on the CEQ (1997) approach but differs from it in sequencing of steps, reliance on "natural range of variability," and omission of an evaluation of impact significance. Appropriate levels of analysis effort are described for different analytical contexts (a highly controversial issue, for example, would require a higher level of effort), and options for content and approach are described for each level of effort. The Eastwide Technical Guide notes that a variety of landscape- and watershed-scale analyses are available for many areas, and that these provide much of the information needed for cumulative impact analyses.

The approach outlined by the Guide differs from the other methods designed for Forest Service use because it consists of a strategy for problem solving that ensures that the appropriate topics will be considered during analysis while allowing the user to choose the analysis scales and tools appropriate for the specific issues and conditions present. Some of the issues important for cumulative impact evaluations in these Regions—and in all other Forest Service Regions—include impacts of forestry, fire, recreation, minerals exploration and mining, oil and gas exploration and mining, and rural development; relevant waterbodies include streams, wetlands, lakes, reservoirs, and groundwater; and potentially impacted resources include flora, fauna, recreational values, domestic and municipal water supplies, recreational and commercial fisheries, and many others.

An important advantage to the Eastwide Strategy is that it incorporates most of the suggestions provided by CEQ (1997). It allows any impact to be evaluated using the specific analytical techniques most appropriate for the issue and for the local conditions; provides for selection of temporal and spatial analysis scales appropriate for the problems present; and makes good use of the technical expertise present within the agency. A potential weakness, however, is the omission of guidance concerning evaluation of impact significance.

Identifying Shortcomings in Cumulative Impact Assessments

Because the need to modify standardized cumulative impact analysis procedures seems likely in view of the suggestions provided by CEQ (1997), it is useful to identify the particular aspects of non-standardized cumulative impact analyses that are likely to present analytical problems. A series of cumulative impact analyses that did not incorporate standardized methods were thus evaluated for technical content. Whether the types of problems found would be considered significant to the implementation of NEPA depends on how those technical flaws are treated by the courts, so recent legal decisions involving cumulative impact analyses were then reviewed to determine how federal courts deal with the common problems.

Problems in Cumulative Impact Analysis for Private Lands

Analytical problems were identified in 14 non-standardized cumulative watershed impact analyses and supporting documents that had been prepared for logging-related activities on private forestlands in northwest California (Reid 2004). Most of these documents were intended at least in part to meet requirements of the California Environmental Quality Act.

Each document incorporated analytical flaws that generally fell into four categories. First, the legal definition of cumulative impacts was often disregarded. Several analyses progressed without an evaluation of existing impact levels or of the contribution from "legacy" impacts, even though the definition of "cumulative impact" focuses on the combined influence of exactly these types of impacts. Another analysis argued that the cumulative impact of the plan would be beneficial because the preferred alternative would have a lower impact than one of the other alternatives. The cumulative impact, however, is the impact "on the environment"—if the overall impact is detrimental, it cannot be presented as "beneficial" simply by comparing the project's incremental addition to that of an even more damaging alternative.

The second set of problems involves evaluation of impact significance. Several analyses argued that an activity would simply perpetuate the existing level of impact instead of increasing its severity, thereby implying that the cumulative impact of the new activity is insignificant. However, the severity of an impact is necessarily defined by the level of damage to the impacted resource, and the level of damage usually increases as an impact persists. Consequently, prolonging an impact generally increases its severity.

In most of the documents examined, impact significance was determined simply through professional opinion, without any documentation of how the significance thresholds were established. One analysis evaluated the sediment input expected from a proposed project and then asserted that the added sediment would result in a negligible risk of significant impact. The analysis neglected to note that the existing impact level for sediment in the watershed had already resulted in the stream's addition to the 303d list under the Clean Water Act, so a regulatory threshold of significance had already been surpassed.

A third set of problems reflected the projects' reliance on mitigation plans to reduce expected impact levels to non-significance. Either BMPs or offsetting mitigation usually were part of the management strategy, but rarely was evidence provided that mitigation would be effective, and the overall effects of the mitigated activities on the impacted resources were not evaluated. Instead, analyses simply noted that the impact levels would be "reduced" by mitigation. However, few of the planned mitigation measures would be capable of preventing the expected impacts because mitigation generally was planned for other sites, was designed to offset different kinds of impacts, or would become effective only after the planned impacts had occurred.

The final set of problems involved technical errors in the analyses, and most of these could have been corrected by technical review. In this case, too, unsupported assertions were common. For example, one document arbitrarily considered bank erosion to be half natural and half anthropogenic. No evidence was provided to support this assumption, yet the analysis conclusions depended on it.

