The Great Trinity Forest Management Plan

Volume 10

Wildland-Urban Interface

GREAT TRINITY FOREST

The Wildland-Urban Interface

Discussion of wildland fire and prescribed burning within the forest.

Volume 10

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FIRE EFFECTS GUIDE

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Fire Effects Guide

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PREFACE

by Dr. Bob Clark and Melanie Miller

A. Purpose

The Federal government manages a variety of ecosystems across the United States, including deserts, grasslands, tundra, shrublands, forestlands, estuaries, and riparian zones. These ecosystems range from arid to humid, warm to cold, and sea level to over 10,000 feet elevation. Fires naturally occur in almost all of these ecosystems, with fire characteristics determined by climate, vegetation, and terrain.

The purposes of this Guide are to summarize available information on fire effects principles and processes, provide references for additional information, and provide guidelines for the collection, analysis, and evaluation of wild and prescribed fire effects data. Basic mechanisms of fire effects are described so that the reader will be able to understand and interpret fire effects literature, and evaluate observed results that conflict with those presented in published reports. The goal is to improve fire management by improving our ability to manage fire effects.

The Guide was written as an aid for resource managers and fire managers. It can be used for managing and evaluating wildfires; developing and implementing emergency fire rehabilitation plans; planning, monitoring, and evaluating prescribed fires; developing activity plans such as timber management plans, allotment management plans, and threatened and endangered species recovery plans; and providing fire management input for land use plans.

B. Assumptions

Ecosystems have evolved with, and adapted to, specific fire regimes. In a particular ecosystem, natural fires occurred with fairly specific, albeit irregular, frequency and typical season of occurrence; with characteristic fireline intensity and severity; and characteristically did or did not involve the crowns of trees or shrubs. Gross differences occurred among ecosystems. For example, frequent, low intensity, surface fires were common in ponderosa pine ecosystems, whereas fires in big sagebrush were probably less frequent, of higher intensity, and killed much of the sagebrush overstory. High intensity, stand replacement fires at long intervals were characteristic of some forest types, while annual fires may have been common on some Great Plains grasslands. Despite this variability in fire regimes, universal principles and processes govern response of ecosystem components to fire. Recognition and understanding of the principles and processes can help our understanding of the variability in postfire effects that is often reported in the literature, and differences between reported results and local observations on burned areas. This knowledge will enable resource and fire managers to predict and evaluate fire effects, regardless of ecosystem or fire regime.

Fire effects are the result of an interaction between the heat regime created by the fire and the properties of ecosystem components present on the site. For example, plant species in vegetation types that have evolved with frequent fire tend to be much more resistant to fire than species from plant communities that rarely burned. The effects of a fire burning under the same conditions may be very different on soils of different textures or chemical properties. Variation in fire effects may also occur within ecosystems because of differences in site characteristics, fuel conditions, and weather prior to, during, and after the fire. A fire may have different effects upon the same site if it occurs in different seasons or within the same season but with different fuel, duff, and soil moisture. For these reasons, it is important to document conditions under which the fire occurred, and the characteristics of the fire, as part of any effort to monitor postfire effects.

The words fire intensity, severity, fireline intensity, and burn severity are often used interchangeably in the literature. The following terminology is used throughout this Handbook to describe the properties of fire. All definitions that describe the behavior of a flaming fire are those used in the Fire Behavior Prediction System (Rothermel 1983), including fireline intensity, the rate of heat release per linear foot of the flaming front. Burn severity is a qualitative assessment of the heat pulse toward the ground, and relates to subsurface heating, large fuel and duff consumption, and consumption of litter and organic layers beneath isolated trees and shrubs. The terms fire intensity and intensity are used by some authors to describe the overall heat regime of a fire. They are generic terms that are often confused with fireline intensity, and are not used as a synonym for fireline intensity in this Handbook.

The Guide recognizes that a natural fire regime cannot be perpetuated in unnatural communities. Timber harvest practices, grazing patterns and degree of use, the accidental or deliberate introduction of exotic plants and animals, other cultural activities that alter fuel continuity and loading, and the modification of historic fire patterns through active suppression have changed many plant communities. Interruption of fuel continuity by livestock grazing, road construction, and other developments has resulted in fires that are less frequent, smaller, and of lower fireline intensity, in some ecosystems. The introduction of exotic plants such as cheatgrass, coupled with anthropogenic ignition sources, has greatly increased fire size and frequency in other ecosystems.

Active suppression has resulted in large areas, especially in shrublands and forests, that are extremely susceptible to fire. The exclusion of fire has resulted in a larger proportion of vegetation in older age classes, except in regulated forests, which are more susceptible to insect and disease infestations. The amount of dead plant material has increased, either accumulated on the ground or retained on plants. In plant communities with historically short fire cycles, the absence of fire has allowed the development of fuel ladders between the surface and the overstory. Fires which do occur are often carried into the tree crowns by large accumulations of down dead woody fuels or understory trees, causing a stand replacement fire in forest types where historically, overstory trees were rarely killed. In vegetation communities with long natural fire cycles, younger, intermixed, less flammable age classes of vegetation are not as prevalent as they would have been under a natural fire regime. Coupled with the increased incidence of insect and disease, the continuity of highly flammable stands has increased, resulting in greater potential for extremely large fires. Plans for the

C. Handbook Organization

The chapters of this Guide discuss different elements that relate to our management of fire effects and specific responses of different ecosystem components to fire. This Handbook recognizes that separate discussions of fire effects on fuels, soils, watershed, plants, and wildlife are artificial, because fire effects are an integration of the responses of all of these components to fire. Despite the fact that fire effects occur holistically, ecosystem components are discussed individually as a means of organizing the information. Chapters describe basic principles and processes that regulate fire effects, including fire behavior and characteristics, fuels, air quality, soils and watershed, plants, wildlife, and cultural values. Considerations for management of fire effects on these resources, and a discussion of appropriate techniques for monitoring fire effects, are contained in each of these chapters. Monitoring is included in this Handbook because techniques that accurately describe long-term trends in plant community condition, for example, are not adequate to detect significant and sudden changes caused by burning. Because an understanding of prefire and postfire grazing management, data analysis, and documentation and evaluation procedures is critical to sound management and monitoring of fire effects, chapters on each of these topics are also included. Resource management is goal oriented. The first chapter in this Guide is a discussion of goals and objectives and how they fit into planning for the use and management of fire.

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CHAPTER I - DEVELOPMENT OF OBJECTIVES

By Dr. Tom Zimmerman

A. Introduction

Management is the process of anticipating the future, setting objectives, implementing an action, achieving an output, and performing an evaluation comparing the output to the objective. Management is not possible without setting objectives. Clear and easily communicated objectives facilitate the management process.

In land management programs, the desired outcome of management actions is expressed as management objectives. Objectives represent an important component of all land management programs and are the single most important factor driving all management actions.

B. Definitions and Qualities of Good Objectives

1. Goals and Objectives - Definition. In land management, both goals and objectives are important. Goals are primary and basic products of the long range management plans. These goals are commonly referred to as land use decisions. Goals are relatively short statements that discuss what the public lands are to be used for and where the uses will occur. Each statement addresses a land use, but is not limited to the principal or major use.

Objectives are a necessary component of the planning process; they provide a bridge between goals and the implementation phase. Objectives describe what procedures will be used and when actions will be completed.

During the planning of fire management projects, objectives are formulated and used as the basis for development of an action plan. Interdisciplinary (I.D.) teams coordinate various concerns and develop objectives for a project. The I.D. teams are composed of resource specialists from different disciplines who address concerns of the affected resources and resolve conflicts among resource disciplines that arise from specific management actions.

2. Qualities of Good Objectives. Fire management objectives must be made up of certain attributes or they will not convey the necessary guidance. Good objectives must be informative and **SMART**. Objectives that are **SMART** are:

S - Specific - what will be accomplished, using limiting factors, and identifying the range of acceptable change from the present to the proposed condition.

M - Measurable - the present and proposed condition must be quantifiable and measurable.

A - Achievable - can be achieved within a designated time period.

R - Related/Relevant - related in all instances to the land use plan goals and relevant to current fire management practices.

T - Trackable - objectives must be trackable over time and must include a definite timeframe for achievement, monitoring, and evaluation.

3. Kinds of Objectives.

a. Land use decisions (goals). These are broad statements, usually specified in land management plans, that deal with large areas over long time periods (e.g., 10 years). Land use decisions establish resource condition objectives; the allowable, limited, or excluded uses for an area (land use allocations) and the terms and conditions for such use; and management actions that will be taken to accomplish multiple use goals.

b. Resource management objectives. Resource management objectives identify the changes in water, soil, air, or vegetation from the present to proposed conditions. Resource objectives can also describe an existing resource condition that should be maintained.

c. Treatment objectives. These are very well-defined statements that describe what a treatment must accomplish in order to meet a stated resource management objective. This type of objective is site-specific and must utilize the **SMART** concept.

Any statement that is an objective **must** identify the change from present conditions to the proposed conditions (the changes that are planned) and the limiting factors.

C. How Objectives Relate to Project Inventory, Development, Implementation, Monitoring, and Evaluation

Objectives are an important part of management actions and are prerequisite to sound land and resource management. Objectives not only drive the planning system, they also drive the full spectrum of project implementation, monitoring, and evaluation.

During the fire planning process, for example, the planner uses resource management objectives (standards) as guidance to determine what fire management responses and activities are necessary. These standards then provide guidance in determining what and how much information should be collected prior to and during project implementation. At this point, knowledge of fire effects becomes a necessary part of the planning process. Fire effects information helps to determine what will be done, how many resources are needed, how much funding the fire program will need, and what should be evaluated to ensure efficient accomplishment of the workload.

D. Relationships of Different Tiers (Levels) of Planning to Objectives

Generally, objectives start as issues when the land use planning process is initiated. (Issues are usually conflicts between two or more resource uses or demands that must be resolved in the plan.) Issues are generally defined in terms of the desired state of achievement for environmental values and socioeconomic conditions affected by management activities and resource decisions. The next step is development of alternatives that include a range of ways to resolve the issues. After the preferred alternative is selected, local guidance for resource functions is developed that contains resource management objectives.

Land use planning systems used by most federal agencies are divided into five distinct tiers: national, geographically defined management areas, individual resource functions, and strategic and tactical site specific implementation (Table I-1). National policy is established in public laws, federal regulations, Executive Orders, and other Presidential, Secretarial, and Director approved documents. Policy guidance for planning is developed, as needed, through interpretation of national policy, public participation activities, and from coordination and consultation with other federal agencies.

Table I-1: Relationship of Planning Tiers to Fire Management Objectives, Products and Fire Effects Applications

Planning Tier	Type of Objectives	Product	Fire Effects Applications
National		Policy and Regulations	National policies and guidance regarding fire presence and exclusion in wildland ecosystems
Geographically defined management area	Land use decisions	Resource Management Plan (BLM, NPS) Comprehensive Conservation Plan (FWS) Integrated Resource Management Plan (BIA) Forest Land Management Plan (FS)	Integration of fire and resource management within a geographically defined management area

Local guidance for individual resource components	Resource management objectives	Habitat Management Plan Compartment Plan Allotment Plan	The role of fire in resource management within a specific administrative unit.
Strategic site specific implementation	Strategic program objectives	Fire Management Activity Plans Fire Management Plan Fire Management Action Plan Wilderness Fire Management Plan	Identification of appropriate allocation of fire suppression, fire use and fuels management activities necessary to achieve resource management objectives
Tactical site specific implementation	Treatment objectives	Prevention Plan Presuppression Plan Escaped Fire Situation Analysis Postfire Rehabilitation Plan Prescribed Fire Plan Other	Interpretation and analysis of site specific fire effects to guide development and implementation of a program of action to accomplish treatment objectives

Resource management plans developed for geographically defined management areas establish the combinations of land and resource uses; related levels of investment, production, and/or protection to be maintained; and general management practices and constraints for various public land resources. These are set forth as the terms, conditions, and decisions that apply to management activities and operations and are presented in the form of multiple-use prescriptions and plan elements.

The third planning tier, developed at a local level, provide guidance for individual resource functions. At this level the role of fire is discussed, and how fire can be used or is detrimental in achieving the individual resource objectives.

Site specific stratigic and tactical implementation plans are the final step in the fire planning process. The primary role of these plans is to identify operational guidance to accomplish site specific treatment objectives. To continue with the example of the fire management component, Fire Management Activity Plans delineate areas to receive different levels of fire suppression, fire use, and fuels treatment. Resource management objectives developed at this level are derived directly from land use decisions. Prescribed fire plans refer to resource management objectives developed in activity plans and identify treatment objectives. Resource management objectives referenced in prescribed fire plans describe second order fire effects, the indirect effects of fire treatment that occur over the longer term, such as increased plant productivity, changes in species composition, or increased off-site water yield. Fire treatment objectives are developed from the resource management objectives and state exactly what immediate effects the fire must create in order to achieve the resource objectives. Fire treatment objectives describe first order fire effects such as plant mortality, fuel reduction, or duff consumption. An example of a fire treatment objective is: to remove 90 percent of existing sagebrush crown cover, using fireline intensities that consume sagebrush crowns, leaving residual stems that are six inches or less in height.

E. Summary

Land management programs are objective driven. Objectives must be based on an amount of information sufficient to determine if a change from the present condition to the proposed condition can be achieved. Establishing objectives is a task of major importance and deserves an allotment of sufficient attention and time. Both objectives and fire effects information become more precise as site specificity increases.

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CHAPTER II - FIRE BEHAVIOR AND CHARACTERISTICS

by Melanie Miller

A. Introduction

Frequently a fire is described as hot or cool, high or low intensity, or flaming or smoldering. Too often fire behavior and characteristics are not described at all. Standard terminology exists for describing the behavior, characteristics, and heat regime of wildland fires. Monitoring and documentation of fire behavior and characteristics according to these standard terms can increase understanding of the relationship between the effects created by a specific fire and that fire's heat regime, and make comparisons among different fires possible.

The behavior of the flaming front of a surface fire can be predicted with fire behavior technology. Other characteristics of the surface fire, such as duration of all phases of combustion and penetration of heat into duff and soil layers cannot be predicted with existing models. The rate of heat release and growth of crown fires can be estimated. It is important to understand some of the different properties a wildland fire can have, how they can be described, which can be predicted, and how the various aspects of a fire's heat regime can be related to the fire treatment.

This chapter contains a brief overview of principles of fire behavior and characteristics. More detailed information can be obtained from formal courses in fire behavior and in courses that are prerequisites for certain prescribed fire positions. The chemistry and phases of the combustion process are described in the Air Quality chapter of this Guide, Chapter IV.B.1. The effect of fuel moisture on fire behavior is described in this chapter, but factors affecting fuel moisture content and fuel consumption are discussed in Chapter III.B, this Guide.

B. Principles and Processes

1. The Fire Environment. Wildland fire is influenced by three interacting classes of variables: fuels, topography, and air mass (Countryman 1972).

a. Fuels. Wildland fuels provide the energy source for fire. Fuels consist of both living and dead vegetation, the latter in various stages of decay. Fuels occur in three fairly distinct strata: ground, surface, and aerial. A fire can burn in one, two, or all three strata at once, or change the layer in which it is burning as fuels and environmental conditions change throughout an area. Fuels are discussed in greater detail in Chapter III. B.1., this Guide.

(1) Ground fuels. Ground fuels are all combustible materials below the surface litter layer. These fuels may be partially decomposed, such as forest soil organic layers (duff), dead moss and lichen layers, punky wood, and deep organic layers (peat), or may be living plant material, such as tree and shrub roots.

(2) Surface fuels. Surface fuels are those on the surface of the ground, consisting of leaf and needle litter, dead branch material, downed logs, bark, tree cones, and low stature living plants.

(3) Aerial fuels. Aerial fuels are the strata that is above the surface fuels and include all parts of tree and tall shrub crowns. The aerial fuel layer consists of needles, leaves, twigs, branches, stems, and bark, and living and dead plants that occur in the crowns such as vines, moss, and lichens.

(4) Ladder fuels. Ladder fuels bridge the gap between surface and aerial fuels. Fuels such as tall conifer reproduction can carry a fire from the surface fuel layer into tree crowns.

b. Topography. Topography includes slope, aspect, elevation, and how these elements are configured. Topography can change suddenly, particularly in mountainous terrain, and its influence on fire behavior can rapidly change as well.

(1) Direct effect.

(a) Slope is an extremely important factor in fire behavior because the flames of a fire burning upslope are positioned closer to the fuels ahead of the fire. This dries and preheats the fuels at a greater rate than if they were on flat terrain.

(b) Topography channels wind and can create turbulence and eddies that affect fire behavior. Topography also affects diurnal air movement, influencing the velocity of day time upslope and night time downslope winds.

(2) Indirect effect.

(a) The combined effects of aspect and elevation create different microclimates that affect vegetation distribution and hence fuel type.

(b) Fuel moisture can vary with aspect, elevation, and vegetation type. This is discussed further in Chapter III.B.4, this Handbook.

c. Air mass. Weather components such as temperature, relative humidity, windspeed and direction, cloud cover, precipitation amount and duration, and atmospheric stability are all elements of the air mass. These values can change quickly over time, and significantly with differences in aspect and elevation. The air mass affects fire both by regulating the moisture content of fuel (discussed in Chapter III.B.4.), and by its direct effect on the rate of combustion. The following is a brief discussion of the effect of air mass factors on fire behavior and characteristics.

(1) Temperature. Atmospheric temperature affects fuel temperature. The ease of ignition, the amount of heating required to raise fuel to ignition temperature (320 C.; 608 F.) (Burgan and Rothermel 1984), depends on initial fuel temperature. The most important effect of temperature, however, is its effect on relative humidity and hence on dead fuel moisture content. (See Chapter III.B.4.).

(2) Windspeed. Wind has a significant effect on fire spread. It provides oxygen to the fuel and, combined with slope, determines which way the fire moves. Wind tips the flame forward and causes direct flame contact with fuel ahead of the fire (Burgan and Rothermel 1984). These fuels are preheated and dried by this increased transfer of radiant and convective heat. Windspeed has the most influence on fire behavior in fuel types with a lot of fine fuels, such as grasslands.

2. Combustion Process.

a. Two stage process. Within a wildland fire, the processes of pyrolysis and combustion occur simultaneously (Ryan and McMahon 1976 in Sandberg et al. 1978).

(1) Pyrolysis. When first heated, fuels produce water vapor and mostly noncombustible gases (Countryman 1976). Further heating initiates pyrolysis, the process by which heat causes chemical decomposition of fuel materials, yielding organic vapors and charcoal (ibid.). At about 400F. (204 C)., significant amounts of combustible gases are generated. Also at this temperature, chemical reactions start to produce heat, causing pyrolysis to be self-sustaining if heat loss from the fuel is small. Peak production of combustible products occurs at when the fuels are about 600 F. (316 C.) (ibid.).