Problems in USFS and BLM Cumulative Impact Analysis for NEPA Documents

Examination of the private-sector analyses disclosed the general types of analytical errors that might be expected in non-standardized cumulative impact analyses, but it did not indicate whether such problems are operationally important. The operational test for the effectiveness of an impact analysis procedure is whether the method produces results that withstand legal challenges. Operational success does not imply technical success. An analysis would be considered technically successful only if its predictions are found to be accurate. Little information is available concerning the accuracy of predictions from analyses prepared under NEPA, and technical success is not considered in this paper.

Whether an analysis is litigated involves a variety of considerations, including how controversial the activity is, the nature of the population likely to be affected by the activity, and the level of legal expertise and funding within that population. An inadequate analysis for a non-controversial plan is thus likely to receive little attention, while a carefully prepared analysis for a controversial project appears to be more subject to challenge.

Federal agencies responsible for managing forestlands have received adverse decisions in a variety of recent cases involving cumulative impact analyses under NEPA. Earlier studies have examined the Forest Service record in NEPA litigation (for example, Jones and Taylor 1995; Malmsheimer 2004; and Smith 2005) evaluated federal agency success in cumulative impact litigation at the appellate level between 1995 and 2004. Most litigation is resolved at the district court level, however, so district-level opinions would also need to be examined to identify the most common operational problems with cumulative impact analysis.

Sixty-two federal district and appellate court opinions were examined to identify analytical deficiencies that were considered significant by litigants or judges. All cases were selected for review that fit the following criteria:

1. the case resulted in a final decision after January 1, 2000, that had been catalogued by LexisNexis as of June 1, 2005, or was accessible on the internet;

2. the complaint involved analysis of cumulative impacts; and either
3. the U.S. Forest Service was a defendant (39 district opinions, 13 appellate; representing 45 individual cases); or
4. the case involved forest or rangeland management (exclusive of mineral exploration) and the Bureau of Land Management was a defendant (9 district opinions, 2 appellate; 9 individual cases).

The USFS and BLM were co-defendants in one case. Of the 47 district-level opinions examined, 10 were appealed; 5 additional appellate opinions were reviewed for which district-level opinions were not available. Two of the district cases were combined into one of the appellate cases.

At the district court level, the Forest Service received adverse decisions regarding cumulative impact analysis in 21 of the 45 cases, and in two other cases, the Forest Service prevailed on cumulative impact issues but lost on other grounds. At the appellate level, the Forest Service received adverse decisions in six of 13 cases; six of the lower court decisions were reversed. Overall, the Forest Service prevailed in 47 percent of the final cumulative impact decisions for the 45 original cases. Regions 6 (Oregon and Washington; involved in 29 percent of the cases) and 9 (Lake States and Northeast; 20 percent) together accounted for nearly half of the cases. District courts under the 9th Circuit Court of Appeals handled 70 percent of the cases.

Twenty-nine of the 53 initial BLM and USFS cases (table 3) involved Environmental Assessments (EAs) leading to a "Finding of No Significant Impact" (FONSI). In most of these cases, plaintiffs argued that the EA does not adequately demonstrate that a significant impact is not likely because either

1. the analysis is technically invalid;
2. it does not consider the appropriate past, present, or foreseeable future actions; or
3. the finding of insignificance is insufficiently supported (in other words, "conclusory").

If the EA is found to be inadequate, either it must be revised to correct its deficiencies, or, if evidence is compelling that the evaluated impacts may be significant, an EIS

Table 3. Issues contested in cases involving cumulative impact analysis.

	All cases (EA, EIS, or CE)					EA		EIS	
Contested issue	**Number of cases[a]**	**Percent of cases[b]**	**Agency losses (number)**	**Category percent lost[c]**	**Total percent lost[d]**	**Number of cases**	**Cases lost**	**Number of cases**	**Cases loss**
Technically inadequate analysis	45	82	25	56	45	23	11	21	14
Combined activities disregarded	36	65	23	64	42	21	13	14	10
future activities	29	53	19	66	35	16	10	13	9
present activities	21	38	14	67	25	12	7	8	7
past activities	14	25	12	86	22	7	6	7	6
Conclusory non-significance	27	49	17	63	31	20	11	6	6
Other regulations violated	26	47	15	58	27	13	6	11	8
Data insufficient or out of date	18	33	10	56	18	9	6	9	4
Mitigation inadequately supported	17	31	11	65	20	10	7	7	4
Inappropriate tiering	17	31	15	88	27	9	9	8	6
Inappropriate analysis area	13	24	7	54	13	8	4	5	3
Other views inadequately disclosed	10	18	8	80	15	3	3	7	5
Specific impact not evaluated	10	18	9	90	16	6	5	3	3
Improper segmentation	6	11	4	67	7	2	1	4	3
Overall	**55**	**100**	**33**	**60**	**60**	**29**	**16**	**24**	**16**