(2) Combustion. Combustion is the process during which combustible gases and charcoal combine with oxygen and release energy that was stored in the fuel (Countryman 1976) as heat and light.

b. Phases of combustion. The following summary is derived from Ryan and McMahon (1976 in Sandberg et al. 1978), except where noted. For a more complete discussion of the phases of combustion, see Sandberg et al. (1978).

(1) Pre-ignition phase. In this phase, heat from an ignition source or the flaming front heats adjacent fuel elements. Water evaporates from fuels and the process of pyrolysis occurs, the heat-induced decomposition of organic compounds in fuels.

(2) Flaming phase. Combustible gases and vapors resulting from pyrolysis rise above the fuels and mix with oxygen. Flaming occurs if they are heated to the ignition point of 800 to 900F. (427 to 482 C.), or if they come into contact with something hot enough to ignite them, such as flames from the fire front (Countryman 1976). The heat from the flaming reaction accelerates the rate of pyrolysis. This causes the release of greater quantities of combustible gases, which also oxidize, causing increased amounts of flaming (Ryan and McMahon 1976 in Sandberg et al. 1978).

(3) Glowing phase. When a fire reaches the glowing phase, most of the volatile gases have been driven off. Oxygen comes into direct contact with the surface of the charred fuel. As the fuel oxidizes, it burns with a characteristic glow. This process continues until the temperature drops so low that combustion can no longer occur, or until all combustible materials are gone.

(4) Smoldering phase. Smoldering is a very smoky process occurring after the active flaming front has passed. Combustible gases are still being released by the process of pyrolysis, but the rate of release and the temperatures maintained are not high enough to maintain flaming combustion. Smoldering generally occurs in fuel beds with fine packed fuels and limited oxygen flow such as duff and punky wood. An ash layer on these fuel beds and on woody fuels can promote smoldering by separating the reaction zone from atmospheric oxygen (Hartford 1993).

3. Fire Behavior Prediction. The Fire Behavior Prediction System is a collection of mathematical models that were primarily developed to predict the behavior of wildland fires (Rothermel 1983). The models include those used to forecast behavior, area and perimeter growth of a surface fire; models that estimate spot fire potential, crowning potential, and crown fire behavior; and fire effects models that predict tree crown scorch height and tree mortality.

Solutions for most of these models can be obtained from nomograms (Albini 1976) and the BEHAVE system. BEHAVE is a set of programs for use on personal computers (Andrews 1986; Andrews and Chase 1989; Burgan and Rothermel 1984). More information about the BEHAVE system is contained in Chapter XII.C.1, this Guide.

4. Fire Spread Model. A fire spread model was developed by Rothermel in 1972 that allows managers trained in the use of the model to make quantitative estimates of fire behavior. The model is a mathematical representation of fire behavior in uniform wildland fuels. The fire spread model describes the processes that control the combustion rate: moisture evaporation, heat transfer into the fuel, and combustible gas evolution (Rothermel 1972).

a. Assumptions. Basic assumptions of the fire spread model are (Rothermel 1983):

(1) The fire is burning in a steady state in homogeneous surface fuels, not in crown or ground fuels.

(2) The percent slope and aspect are uniform.

(3) The wind is constant in both velocity and direction.

(4) The model describes fire behavior within the flaming front. The model does not describe behavior after the fire front has passed, such as during fuel burnout.

(5) The behavior of the fire is no longer influenced by the source of ignition or by suppression activities.

These assumptions are often violated when prescribed burning because ignition is often used to manipulate the fire. A common objective for burning is to consume fine fuels before the fire reaches a steady state. The predicted values do provide an estimate of fire behavior if a prescribed fire escapes.

b. Inputs to the Fire Spread Model. Required inputs to the fire spread model include fuel model, fuel moisture content, slope, and wind.

(1) Fuel model. A fuel model is a mathematical representation of the amount and kind of fuels present. The Fire Behavior Prediction System provides 13 standard fuel models that describe the characteristics of the portions of the fuel complex carrying the fire. Custom fuel models more closely describing a specific fuel situation also can be developed using the BEHAVE program (Burgan and Rothermel 1984). (See XII. C.1.a., this Guide.)

(a) Categories. In most situations, the flaming front of a fire advances through fine fuels such as grass, shrub foliage, litter, and small diameter down dead woody fuels. Wildland fuels can be grouped into four categories, according to the nature of the carrier fuels.

i. Grass or grass dominated: the primary carrier of the fire is grass.

ii. Shrub dominated: the primary carrier of the fire is either shrubs or litter beneath shrubs.

iii. Timber litter dominated: the primary carrier of the fire is litter beneath a timber (tree) stand.

iv. Logging slash: the primary carrier of the fire is residual material left from logging operations.

(b) Fuel properties. Fuel particles within a fuel complex have physical properties that influence the way they burn. The 13 standard fire behavior fuel models have specified physical properties (Anderson 1982). Properties can be changed to create a custom fuel model that may better describe a particular fuel complex. (See XII.C.1.a., this Guide.) Fuel properties that are the most important for determining the way a fire will behave include the following.

i. Fuel loading. The amount of live and dead fuel is expressed in weight per unit area. Loadings are grouped by particle size class and are usually expressed in tons per acre (kilograms per square meter). Total fuel is all plant material both living and dead present on a site. Available fuel is the amount of fuel that will burn under a specific set of fire conditions.

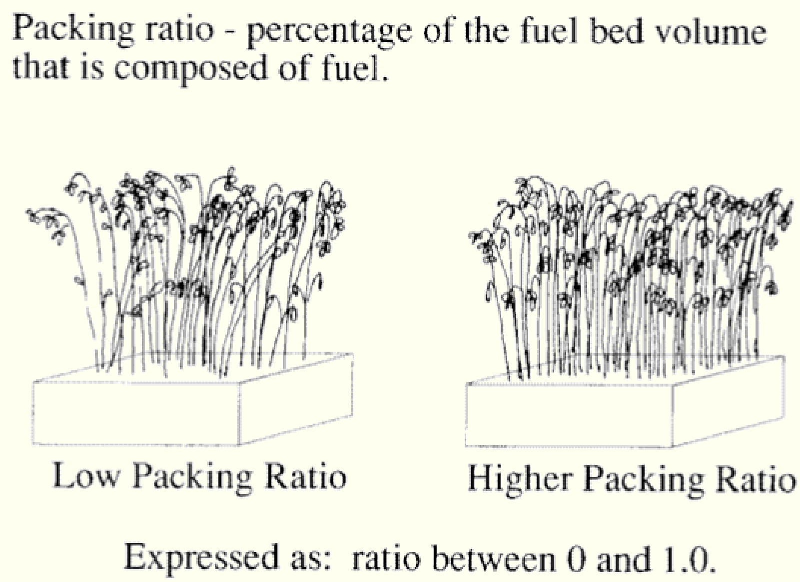
ii. Fuel size class. Dead fuels are divided into size classes based on diameter: less than 1/4-inch, 1/4 to 1-inch, 1 to 3 inches, and greater than 3 inches. (Metric equivalents of these size classes are: less than 0.6 centimeters, 0.6 to 2.5 centimeters, 2.5 to 7.6 centimeters, and greater than 7.6 centimeters.) Fuel size class is related to the rate at which particles wet and dry. This is discussed further in Chapter III.B.4., this Guide.

iii. Size class distribution. Fires usually start and spread in fine fuels, that is, those less than 1/4 inch in diameter. These fuels ignite increasingly larger size classes of fuels. If fine fuels or an intermediate size class are missing, a fire may not ignite or may not spread.

iv. Surface area to volume ratio. The surface area to volume ratio is a function of the particle size: the more finely divided the fuel material, the larger the ratio. Because small fuel particles have a large surface area compared to their volume, they dry out and ignite more rapidly than larger particles. Therefore, fine fuels usually have the most influence on fire behavior.

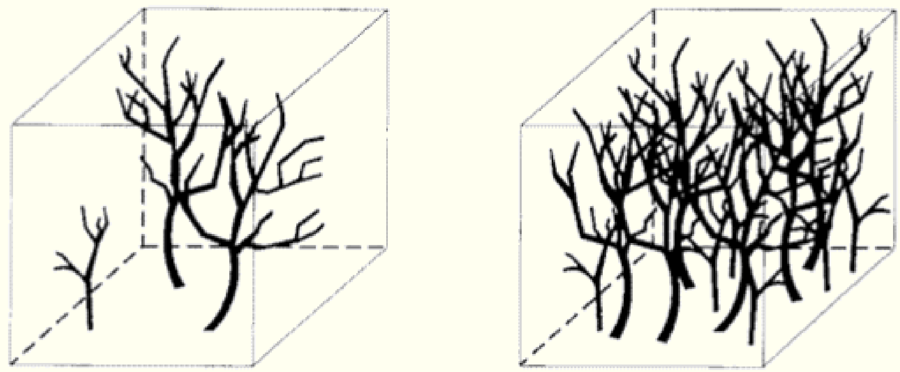
v. Fuel bed depth. Fuel bed depth is the depth of the surface fuel layer, i.e., the average height of surface fuels contained in the combustion zone of a spreading fire front.

vi. Packing ratio. The packing ratio is a measure of the compactness of the fuel bed. Expressed as a percentage, the packing ratio is the percentage of the fuel bed that is composed of fuel, the remainder being air space between the individual fuel particles (Burgan and Rothermel 1984). A fuel bed with no fuel has a packing ratio of zero, while a solid block of wood has a packing ratio of one (ibid.). A very open or porous fuel bed burns slowly because individual fuel particles are located so far apart that little heat is transferred among particles. A very compact fuel bed also burns slowly because airflow among the fuel particles is impeded, and there are large numbers of fuel particles that must be heated to ignition temperature. For every size of fuel particle, there is an optimum packing ratio at which heat transfer and oxygen produce the most efficient combustion (Burgan and Rothermel 1984). Compactness also influences the drying rate of fuel.



vii. Bulk density. Bulk density is the actual fuel weight per unit. It is calculated by dividing the weight per unit area by the fuel bed depth. It is a measure of the oven dry weight of fuel per cubic foot of the fuel bed, usually expressed as pounds per cubic foot. The higher the bulk density of the fuel, the slower the spread rate, because more fuel must be preheated to ignition temperature in order for the fire to spread.

The actual weight of the fuel within a volume of a fuel bed.



Expressed as: pounds per cubic foot (lbs./ft^3), or grams per cubic centimeter (g/cm^3).

viii. Fuel continuity. Fuel continuity is a description of the distribution of fuels. Fire spread is most likely in continuously distributed fuels. The greater the fuel discontinuity, the higher the fireline intensity required for fire spread. Fuel continuity is described in terms of both horizontal and vertical continuity. Horizontal continuity relates to the horizontal distances between fuel particles and relates to percent cover. The proximity of tree or shrub crowns affects the ease with which fire can spread in a live fuel strata. Vertical continuity describes the proximity of surface fuels to aerial fuels and affects the likelihood that a fire can move into the vegetative canopy.

ix. Heat content. The most important aspect of fuel chemistry influencing fire behavior is heat content. This value expresses the net amount of heat that would be given off if the material burns completely (or at 100 percent efficiency), rated as Btu per pound of fuel. The heat content for all species of dead woody fuel is essentially the same (Albini 1976). The presence of pitch in wood, and of volatile compounds such as oils and waxes in some live fuels, increases heat content, and thus flammability.

x. Live fuels. Some fuel types contain a significant component of live fuels in the surface fuel layer, including shrubs, grasses, and forbs. The importance of live fuels to fire behavior can change throughout the year. Their volume can increase significantly during greenup and the early part of the growing season. They can lose their foliage at the end of the growing season or during a drought. Seasonal fluctuations in moisture content occur that significantly affect flammability. The moisture cycles within live fuels are discussed in more detail in Chapter III.B.5., this Guide.

While technically live fuels, mosses and lichens do not have physiologically controlled seasonal moisture cycles. Their moisture content is very sensitive to changes in temperature and relative humidity and can become as low as that of surface litter layers. A dry surface layer of mosses and lichens can readily carry a fire in black spruce forests in Alaska (Dyrness and Norum 1983).

The volatile compounds in some species of live fuels allow them to burn at a higher moisture content than if there are few or no volatiles (Norum 1992). Sagebrush (*Artemisia* spp.) is considered to be a moderately volatile fuel, while chaparral shrubs, conifers, and dead juniper are highly volatile fuels (Wright and Bailey 1982).

Fire behavior in stands of shrubs containing volatile compounds can be extreme. This is attributed not only to their chemical content, but also to the high percentage of dead material that some of these stands of shrubs contain, and the ideal mixture of fuel to air within the shrub canopy (Burgan 1993).

(2) Fuel moisture. Fuel moisture content describes how wet or dry the fuels are. Moisture content is the single most important factor that determines how much of the total fuel is available for burning, and ultimately, how much is consumed. Fuel moisture determines if certain fuels will burn, how quickly and completely they will burn, and what phases of combustion the fuels will support. Fuels with a higher moisture content reduce the rate of energy release of a fire because moisture absorbs heat released during combustion, making less heat available to preheat fuel particles to ignition temperature (Burgan and Rothermel 1984). Ignition will not occur if the heat required to evaporate the moisture in the fuels is more than the amount available in the firebrand (Simard 1968). Environmental factors regulating dead fuel moisture content, and the relationship between fuel moisture content and fuel consumption, are discussed in III.B.4., this Guide.

(a) Fuel moisture formula. Fuel moisture content is the percent of the fuel weight represented by water, based on the dry weight of the fuel. In a word equation, it is:

Percent Moisture Content = Weight of Water / Oven-dry Weight of Fuel x 100

Moisture content can be greater than 100 percent because the water in a fuel particle may weigh considerably more than the dry fuel itself. For example, a green leaf may contain three times as much water as there is dry material, leading to a moisture content of 300 percent. Moisture content of duff and organic soil can be over 100 percent. Methods to measure and calculate fuel moisture content are described in Chapter III.D., this Guide.

(b) Moisture of extinction. The extinction moisture content is the level of fuel moisture at which a fire will not spread. It is a function of the fuel type and fuel bed geometry (Byram et al. 1966 in Albini 1976). The moisture of extinction is much lower for light, airy fuels such as fine grass, about 12 to 15 percent (Sneeuwjagt 1974 in Albini 1976), than it is for dense fuel beds such as pine needles, in which it has been measured at 25 to 30 percent (Rothermel and Anderson 1966 in Albini 1976). Under favorable burning conditions, the moisture of extinction has little effect on fire behavior, but when "conditions for burning are poor, it can cause significant changes in predicted fire behavior" (Rothermel 1983).

(3) Slope. The steepness of slope is measured as the rise of the ground in feet for every horizontal foot traversed, commonly referred to as "rise over run."

Percent Slope = Rise / Run x 100

Percent slope can be measured directly with instruments or calculated from topographic maps.

(4) Wind. Both windspeed and direction are used as inputs to the Fire Behavior Prediction System.

(a) Midflame windspeed. The speed of the wind is measured at the midpoint of the height of the flames because this best represents the wind that blows directly on the fire. Most weather forecasts, and most weather measurement stations, give the windspeed at 20 feet (6 meters) above the ground or above local obstructions. For fire behavior calculations, the 20-foot windspeed is reduced to the speed occurring at the midflame height. This compensates for the friction effect of vegetation and land surface that slows the speed of the wind. The adjustment factor varies with vegetation type, amount of canopy closure, and position on slope.

20 Foot Windspeed x Wind Adjustment Factor = Midflame Windspeed

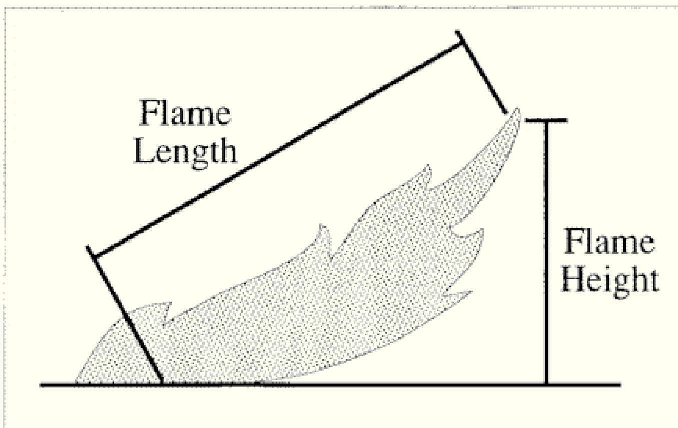
(b) Effective windspeed. As an intermediate step in obtaining solutions to the fire spread model, effective windspeed is determined. This value integrates the additive effects of slope steepness with a wind that is moving across or up a slope.

c. Outputs of the Fire Spread Model. The accuracy of predictions depends on how representative the fuel model chosen is of the fuels on the site, how accurately inputs are measured or estimated, and to what degree the situation meets the spread model assumptions. For predictions to be within a factor of two of actual fire behavior (from one-half to two times) is considered to be an acceptably accurate estimate (Norum 1993). The model is flexible enough that an experienced practitioner can make fairly good projections of fire behavior by carefully estimating or measuring the input values and tempering the results with judgment. Personal experience in a particular fuel type is necessary for refining output from this model.

(1) Forward rate of spread. One of the most important measures of fire behavior is the speed at which the fire moves across the landscape. The spread model calculates the rate of spread at the head of fire when the fire reaches its full, steady state speed. It predicts the speed of a fire burning in surface fuels, spreading on a single, unified front, that is not influenced by other ignitions. Rate of spread is generally stated in chains per hour, feet per minute, or meters per minute.

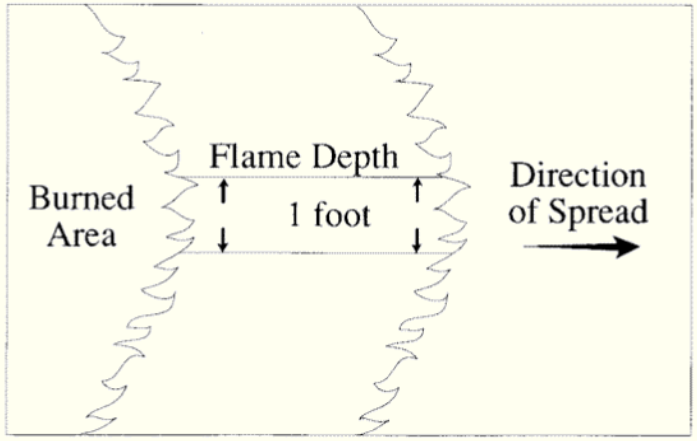
(2) Flame length. A second spread model output is the length of the flames when the fire has reached its full, forward rate of spread. Flame length is the distance along the slant of the flame from the midpoint of its base to its tip Flame height is the perpendicular distance from the ground to the flame tip and is not predicted by the fire spread model.