[a] Two cases involving both an EA and an EIS are evaluated separately for each category.
[b] Percentage of the cases examined in which the issue was contested.
[c] Calculated from the number of decisions adverse to an agency on a particular issue divided by the number of cases in which the issue was contested (e.g., agencies received unfavorable opinions regarding analysis of future activities in 19 of the 29 cases regarding future activities, resulting in adverse decisions on that issue in 66 percent of those cases).
[d] Calculated from the number of decisions adverse to an agency on a particular issue divided by the total number of cases examined (e.g., agencies received unfavorable opinions regarding analysis of future activities in 19 of the 55 cases, resulting in adverse decisions on that issue in 35 percent of the cases).

must be prepared—the latter was required in nearly one-third of the adverse decisions. Plaintiffs ultimately prevailed on cumulative impact issues in 55 percent of the EA cases, five of them after appeal.

Twenty-two of the cases challenged an EIS, and two others challenged both an EA and an EIS. Agencies ultimately prevailed on cumulative impact issues for an EIS in only 33 percent of these cases. The significance of a prospective impact is not itself a central issue in most EIS cases because lack of impact is not a requirement for proceeding with a project. Instead, adequacy of the document is based primarily on whether it demonstrates that the agency has the appropriate information to allow an informed decision. However, significance becomes important indirectly because an informed decision cannot be made if an impact was omitted from the analysis because it was incorrectly determined to be insignificant. Although the commonly alleged deficiencies are similar to those raised for EA cases, technical challenges are slightly more prevalent in EIS cases. If the EIS is found to be inadequate, it must be revised to correct the deficiencies.

The two remaining cases concern Categorical Exclusions (CE), and these decisions focus on whether the planned activity fits the criteria needed to allow the activity to proceed without further NEPA documentation. Categorical Exclusions are based on an assumption that particular types of activities usually do not generate impacts and so usually do not require impact analysis. However, the Forest Service Handbook indicates that in some cases the generalization does not apply (FSH 1909.15 ch. 30.3.3). Excluded activities thus must be screened to ensure that the particular application of the activity will be as benign as expected, and if this initial evaluation suggests that the specific implementation of the activity may be problematic, additional analysis is required. Cumulative impact arguments can become important in such cases if the exclusion is being challenged on the basis of potential cumulative impacts.

Overall, the two most common complaints are inadequate technical evaluation and inadequate consideration of the impacts of other past, present, or future actions. Even though courts generally accord agency experts deference in technical issues, plaintiffs prevailed in 56 percent of the cases in which the quality of the technical evaluation was in question. The most common technical inadequacies are conclusory assessments unsupported by evidence and reliance on models that had not been validated for the conditions present. In addition, data were alleged to be insufficient or out of date in one-third of the cases, and plaintiffs prevailed in 56 percent of the cases in which data sufficiency was challenged.

Plaintiffs also prevailed in 64 percent of the cases in which adequacy of the evaluation of impacts from past, present, or reasonably future activities was at issue. The most frequent flaw in this category was the failure to include analysis of specific future actions that either (1) would follow logically from the proposed action (such as maintenance activities that necessarily follow if a particular activity is initiated) or (2) were already proposed and could have impacts that would overlap with those of the proposed activity. Analyses were generally considered flawed if past or future actions were simply listed or if activities on other ownerships were ignored.

The need to evaluate interactions between the effects of the proposed project and those of reasonably foreseeable future projects was carefully distinguished from the need to avoid improper segmentation of analyses: projects that are themselves “cumulative,” “similar,” or “related” need to be combined under a single NEPA document to avoid obscuring potential interactions between impacts. Courts generally provided agencies with the discretion to determine which projects should be combined for consideration in a single EA or EIS. However, even when segmentation was found to be justifiable, the individual projects needed to be evaluated as reasonably foreseeable future projects in each others’ EA or EIS—cumulative impact analysis could not be avoided by preparing multiple documents.

The third most common issue of contention was lack of support for determinations of non-significance. This issue was most common in EA litigation because a FONSI cannot be obtained unless the project will have no significant impact, but an EIS must also demonstrate non-significance for particular impacts if further analysis is not to be provided for those impacts. Findings of non-significance were determined to be inadequately supported for particular impacts in all six EISs for which the issue was raised.

Assessment of the effectiveness of mitigation becomes important in EAs if a potentially significant impact is found that must be mitigated to insignificance in order to obtain a FONSI. The effectiveness of mitigation thus was contested in 34 percent of the EA cases. Here, too, agency analyses tended to prevail unless no evidence was provided for mitigation effectiveness.