The average length of flame, measured along the slant of the flame from the midpoint of its base to its tip.



(3) Fireline intensity. Fireline intensity describes the nature of a fire in terms of its rate of energy release. Fireline intensity is the amount of heat given off by a fire along each foot of the leading edge of the fire each second, usually expressed as Btu per lineal foot of fireline per second.

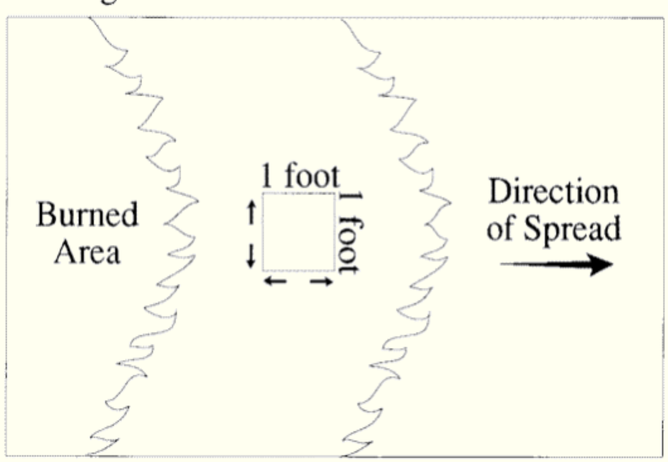
Rate of heat release in each **lineal foot** of the flaming front of a fire.



Btu's/lineal foot of fireline/second

(4) Heat per unit area. Another measure of the energy released from a fire is heat per unit area. It is the total amount of heat released in each square foot of the flaming fire front, usually expressed as Btu per square foot. All of the heat given off in the flaming front is included in this value, regardless of the length of time that the flaming front persists. For a given area with a specific amount and distribution of fuel, heat per unit area is inversely related to fuel moisture content. Heat released in flaming combustion that occurs as fuels burn out after the flaming front has passed is not included in the heat per unit area value.

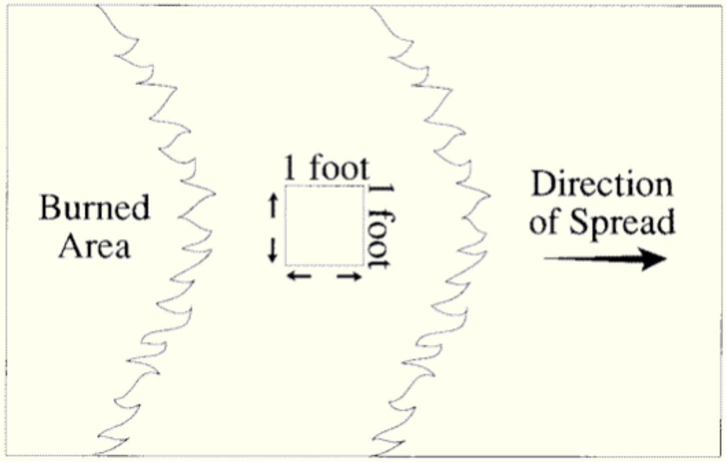
Rate of heat release in each **square foot** of the flaming front of a fire.



Btu's/square foot/minute

(5) Reaction intensity. Reaction intensity is a rate of heat release per unit area of flaming fuels, usually expressed in Btu per square foot per minute. This is the amount of energy released each minute by a square foot of flaming front, compared to heat per unit area which measures the total amount of energy given off per square foot. For a given fuel complex, reaction intensity can vary significantly with differences in moisture content.

Rate of heat release in each **square foot** of the flaming front of a fire.



Btu's/square foot/minute

d. Other predictable aspects of fire behavior.

(1) Probability of ignition. The probability of ignition, expressed as a percentage, is an estimate of the probability that a spark or firebrand landing on representative fuels will start a fire (Rothermel 1983). It is based on the amount of heat required to bring fine fuel to ignition temperature. Model inputs are fine fuel moisture, ambient air temperature, and the amount of shade.

(2) Maximum spotting distance. For many fuels situations, it is possible to make reasonably accurate estimates of the maximum distance to which fire may spot ahead by airborne embers (Rothermel 1983). The inputs required include the source of the embers, i.e. whether it is burning piles or trees; the species of tree, and their size and shape; the topography at and downwind from the fire; and the 20 foot windspeed. The model calculates the farthest distance a live ember is likely to be carried. It does not estimate how many burning embers will be lofted, or if the ember will ignite a spot fire. However, a combination of maximum spotting distance with the probability of ignition provides a workable idea of how far a fire may spot and the probability that it will cause a new fire.

(3) Crown fires.

(a) Classes. Van Wagner (1977) grouped crown fires into three classes based upon their dependence on the behavior of the surface fire.

i. Passive crown fires are those in which trees torch as individuals, ignited by the surface fire. These fires spread at essentially the same rate as surface fires. Trees torch within a few seconds with the entire crown enveloped in flames from its base to the top.

ii. Active crown fires are those in which a solid flame develops in the crowns. The surface and crown fires advance as a single unit dependent upon each other.

iii. Independent crown fires advance in the crowns alone, independently of the behavior of the surface fire.

(b) Crowning potential. The conditions necessary to cause the ignition of the crowns of trees or tall shrubs can be estimated. A probability of crown fires can be calculated, given the foliar moisture content and the height of the lowest part of the crowns. From these, an estimate can be derived of the fireline intensity needed to ignite the crowns (Rothermel 1983).

(c) Wind-driven vs. plume-dominated crown fires. The following discussion is taken from Rothermel (1991).

i. Wind-driven crown fire. A running crown fire can develop when winds blow flames from torching trees into adjacent tree crowns, or slope effectively accomplishes the same thing. Strong winds are the major force pushing the fire, and its spread rate can be greatly accelerated by slope. A strong convection column rapidly develops that is tipped over by the wind.

ii. Plume-dominated crown fire. A plume-dominated crown fire behaves quite differently from one driven by wind. Plume-dominated crown fires occur when windspeeds are fairly low. A strong convection column develops that rises above the fire, rather than leaning over before the wind. Air movement within the convection column generates the winds that cause significant rates of crown fire spread.

(d) Predicting size and intensity of crown fires. Rothermel (1991) presents methods for estimating and displaying the important elements of the behavior of a wind-driven crown fire. The model is applicable to coniferous forests of the northern Rocky Mountains, or forests with similar structure and fuels. Using these methods, an experienced fire behavior analyst can predict the rate of spread of a wind-driven crown fire, the length of flames, the time period when a particular crown fire will run, the probable area and perimeter of the crown fire, and the maximum rate of crown fire spread.

A method for calculating and comparing the power of a fire with the power of the wind is provided. The power of the fire is the heat energy released by combustion that drives the convection column, expressed as foot pounds per second per square foot. If the power generated by the fire is close to or exceeds that of the wind, a plume-dominated crown fire may develop. The onset of a plume-dominated fire may cause a sudden acceleration of the fire and faster spread rates than predicted. The model can thus predict the potential for onset of a plume-dominated fire, but not its behavior.

5. Relationships between Fire Behavior and Fire Effects. Few fire effects are known to be directly related to the behavior of the surface fire, that is to its spread rate, flame length, or rate of heat release. The following effects can be estimated from outputs of the Fire Behavior Prediction System.

a. Crown scorch height. There is a direct relationship between crown scorch height and flame length, ambient air temperature, and wind. All of these variables are used to measure the height above the surface of the ground that lethal temperatures occur. The height of tree crown scorch can be predicted from these values, using a model (SCORCH) in the Fire Behavior Prediction System. (See XII.D.1.a, this Guide.)

b. Tree mortality. A model (MORTALITY) estimates the percentage of tree mortality from scorch height, tree height, tree diameter, and crown ratio for eight species of conifers that occur in the northern Rocky Mountains. (See XII.D.1.b, this Guide.)

c. Total heat pulse to the site. Heat per unit area is a good estimate of the total heat pulse to the site when all of the fuel that is burned is consumed by the passing flame front. However, because this value does not account for long-term burnout of heavy fuels or organic soil layers, heat per unit area is not a very good estimate of the heat regime of the fire when much of the fuel, litter, or duff consumption occurs after the flaming front has passed.

6. Aspects of a Fire's Heat Regime that Cannot Presently Be Predicted. Many aspects of the heat regime of a fire cannot presently be predicted with any known model. Many of the most important and influential effects of fire on a site and its biological components are related to aspects of the heat regime of a fire that are not described by the Fire Behavior Prediction System.