Twenty-seven percent of the documents were found to be flawed in part because they were tiered to other documents that were either outdated, incomplete, or were not themselves NEPA documents. "Tiering" under NEPA refers to the common practice of incorporating the findings of a more general NEPA document—such as an EIS for a Forest Plan—into a project-specific NEPA document to avoid duplication of effort. With tiering, the NEPA document can simply state that a particular analysis was carried out in an earlier NEPA document that has already undergone review; the earlier analysis is then not subject to additional review. However, an EA or EIS cannot tier to a document that has not already been reviewed under NEPA. Information from non-NEPA documents, such as watershed analyses and published papers, is expected to be used in an EIS or EA, but the NEPA document applies the information to the specific project in question. The apparent importance of tiering as a legal issue may in part reflect *post hoc* rationalizations for why analyses of particular issues were not included in an EIS or EA.

The area selected for the cumulative impact analysis was at issue in 24 percent of the cases. For this issue, too, courts are strongly inclined to defer to agency judgment unless the decision is not explained or clearly conflicts with legislative intent. Agencies lost on this issue, for example, if the analysis area was shown to exclude a potentially significant impact to which the planned activity would contribute.

Courts' Requirements for Valid Analyses

The most common flaws identified by federal courts in cumulative impact analyses prepared by federal land management agencies are similar to those found in private-sector analyses in northwest California In both contexts, misunderstandings of the definition of cumulative impacts and inappropriate evaluations of significance and mitigation effectiveness are common, as are technical errors. For many of the problem areas, courts have described the standards expected of a valid analysis as they explained why particular analyses are inadequate. Such instructions from an appellate court are held as precedent-setting by district courts in the same circuit (for example, a California or Idaho District Court would apply the standards set by the 9th Circuit Court of Appeals), and are also often cited by appellate courts in other circuits (for example, 9th Circuit decisions regarding cumulative impact analysis are often cited as a basis for opinions in other Circuits where such cases are uncommon). District-level opinions, in contrast, do not establish precedent and are rarely cited, but are useful indicators of how the precedents set by appellate courts are being applied.

General Standards for Evaluation

Cases concerning documentation required by NEPA are usually evaluated according to standards set by the Administrative Procedure Act. The Supreme Court described what constitutes a violation of that Act: "Normally, an agency rule would be arbitrary and capricious if the agency has relied on factors which Congress has not intended it to consider, entirely failed to consider an important aspect of the problem, offered an explanation for its decision that runs counter to the evidence before the agency, or is so implausible that it could not be ascribed to a difference in view or the product of agency expertise" (Motor Vehicle Mfrs. Assn. v. State Farm Mutual, 463 U.S. 29, Supreme Court 1983). Courts further indicate that an agency decision is not valid if it "is contrary to the governing law" (Lands Council v. Powell, 379 F.3d 738, 9th Cir. 2004), and that agency expertise need not be accorded deference if there is a "clear error of judgment" (Kern v. BLM, 284 F.3d, 9th Cir. 2002). In Earth Island Inst. v. USFS (351 F.3d 1291, 9th Cir. 2003) the 9th Circuit Court of Appeals noted that factual errors, too, may undermine agency discretion if it can be shown that an agency "unreasonably relied upon inaccurate data," because of the requirement that "Agencies shall insure the professional integrity, including the scientific integrity, of the discussions and analyses in environmental impact statements" (40 C.F.R. § 1502.24). Barring such conditions, however, courts generally defer to agency personnel on technical issues.

The governing laws in most litigation regarding USFS cumulative impact analyses include NEPA, the National Forest Management Act, the Endangered Species Act, and occasionally the Clean Water Act. Under NEPA, agencies are required to take a "hard look" at the environmental consequences of planned actions to ensure that agency decisions are fully informed. Litigation thus commonly focuses on whether agency personnel "abused their discretion" by approving a NEPA document that failed to "take a hard look" at environmental impacts.

Overall Analysis

Several appellate opinions discuss the types of information needed in cumulative impact analyses under NEPA. The 9th Circuit Court of Appeals explained, "To 'consider' cumulative effects, some quantified or detailed information is required. Without such information, neither the courts nor the public, in reviewing the Forest Service's decisions, can be assured that the Forest Service provided the hard look that it is required to provide...general statements about 'possible' effects and 'some risk' do not constitute a 'hard look' absent a justification regarding why more definitive information could not be provided" (Neighbors of Cuddy Mountain v. USFS, 137 F.3d 1372, 9th Cir. 1998).

Courts frequently stress the intended utility of the analysis: "The cumulative impact analysis must be more than perfunctory; it must provide a 'useful analysis of the cumulative impacts of past, present, and future projects' " (Kern v. United States Bureau of Land Mgmt. 284 F.3d 1075, 9th Cir. 2002). And again, "The EIS must analyze the combined effects of the actions in sufficient detail to be 'useful to the decisionmaker in deciding whether, or how, to alter the program to lessen cumulative impacts' " (Muckleshoot Indian Tribe v. USFS, 177 F.3d 800, 9th Cir. 1999).

The 1985 decision for *Fritiofson v. Alexander (*772 F.2d 1225, 5th Cir. 1985, reversed on other grounds by Sabine River Auth. v. Dep't of the Interior, 951 F.2d 669, 5th Cir. 1992) explained, "Given the CEQ regulations, it seems to us that a meaningful cumulative-effects study must identify:

1. the area in which the effects of the proposed project will be felt;
2. the impacts that are expected in that area from the proposed project;
3. other past, present, and reasonably foreseeable actions that have or are expected to have impacts in the area;
4. the impacts or expected impacts from these other actions; and
5. the overall impact that can be expected if the individual impacts are allowed to accumulate."

This list was subsequently cited by the DC. Circuit Court of Appeals in Grand Canyon Trust v. FAA (290 F.3d 339, DC Cir. 2002) and by district courts in the 1st, 2nd, 4th, 5th, and 11th Circuits.

Past, Present, and Reasonably Foreseeable Future Actions

Several decisions provide insight into requirements for evaluating the effects of other actions. According to the 9th Circuit, for example, "The general rule under NEPA is that, in assessing cumulative effects, the Environmental Impact Statement must give a sufficiently detailed catalogue of past, present, and future projects, and provide adequate analysis about how these projects, and differences between the projects, are thought to have impacted the environment" (Lands Council v. Powell, 379 F.3d 738, 9th Cir. 2004).

A district court opinion pointed out, in addition, that because the cumulative impact is the overall impact, the aggregate impact of past, present, and future actions would need to be evaluated (EPIC v. Blackwell, no. C-03-4396 EMC, N. Dist. California 2004). This comment underscores the central point of cumulative impact analysis: the cumulative impact on a resource is the overall anthropogenic impact experienced by the impacted resource.

The CEQ has recently provided additional guidance on this issue (Connaughton 2005), noting that a listing of individual past, present, and future projects is not necessarily required because it is the aggregate effect of projects that is important. However, individual projects would still need to be identified "if such information is necessary to describe the cumulative effect of all past actions combined." Courts are explicit in indicating that the present condition, representing an existing aggregated impact level, cannot be interpreted as the condition against which the cumulative level of impact is to be judged (see, for example, Grand Canyon Trust v. FAA, 290 F.3d 339, D.C. Cir. 2002). Instead, the current condition would need to be compared to the naturally occurring condition to identify the current and likely future levels of impact: the CEQ specifies that a description of the "baseline" condition of the resource of concern "should include a description of how conditions have changed over time and how they are likely to change in the future without the proposed action" (CEQ 1997). For example, the significance of a project's addition of 50 t yr^{-1} km^{-2} of sediment to an existing sediment load of 800 t yr^{-1} km^{-2} can be interpreted only if the naturally occurring sediment load is estimated. If the natural load were itself 800 t yr^{-1} km^{-2}, the cumulative post-project input of 6 percent over the natural background load would not likely be significant. In contrast, if the natural load were originally 30 t yr^{-1} km^{-2} and other human activities had already raised the pre-project load to 800 t yr^{-1} km^{-2}, the cumulative post-project input of about 2,700 percent over the natural background load would be highly significant. In the latter case, the existing aggregated impact level at the time of the project would already be about 2,600 percent over naturally occurring levels, and the project would add further to an already significant cumulative level of impact.

Information concerning individual past projects also may be needed to allow prediction of a project's direct and indirect impacts (Connaughton 2005), which then provides the basis for analyzing the project's contribution to the cumulative impact. This is the function noted by the 9th Circuit's opinion in Lands Council v. Powell (379 F.3d 738, 9th Cir. 2004) regarding the need to "...provide adequate analysis about how these projects, and differences between the projects, are thought to have impacted the environment," and by an opinion from a district Court: "While this argument [assessing current aggregated impact levels without identifying activities that contributed to the current levels] may have some validity in that current conditions will obviously reflect in some measure the effects of past environmental degradation, it ignores the failure of the Whiskey South EA to discuss in sufficient detail the connection between prior activities and the current project, which is critical to understanding what alternatives may produce the least environmental harm while still meeting project goals" (Idaho Conservation League v. Bennett, CV 04-447-S-MHW, District of Idaho, 29 Apr 2005). Unless the projects that led to existing impacts are identified and their characteristics are contrasted with those of the proposed project, it is difficult to support an argument that the outcome from proposed activities will be different than in the past.