Fireline intensity, as described in II.B.3.b.(3), is a rate of heat release that is related to flame length. However, the release of energy in flames has little relationship to the amount of subsurface heating (Hungerford 1989). Peak subsurface temperatures and the amount and duration of soil heating are not related to any value predicted by the Fire Behavior Prediction System. The effects of fire on fuels, soils, watershed, understory vegetation, and wildlife habitat cannot be estimated from measures of fireline intensity or flame length alone.

a. Fuel burnout time. This is the length of time that fuels continue to burn after the flaming front has passed, including all phases of combustion. The length of the fuel burnout period is related to fuel properties and fuel moisture but cannot be estimated by any known method.

b. Duration of smoldering and glowing combustion. Smoldering and glowing combustion are related to the amount of fuel, its size class distribution, thickness of duff and organic layers, and the moisture content of heavier fuels and duff. We cannot presently predict the duration of time during which these combustion phases will occur.

c. Total heat pulse to the site. Total heat pulse considers not only the heat released in flames but also that released by smoldering and glowing combustion. Heat per unit area only includes the amount of heat that is released in the flaming front. Extensive studies in physics modelling is currently underway at the Intermountain Fire Sciences Lab which may provide means to calculate the total heat pulse to the site.

d. Soil heating. Most heat produced by the flaming front moves upward. Downward movement of heat from flames cannot presently be predicted, but it is not believed to be a significant source of subsurface heat. Most soil heating results from long term fuel, duff, and organic layer burnout. Neither this heat, nor its penetration into soil layers, has been modelled.

e. Burn severity. Burn severity is a term that qualitatively describes classes of surface fuel and duff consumption. Large diameter down, dead woody fuels and organic soil horizons are consumed during long-term, smoldering and glowing combustion. The amount of duff or organic layer reduction is also called depth of burn, or ground char (Ryan and Noste 1985). Because the amount and duration of subsurface heating can be inferred from burn severity, this variable can be related to fire effects on plants and soils. Factors regulating fuel and duff consumption, and thus burn severity, are discussed in Chapter III.B.2. and 3. The relationship between burn severity and its effects on plants is described in Chapter VI.B.1.c. and VI.B.2.c.

(1) Descriptive classes. An example of a set of burn severity classes is given below. Agency specific guidelines for assessing burn severity are described in USDI-NPS (1992).

(a) Unburned.

(b) Scorched. Foliage is yellow; litter and surface vegetation are barely burned or singed.

(c) Low severity. Small diameter woody debris is consumed; some small twigs may remain. Leaf litter may be charred or consumed, and the surface of the duff may be charred. Original forms of surface materials, such as needle litter or lichens may be visible; essentially no soil heating occurs.

(d) Moderate severity. Foliage, twigs, and the litter layer are consumed. The duff layer, rotten wood, and larger diameter woody debris is partially consumed; logs may be deeply charred; shallow ash layer and burned roots and rhizomes are present. Some heating of mineral soil may occur if the soil organic layer was thin.

(e) High severity. Deep ash layer is present; all or most organic matter is removed; essentially all plant parts in the duff layer are consumed. Soil heating may be significant where large diameter fuels or duff layers were consumed. The top layer of mineral soil may be changed in color; the layer below may be blackened from charring of organic matter in the soil.

(2) Relationship between fireline intensity and burn severity. There can be many combinations of fireline intensity and burn severity on any site, depending on fuel loading and distribution, and site weather and moisture conditions at the time of the fire. For example, given a site with good, continuous surface fuels, and a deep litter/organic layer, any of the following combinations of fireline intensity and burn severity can occur (as well as a lot of intermediate combinations).

(a) High fireline intensity/high burn severity. Both the carrier fuels and organic layer are dry. The result is a fire with high fireline intensity that exhibits vigorous fire behavior, that is also a deep burning, high severity fire. Flames are long, large fuels are removed, soil organic layers are consumed, and the long duration fire causes a significant amount of subsurface heating.

(b) High fireline intensity/low burn severity. The carrier fuels are dry, but the litter/duff layer is wet. The result is a fire with high fireline intensity, that exhibits vigorous fire behavior, but which is a very low severity fire because the organic layer is too wet to burn. Flames are long but little subsurface heating occurs.

(c) Low fireline intensity/high burn severity. The carrier fuels and surface litter are moist, and litter/duff layers are dry. The result is a fire of low fireline intensity that may barely cover the area, but of high burn severity wherever the litter/duff layer ignites because it is dry enough to burn. Even though the surface fire was of little apparent consequence, a significant amount of soil heating can occur, caused by the consumption of dry duff layers, peat, and/or large diameter downed woody fuels.

(d) Low fireline intensity/low burn severity. The carrier fuels are moist and the litter/duff layer is wet. The result is a fire with low fireline intensity that also has low severity.

(3) Application to shrub dominated communities. Burn severity concepts can also apply to litter and duff layers beneath isolated trees and shrubs.

(1) Low severity fire may just scorch the litter beneath the shrub or tree crown.

(2) A moderate severity fire may consume some basal litter and organic matter, but residual material remains. Some heating of deeper organic layers and soil may occur.

(3) A high severity fire removes all litter and duff, leaving only an ash layer. Significant amounts of soil heating can only occur where there is a high degree of consumption of thick, basal organic layers. Isolated patches of severely burned ground may occur where shrubs used to be, surrounded by extensive areas where little soil heating occurred.

f. Burn pattern. The pattern of a fire is the mosaic of burned and unburned vegetation and fuels. It can be further defined in terms of the degree of heating and consumption of fuels and vegetation, such as scorched compared to severely burned areas. A pattern can occur in the tree canopy, shrub canopy, in surface fuels, or in litter, duff, and organic layers. The size of the mosaic can vary from acres of scorched, consumed, and unburned patches in the canopy, to mosaic patterns of burned and unburned fuels and litter layers of only a few feet, or even inches. The effects of fire are closely related to the pattern of the fire, both on a large and small scale. Fire effects vary considerably with burn pattern because it reflects the variation in the fire's heat regime, above, at, and below the surface.

Significant variations in burn pattern are the result of differences in fuel continuity, fuel loading, fuel moisture, aspect, wind, and ignition methods and techniques. Whether a fire will become a surface or crown fire, and its effects on fuel consumption and soil heating, can be estimated by a person skilled in fire behavior or prescribed fire. However, there are presently no computational tools with which to predict the exact burn pattern that will occur.

C. Resource Management Considerations

1. Levels of Fireline Intensity. Different levels of fireline intensity, along with corresponding flame lengths, have special meaning both for the design of prescriptions for prescribed fire, and in wildfire management activities. From widely held and commonly agreed upon experience, the following are reliable rules (Rothermel 1983).

a. When fireline intensity is below 100 Btu per foot of fireline per second, flame lengths are less than 4 feet (1.2 meters).

(1) Fires can generally be attacked at the head or flanks of the fire by persons using hand tools.

(2) Handlines should be adequate to hold the fire.

b. Fireline intensity 100 to 500 Btu per foot of fireline per second; flame lengths are between 4 and 8 feet (1.2 to 2.4 meters).

(1) Fires are too intense for direct attack at the head of the fire by persons using hand tools.

(2) Handline cannot be relied upon to hold the fire.

(3) Equipment such as bulldozers, pumpers, and retardant aircraft may still be effective.

(4) Fires are potentially dangerous to personnel and equipment.

c. Fireline intensity 500 to 1,000 Btu per foot of fireline per second; flame lengths are between 8 and 11 feet (2.4 to 3.4 meters).

(1) Fires may present serious control problems, such as torching out, crowning, and spotting ahead.

(2) Control efforts at the head of the fire probably will be ineffective. Indirect attack is probably the only means of suppression.

(3) Fires are definitely dangerous to personnel and equipment.

d. Fireline intensity above 1,000 Btu per foot of fireline per second; flame lengths are greater than 11 feet (3.4 meters).

(1) Crowning, spotting, and major fire runs are probable.

(2) Control efforts at the head of the fire are ineffective by any known means of suppression. Indirect attack and tactical counterfiring may be the only means to slow the spread of the fire in certain directions.

(3) Fires are extremely dangerous to personnel and equipment in the immediate vicinity of the fire.

These values have obvious implications for holding actions on prescribed fires and suppression actions on wildfires. If only hand crews are available to hold a prescribed fire, and handlines are the only lines of control, then prescription variables (inputs to the spread model) should be set so that surface fires do not exceed 100 Btu per second per foot, nor flame lengths exceed 4 feet (1.2 meters).

2. Relationship between Moisture Content of Big Sagebrush and Fire Behavior. In vegetation types dominated by shrubs, moisture content of foliage can be a dominant factor in the behavior of wildland fires. Within a given geographical area, it is possible to determine threshold levels of foliar moisture content that relate to degrees of fire behavior activity and difficulty of control. In order to obtain such a database, foliar moisture content levels must be documented in areas where fires are occurring and fire behavior observations must be recorded. Sampling at established intervals over a period of several years and relating moisture levels to easily identifiable growth stages of the plants would provide the most useful information.

Threshold levels of moisture that relate to fire behavior in sagebrush have been determined for Nevada and eastern Oregon.

a. Nevada. When Greg Zschaechner worked for the Bureau of Land Management in Nevada on the Great Basin Live Fuel Moisture Project, he established guidelines that relate the moisture content of big sagebrush *(Artemisia tridentata)* to fire behavior and effective suppression tactics. Suppression tactics are included in the following descriptions because they provide additional description of the behavior of the fire. These levels are most accurate within Nevada but may serve as general guidelines elsewhere.

(1) 181 percent and above. Fires will exhibit VERY LOW FIRE BEHAVIOR with difficulty in continued burning. Residual fine fuels from the previous year may carry the fire. Foliage will remain on the stems following a burn. Fires can generally be attacked at head or flanks by persons using handtools. Handlines should hold the fire without any problems. Fires will normally go out when the wind dies down.

(2) 151 percent to 180 percent. Fires will exhibit LOW FIRE BEHAVIOR with fire beginning to be carried in the live fuels. Both foliage and stem material up to 1/4 inch (0.6 centimeters) in diameter will be consumed by the fire. Burns will be generally patchy with many unburned islands. Engines may be necessary to catch fires at the head. Handline will be more difficult to construct but should hold at the head and flanks of the fire.