Incorporation of Professional Opinion

Courts do not consider simple reliance on professional judgment to be adequate—decisions based on expert opinion are considered "conclusory" unless some supporting evidence is provided. The 9th Circuit explained the limitations of professional judgment: "...allowing the Forest Service to rely on expert opinion without hard data either vitiates a plaintiff's ability to challenge an agency action or results in the courts second guessing an agency's scientific conclusions. As both of these results are unacceptable, we conclude that NEPA requires that the public receive the underlying environmental data from which a Forest Service expert derived her opinion. In so finding, we note that NEPA's implementing regulations require agencies to "identify any methodologies used and ... make explicit reference by footnote to the scientific and other sources relied upon for conclusions" used in any EIS statement. 40 C.F.R. § 1502.24." (Idaho Sporting Congress v. Thomas, 137 F.3d 1146, 9th Cir. 1998; internal citations omitted). In contrast, when the basis for the expert opinion is documented and supporting evidence is provided, courts routinely defer to the opinions of agency experts on technical matters (for example, Shenandoah Ecosystems Defense Group v. USFS, 144 F. Supp. 2d 242, W. Dist. Virginia, Harrisonburg Div., 2001).

Courts have also indicated the importance of disclosing and discussing reasonable opposing opinions or alternative interpretations: "...the court should 'ensure that the statement contains sufficient discussion of the relevant issues and opposing viewpoints to enable the decisionmaker to take a 'hard look' at environmental factors, and to make a reasoned decision'" (Natural Resources Defense Council v. Hodel, 865 F.2d 288, D.C. Cir. 1988; and Izaak Walton League of America v. Marsh, 655 F.2d 346, D.C. Cir. 1981).

Reliance on Modeling

Courts often find expert opinion to be inadequately supported if the experts have relied on modeling tools that have not been validated for the particular areas or conditions to which they were applied. In a recent case, the 9th Circuit noted, "The Forest Service, granted appropriate deference, still does not demonstrate the required reliability of the spreadsheet model. We are asked to trust the Forest Service's internal conclusions of the reliability of the spreadsheet model when the Forest Service did not verify the predictions of the spreadsheet model. Under the circumstances of this case, the Forest Service's basic scientific methodology, to be reliable, required that the hypothesis and prediction of the model be verified with observation" (Lands Council v. Powell, 379 F.3d 738, 9th Cir. 2004).

The same decision explained that "NEPA requires that the Environmental Impact Statement contain high-quality information and accurate scientific analysis. 40 C.F.R. § 1500.1(b). If there is incomplete or unavailable relevant data, the Environmental Impact Statement must disclose this fact. 40 C.F.R. § 1502.22." On this basis, the court also found the use of a model relating hydrologic change to increased sedimentation to be inappropriate: "The Forest Service's heavy reliance on the WATSED model in this case does not meet the regulatory requirements because there was inadequate disclosure that the model's consideration of relevant variables is incomplete. Moreover, the Forest Service knew that WATSED had shortcomings, and yet did not disclose these shortcomings until the agency's decision was challenged on the administrative appeal. We hold that this withholding of information violated NEPA, which requires up-front disclosures of relevant shortcomings in the data or models."

Assessing Impact Significance

Conclusory statements are particularly prevalent in findings that impacts will not be significant. Accordingly, the 9th Circuit stressed that "An agency's decision not to prepare an EIS will be considered unreasonable if the agency fails to supply a convincing statement of reasons why potential effects are insignificant" (Save the Yaak Committee v. Block, 840 F.2d 714, 9th Cir. 1988).

In assessments of significance, courts frequently refer explicitly or implicitly to the CEQ definition of cumulative impact to determine whether the intent of the regulation is being met. For example, a district court in Ohio found that a Forest Service analysis of significance did not actually address cumulative impacts: "...The Forest Service considers the following statement to be cumulative impact analysis: 'If the 150 bats wintering on the Forest were to be lost for any reason, over 99.9 percent of the total population would remain unaffected.' This is not cumulative impact analysis. The Forest

Service fails to acknowledge whether all of the other federal agencies with populations of Indiana bats present take such a cavalier attitude toward the preservation and recovery of the species. If each National Forest, National Refuge, National Park, and other federal lands does take such an attitude, then the effects are truly significant and, in layman's terms, catastrophic... " (Buckeye Forest Council v. United States Forest Service, No. 1:04-Cv-259, S. Dist. Ohio, W. Div., 2004). The Forest Service analysis was clearly inconsistent with the CEQ definition: "...Cumulative impacts can result from individually minor but collectively significant actions...."

Legal opinions provide additional guidance concerning the standards against which significance is to be assessed. For example, the Forest Service had argued in an EA that sediment inputs from salvage logging following a wildfire would be insignificant because they would be small compared to those caused by the fire. The Court responded, "Whether the increased erosion from logging and roadbuilding is smaller or larger than that produced by the fire is irrelevant. The proper evaluation should identify the impact of the increased sediment from the logging and roadbuilding on the fisheries habitat in light of the documented increases that already have resulted from the fire" (Blue Mountains v. Blackwood, 161 F.3d 1208, 9th Cir. 1998).

Similarly, defendants in a case not included in the present survey argued that the relevant standard against which to evaluate the significance of an impact is the current condition, the "no-action alternative." By doing so, they found that the proposed project would not provide a significant increase in impact over existing levels. The court, however, held the argument untenable, reaffirming that significance is to be evaluated for the overall impact and not for the incremental addition caused by the project: "Because there is no analysis of cumulative noise impact on the Park against which the additional noise impact of the replacement airport can be evaluated, the FAA's error in ignoring cumulative impact of man-made noise is not harmless...for the FAA has impermissibly taken 'a foreshortened view of the impacts which could result from the act' of constructing the replacement airport" (Grand Canyon Trust v. FAA, 290 F.3d 339, D.C. Cir. 2002).

Once the overall impact level is evaluated, assessment of the significance of that impact level must then take into account both the context of the impact and its intensity, and regulations identify particular aspects of intensity that should be considered (40 CFR 1508.27). As noted by the regulations (table 4), other governing legislation may establish specific criteria for significance. For example, if a waterway is listed as impaired

Table 4. Factors to consider for evaluating impact "significance" under NEPA (40 CFR 1508.27).

(a) **Context.** This means that the significance of an action must be analyzed in several contexts, such as society as a whole (human, national), the affected region, the affected interests, and the locality. Significance varies with the setting of the proposed action. For instance, in the case of a site-specific action, significance would usually depend upon the effects in the locale rather than in the world as a whole. Both short- and long-term effects are relevant.

(b) **Intensity.** This refers to the severity of impact. Responsible officials must bear in mind that more than one agency may make decisions about partial aspects of a major action. The following should be considered in evaluating intensity

- (1) Impacts that may be both beneficial and adverse. A significant effect may exist even if the Federal agency believes that on balance the effect will be beneficial.
- (2) The degree to which the proposed action affects public health or safety.
- (3) Unique characteristics of the geographic area such as proximity to historic or cultural resources, park lands, prime farmlands, wetlands, wild and scenic rivers, or ecologically critical areas.
- (4) The degree to which the effects on the quality of the human environment are likely to be highly controversial.
- (5) The degree to which the possible effects on the human environment are highly uncertain or involve unique or unknown risks.
- (6) The degree to which the action may establish a precedent for future actions with significant effects or represents a decision in principle about a future consideration.
- (7) Whether the action is related to other actions with individually insignificant but cumulatively significant impacts. Significance exists if it is reasonable to anticipate a cumulatively significant impact on the environment. Significance cannot be avoided by terming an action temporary or by breaking it down into small component parts.
- (8) The degree to which the action may adversely affect districts, sites, highways, structures, or objects listed in or eligible for listing in the National Register of Historic Places or may cause loss or destruction of significant scientific, cultural, or historical resources.
- (9) The degree to which the action may adversely affect an endangered or threatened species or its habitat that has been determined to be critical under the Endangered Species Act of 1973.
- (10) Whether the action threatens a violation of Federal, State, or local law or requirements imposed for the protection of the environment.

under section 303(d) of the Clean Water Act, the impact level is already deemed significant by regulatory agencies.

Effectiveness of Mitigation

If an impact being evaluated in an EA is potentially significant, plans often include a mitigation program to reduce the overall impact to insignificant levels, and this practice creates additional analytical challenges because the document must then evaluate the effectiveness of mitigation (FSH 1909.15, Ch.10.15). Courts then may need to determine whether the mitigation plans are indeed adequate to ensure that impacts will be insignificant: "In evaluating the sufficiency of mitigation measures, we consider whether they constitute an adequate buffer against the negative impacts that may result from the authorized activity. Specifically, we examine whether the mitigation measures will render such impacts so minor as to not warrant an EIS" (National Parks and Conservation Association v. Babbitt, 241 F.3d 722, 9th Cir. 2001).

A District Court opinion further indicates the kind of analysis considered appropriate: "BLM...relied entirely upon future permits and mitigation plans in determining that these impacts would not be significant.... The mitigation measures must be 'developed to a reasonable degree,' however, and neither a 'perfunctory description' nor a 'mere listing' of measures, in the absence of 'supporting analytical data,' is sufficient to sustain a finding of no significant impact" (Western Land Exchange Project v. BLM, 315 F. Supp. 2d 1068, Dist. Nevada 2004; internal citations omitted). The same opinion also notes that BLM intended to institute a monitoring program to determine the nature of likely impacts and plan future mitigation efforts, but "Where research is necessary to determine the extent of an unknown and possibly significant environmental risk, an EIS should be prepared so that the research can be done and the decision made in reliance on that information, rather than the other way around....BLM's reliance on an unwritten, untested, and unsupported monitoring plan in the face of unknown and possibly significant environmental impacts does not excuse its decision to forego an EIS" (internal citations omitted).