(3) 126 percent to 150 percent. Fires will exhibit MODERATE FIRE BEHAVIOR with a fast continuous rate of spread that will consume stem material up to 2 inches (5.1 centimeters) in diameter. These fires may be attacked at the head with engines but may require support of dozers and retardant aircraft. Handline will become ineffective at the fire head but should still hold the flanks. Under high winds and low humidities, indirect line should be given consideration.

(4) 101 percent to 125 percent. Fires will exhibit HIGH FIRE BEHAVIOR leaving no material unburned. Head attack with engines and dozers will be nearly impossible on large fires, but may still be possible on smaller, developing fires. Flanking attack by engines and indirect attack ahead of the fire must be used. Spotting should be anticipated. Fires will begin to burn through the night, calming down several hours before sunrise.

(5) 75 percent to 100 percent. Fires will exhibit EXTREME FIRE BEHAVIOR. Extreme spread rates and moderate to long range spotting will occur. Engines and dozers may be best used to back up firing operations and to protect structures. Indirect attack must be used to control these fires. Fires will burn actively through the night.

(6) 74 percent and below. Fires will have ADVANCED FIRE BEHAVIOR with high potential to control their environment. Large acreage will be consumed in very short time periods. Backfiring from indirect line such as roads must be considered. Aircraft will need to be cautious of hazardous turbulence around the fire.

b. Eastern Oregon. Fire behavior and its relationship to moisture content in sagebrush has been monitored on Oregon rangelands east of the Cascades (Clark 1989). The following moisture levels indicate how readily a fire can propagate, given that adequate fine fuels are present between sagebrush plants to carry the fire, or that sagebrush density is high enough for flames to reach between plants where herbaceous fuels are sparse.

(1) Above 90 percent. Fire behavior is docile. The fire may or may not spread and is easy to control.

(2) 60 to 90 percent. Fire is much more difficult to control. Fire is likely to burn actively throughout the night, especially if wind is present.

(3) Less than 60 percent. The fire displays extreme fire behavior and rates of spread, and is essentially uncontrollable by normal suppression methods.

3. Effect of Fuel Type Changes on Fire Behavior. The dominating factor regulating fire behavior is wind in some fuel types and moisture content in others. The behavior of fires in fuel types with a large component of fine materials, such as the grass models, is most influenced by wind. Fuel moisture is much more important than wind in regulating the activity of fires in fuel types with a lot of larger diameter, dead woody fuels. Wind, while influential, is not so dramatically important in heavy fuels as it is in grass or shrub type fuels.

Fire behavior can drastically change when a fire moves into a different fuel type. If a fire moves from an area of logging residue to one dominated by cured grassy fuels, flame length and fireline intensity will probably decrease, but the rate of spread is likely to increase significantly. An optimal prescription for burning the logged unit to reduce hazard fuels would include low moisture content in smaller size classes of fuels and low windspeeds. Under these conditions, the desired amount of consumption in the harvested area would be achieved, and any escape into the grass fuels outside of the unit could easily be caught.

4. Effect of Long-Term Drying on Heat Release. Long periods of limited precipitation result in deep drying of the surface organic layers. Deeper drying of the entire fuel complex leads to an increase in fire behavior because of greater involvement of larger fuels and surface organic fuels in the fire front. Because more fuels burn in the initial stages of flaming combustion, fireline intensities and flame lengths can be greater. More heat can also be generated during smoldering and glowing phases of combustion, as deeper organic layers may burn. Fire effects may be much more notable than if the site had burned under less droughty conditions.

5. Burn Pattern. When igniting a prescribed fire, the pattern of burn that is being obtained should be noted throughout the ignition period. If the desired mosaic is not being obtained, alteration of ignition pattern may change the percent of the prescribed fire area that is actually being covered by fire.

6. Firewhirls. Firewhirls are tight, spinning vortices filled with flame and hot gas that have the appearance of a small tornado of fire. They can cause severe difficulty in controlling a wildfire or prescribed fire by spreading pieces of flaming material great distances beyond the project area.

When igniting a prescribed fire using strip headfires, it is important to let one strip of fire burn down in intensity before igniting the next strip. This avoids concentrated mutual convection, competition for incoming air, and a high probability of initiating firewhirls at the ends of the strips. Also, by skillfully designing the ignition pattern and sequence, the risk of firewhirls developing on lee slopes, and where two fires merge, can be minimized.

D. Methods to Monitor Fire Behavior and Characteristics

Fire prescriptions contain elements that define ranges of acceptable weather and moisture conditions that produce the desired fire behavior and characteristics. Monitoring for a prescribed fire can include monitoring of weather and fuel moisture before the fire to determine the daily weather patterns in a particular area and to determine how close moisture conditions are to the prescribed range. Some factors vary diurnally, such as temperature, relative humidity, and the associated moisture content of small diameter fuels. Other prescription elements, such as moisture content of soil organic layers or live fuel moisture content, decrease slowly, and weekly monitoring is often adequate to detect change.

It is important to monitor all elements of the prescription during a fire to determine that the fire remains within prescription, that the fire behavior predictions were adequate, and to correlate with the subsequent effects of the fire treatment. Whenever possible, information on fuel moisture, fire behavior, and fire characteristics should be obtained in the same location(s) as fire effects data collection occurs. In order to most effectively monitor rates of fire spread, flame length, and burn severity, some equipment may need to be installed before a fire.

If site characteristics vary on the burned area, specific site attributes should be documented as observations about fire behavior are made. Fuel type, vegetation type, slope, and aspect should be recorded, as well as a notation about the location where the observation is made. Whether the fire is a heading, backing, or flanking fire should be noted at the same time as observations are made.

1. Burning Conditions.

a. Fuel moisture. Fuel moisture is a critical determinant of fire behavior and characteristics. Techniques to monitor fuel moisture are described in Chapter III.D.4, this Guide.

b. Weather. An important part of monitoring fire behavior and characteristics is to have a good record of weather that occurred during the time that a fire occurs. A standard set of weather observations should be taken at regular intervals during the fire: temperature, relative humidity, windspeed and direction, clouds or other indicators of instability, and the presence of thunderstorms. Standard procedures for monitoring weather are detailed in Finklin and Fischer (1990). Agency specific guidance on weather data collection is available in USDI-NPS (1992).

2. Fire Behavior and Characteristics. Agency specific guidelines for monitoring fire behavior and characteristics are described for forests and for grassland and brush types in USDI-NPS (1992). The following description provides additional methods.

a. Rate of spread. Observations of rate of fire spread should only be taken after the fire has reached a steady state, because this is what the fire behavior system predicts. Rate of spread measurements are difficult to document on prescribed fires with center or perimeter firing patterns, or narrow strip headfires. In these situations the fire often has not reached a steady state, or its behavior is influenced by the ignitions that have occurred in adjacent areas. Whether the fire is heading, backing, or flanking at the point of observation should be noted. The following discussion is taken largely from Zimmerman (1988).

(1) Visual observation. Visual observation and pacing of distances can be used to take rate of spread measurements, particularly on a slowly moving fire. A stopwatch is used to determine how long it takes a fire to cover a specific distance. Rate of spread is calculated from the time/distance relationship.

(2) Metal tags. Numbered metal tags can be thrown at or near the flaming front. A stopwatch is started when the front crosses a tag and a second tag is thrown ahead of the fire. When the flaming front crosses the second tag, the stopwatch records the elapsed time. The distance between the two tags is measured by pacing or steel tape, and the spread rate calculated.

(3) Grid marking system. When high spread rates are expected, and/or when it is not safe to be immediately adjacent to the fire, fire behavior can be measured using a grid system installed before the fire. Spacing of markers should be related to the expected rate of forward spread of the fire. Reference point markers can consist of materials such as flagging tape tied to branches or poles painted with bright paint. Times are recorded with a stopwatch or wrist watch as the fire burns past each marker, and rate of spread is determined.

(4) Sketch map. Sketch maps of the fire perimeter can be made at different times during the period of fire growth, a useful technique if reference points are plentiful or the fire will cover a large area. This method requires a good vantage point or the use of an aircraft. Rate of spread can later be calculated by dividing the distances between different landmarks by the time periods it took to cover these distances.

(5) Photography. Pictures of the fire can be taken at specific intervals and time noted when each photo is taken. A 35 mm camera with split lens can be used (Britton et al. 1977), in which one side of the image is focused on a watch, and the other on the flame. Cameras are now commercially available that record date and time on each image. The use of black and white infrared film greatly increases the value of this technique because it increases the quality of an image recorded through visually obscuring smoke.

(6) Video camera. Video cameras can be very successfully used when monitoring fire behavior. Time and other observations can be recorded on an audio track while recording the visual image. The advantages of video cameras include the potential for making a complete record of fire as it burns in specific areas, the fact that image quality can be immediately assessed, and that cameras are relatively inexpensive and very portable.