Courts also have noted that the timing of mitigation is important if impacts are to be reduced to insignificance. In Klamath-Siskiyou Wildlands Center v. USFS (no. Civ. S 03-1334 FCD DAD, N. Dist. California 2004), mitigation work was scheduled after the planned logging, so the near-term impacts of logging on runoff would not be offset by mitigation: "The fact remains, however, that short term impacts to the watershed will occur, and those impacts appear significant and highly uncertain. Neither the net long term benefits of the program, nor the risk associated with not implementing the project, relieve the Forest Service of its duty to conduct an EIS when the project will have significant environmental impacts. 40 C.F.R. §1508.27(a)."

Conclusions

As federal courts repeatedly note, the role of NEPA documents is to demonstrate that there has been a "hard look" taken at the potential impacts of a project and that those impacts are understood well enough that surprises are unlikely. It is then the responsibility of agency personnel to decide how to use the resulting information: "NEPA merely prohibits uninformed—rather than unwise—agency action" (Robertson v. Methow Valley Citizens Council, 490 U.S. 332, Supreme Court 1989). Given the guidance provided by the CEQ and the federal courts, it is clear that the preparation of a useful cumulative impact analysis is not unduly complicated or onerous ("NEPA does not require the government to do the impractical" Inland Empire Public Lands Council v. USFS, 88 F.3d 754, 9th Cir. 1996).

Much of the confusion surrounding analysis of cumulative watershed impacts for forest management applications appears to arise from misunderstandings of the nature of cumulative impacts. In this context, "cumulative" simply means "total" or "combined";

a directive to evaluate the cumulative impact on a resource simply means that the overall impact on that resource is to be assessed. The influence of an individual project is then evaluated as the extent to which that project will augment or diminish the overall impact level for the resource. Such an analysis would take into account potential interactions between environmental changes caused by the project and by past, present, and reasonably foreseeable future projects. Spatial and temporal scales for analysis would be determined by the spatial and temporal scales at which potential impacts would be expressed, and would vary by impact. Similarly, the level of quantitative rigor employed would be selected to be appropriate for the context and issues present.

Such an approach might be facilitated by rephrasing the underlying question from "What are the cumulative impacts of the project?" to "What are the impacts of concern in the area, and how might the proposed project influence those impacts?" With this reformulation, the focus is shifted to specific impacts, those impacts are defined from the perspective of the impacted resources, and the problem becomes recognizable as one that agency resource specialists already know how to address.

Most of the problems encountered with the private-sector and agency analyses that were examined would have been avoided had the analyses been subject to rigorous technical review and had the procedure outlined by CEQ (1997) been followed. The analysis steps required for such an evaluation have already been carried out to some extent if a watershed analysis is available for the area in question. Such analyses should already outline the impacts of concern and their severity, distribution, and causal mechanisms. Watershed analysis carried out under the Northwest Forest Plan (REO 1995), for example, accomplishes steps 1, 2, 3, 4, 6, 7, and 8 of the CEQ procedure outlined in table 1. Although a project-based NEPA document cannot simply state that cumulative impacts have already been analyzed through watershed analysis (in other words, it cannot "tier" to a watershed analysis), it can take advantage of the analytical work already done by watershed analysis in the same way that it makes use of other scientific literature, as long as the "...study is reasonably available to the interested public" (Connaughton 2005). The increment added by the NEPA document would then evaluate how the particular project influences the mechanisms and impacts already evaluated by the watershed analysis. This approach has the added advantage of efficiency: a single watershed analysis can provide the background information necessary for analyzing the cumulative watershed impacts of any number of projects occurring within the watershed. The most useful route to development of effective cumulative impact assessments for federally managed forestlands may be through adaptation of the watershed analysis procedure to more consistently provide the information that will be needed by future cumulative impact analyses.

Whatever procedure is used for analysis, three issues remain that are of particular importance. First, more guidance would be useful for assessing the significance of impacts. The CEQ has provided a list of factors to be considered to assess significance (table 4) and calls for evaluating the capacity of a resource to withstand additional change, but the ultimate determination of significance is often a value-based decision. Development of a defensible procedure for making such determinations would simplify future analyses. Second, the information needed to determine whether past analyses provided accurate predictions is not available. A study to compare on-the-ground outcomes with predicted impact levels would be useful, as would a program to monitor the accuracy of future analyses. Only through such work can analytical procedures, mitigation, and management practices be improved. Finally, and perhaps most importantly, the quality of any analysis ultimately depends on the expertise and technical knowledge of those preparing the analysis.

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