A computerized image analysis system has been used to study video tapes. A grid representing a known size or distance is set on the first frame, and subsequent measurements can be made from the screen image (McMahon et al. 1987).

b. Flaming residence time. Residence time is the amount of time that it takes the flaming front of the fire to pass a particular point. Residence time can be difficult to measure because of the indefinite trailing edge of the fire as concentrations of fuel continue to flame. It can be estimated from observations, still photography, or a video camera. The video position analyzer system (ibid.) also can be used to obtain more accurate residence time estimates. Use of infrared sensors or film are extremely useful when smoke obscures flames.

c. Fuel burnout time. Residence time discussed above is only a measure for flaming combustion. For monitoring that will later be related to fire effects, an estimate of the total duration of smoldering combustion of large diameter fuels and duff layers is important. Observation, repeated photography of the same points particularly with black and white infrared film, repeated video camera images, or a Probe-eye® can record fuel burnout time. Infrared images can sense higher temperatures caused by continued smoldering or glowing combustion when no visible signs of combustion are present. While it is not important to document the duration of long-term combustion to the exact minute, it is important to note whether smoldering combustion lasts for only a few minutes, or a few hours, or several days.

d. Flame length. Flame length is measured along the slant of the flame. The accuracy of estimation of flame length can be increased by installing reference points that provide scale. Steel posts with 1-foot sections alternately painted red and white or metal flags attached every 3 feet (the choice depends on the expected scale of the flames) set in the burn area work very well (Rothermel and Deeming 1980). These markers can be the same as those used to measure rate of fire spread.

(1) Observations. Flame length data are usually obtained from visual observations of average flame length at set intervals. Flame length is usually recorded at the same time as rate of spread observations are made.

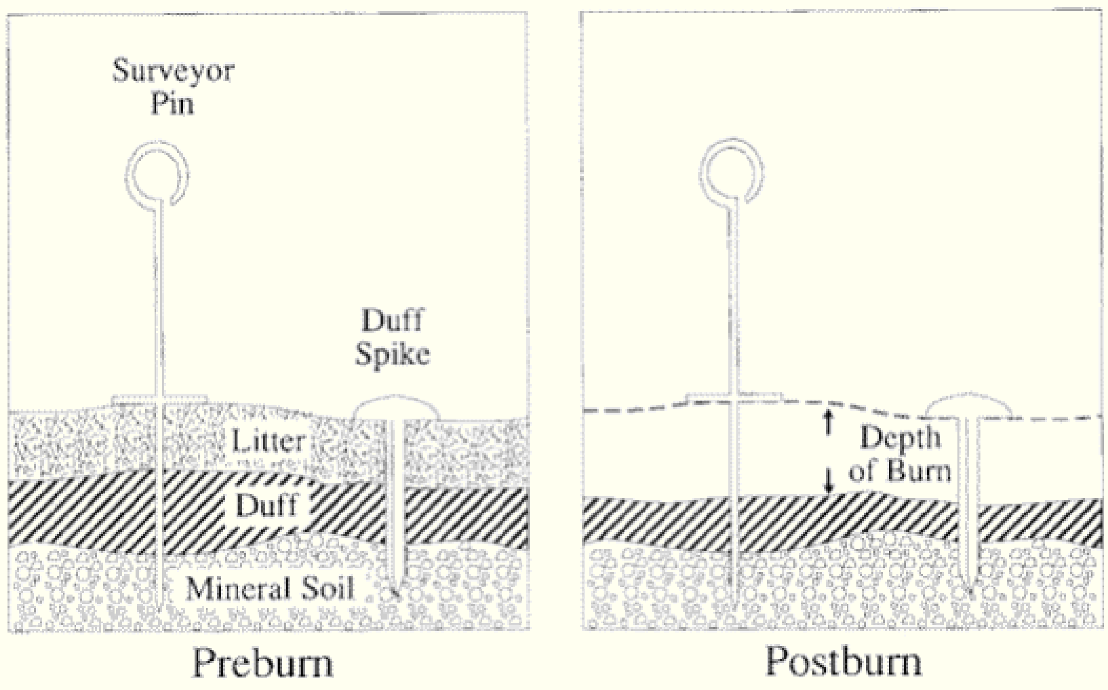
(2) Photography. Flame length can be documented with cameras and time and location of observation of each exposure recorded. Accuracy is enhanced by use of infrared film.

(3) Video camera. Not only are video cameras an excellent way of documenting fire behavior, the passive image analyzer mentioned above (McMahon et al. 1987) allows a very accurate measurement of flame length. After a grid of known size is established on the first frame, the tape is advanced until a representative flame is seen on the screen. The image is frozen on the screen, and the flame is outlined on the screen with a cursor. Computer software then calculates flame length.

e. Burn pattern. A map of the burned area can be made at both a gross and detailed scale. For general monitoring purposes, a map of the burned area can show areas where the tree or shrub canopy was removed, areas where the fire was an underburn, and areas the fire did not burn at all. Information on burn pattern can be obtained by a walk through the burned area, by long transects, or with photography. A low elevation aerial photo, or an oblique photo taken from a high vantage point such as a hill or a tree, can be measured with a dot grid to determine burn pattern. For large wildfires, satellite imagery can be used to obtain information on the pattern of burned and unburned areas, and where the fire was a surface fire or a crown fire. When choosing imagery for analysis, it must be remembered that up to about 2 weeks may pass before scorch damage to overstory tree foliage is apparent.

f. Burn severity/Depth of burn. The pattern of burn severity on the surface of the ground can be quite complex, because it varies with the distribution of prefire fuel loading and arrangement, thickness of litter and duff layers, and moisture content of surface and ground fuels. While mapping the pattern of burn in the surface fuels and vegetation for an entire burned area may be too large a task, burn severity, and the degree of canopy removal, should be noted in the areas where fire effects monitoring sites are located.

Surveyor pins or bridge spikes can be used in easy and practical way to monitor depth of burn. The pins or spikes are pounded into the ground before the fire, with a cross piece or top of the spike level with the top of the litter layer. After the fire, the amount of pin exposed is a measure of the depth of organic material removed. The amount of residual organic layer at each pin site can be measured to obtain an estimate of duff removal. Use of an inexpensive metal detector can make it much easier to relocate metal pins after the fire.



3. Potential Control Problems. The occurrence of any of the following during a prescribed fire should be noted and recorded and the Burn Boss or Fire Behavior Analyst notified.

a. Spotting. If spotting is occurring outside the burn perimeter, record the time of occurrence, distance from the fire front, and location on a map.

b. Torching or crowning. Torching or crowning trees may produce spots and may indicate that the situation requires extra caution. Note the time and location of occurrence and any relationship observed with surface fire behavior.

c. Firewhirls. The location and time of any firewhirls should be recorded. Observations about the fuels in the area of the fire whirls, or any relation to ignition method or technique, should be noted.

d. Fire behavior exceeding prescription limits. Any observation of rate of spread or flame length that exceeds specified limits could provide a potential control problem. Fire behavior less than that predicted is not necessarily a control problem, but can lead to an improper site treatment, and should be reported.

E. Summary

Knowledge of the behavior and characteristics of wildland fire are important both for managing fire and for understanding and interpreting the effects of fire. The heat regime created by a fire varies with the amount, arrangement, and moisture content of flammable materials on a site. Trained and experienced people can predict (within a factor of two) some aspects of the behavior and heat release of a flaming front of a fire, and some associated fire effects such as crown scorch. However, many fire effects are related to characteristics of fire that are not related to the behavior of the flaming front and cannot presently be forecast.

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CHAPTER III - FUELS

By Melanie Miller

A. Introduction

Fuel characteristics strongly influence how a given site will burn under specific environmental conditions. Fire also has an effect on fuels, because fire requires the consumption of fuel. Besides removing fuel, fire can result in the creation of additional surface fuel, as vegetation drops fire-killed leaves, needles, and twigs, and dead shrubs and trees fall and become part of the surface fuel layer.

All effects of fire on resources result from its effect on fuels, because the way that fuels burn determines the heat regime of a fire. Each fire varies in the amount of fuels that burn, the size class distribution of fuels that burn, the rate at which fuels burn, how much soil organic matter burns, and whether living plants become fuel. The nature of fuel consumption determines the peak temperatures reached, the duration of heat, and the stratification of heat above and below the surface.

Fuels such as litter, snags, and downed trees, have important effects on a site. Freshly deposited litter protects the soil surface from erosion by raindrops. Unburned logs and fallen fire-killed trees provide locations for mycorrhizae, nitrogen fixation, and habitat for birds, mammals, and insects. Standing snags provide habitat for many animal species that utilize this specific habitat.

This chapter will discuss the factors that regulate the effect of fire on fuels. Different types of fuels are defined, and the factors that control the amount of fuel and organic layer consumption are described. Dead and live fuel moisture content are discussed in great detail because moisture content is the most important determinant of the combustion process and the heat regime of the fire. Basic principles of fuel succession, the changes in the fuel complex over time, are summarized. Those properties of fuels that affect the behavior of a flaming fire are described in Chapter II of this Guide, Fire Behavior and Characteristics. Those aspects of combustion that affect smoke production are discussed in Chapter IV, Air Quality, this Guide. The role of downed logs and organic matter in regulating soil nutrients and their relationship to fire are discussed in the Soils chapter of this Guide, Chapter V. Wildlife use of dead woody material as habitat is described in Chapter VII, Wildlife Habitat.

B. Principles and Processes

1. Fuel Classification. Fuel is all vegetative biomass, living or dead, that can be ignited by lightning or an approaching fire front. Wildland fuels have been grouped into various classes.

a. Natural fuels vs. activity fuels. Natural fuels result from plant growth and death, loss of foliage, branch breakage, and tree blowdown. Activity fuels are similar to natural fuels but they are distributed differently in time and space due to human activity such as logging, thinning, chaining, and herbicide use.

b. Down, dead woody fuels. This class of fuels includes dead twigs, branches, stems, and boles of trees and shrubs that have fallen and lie on or above the ground (Brown et al. 1982). Wood can be either sound or rotten. Sound wood is essentially intact. It may have checks or cracks, but it still retains its structure. Rotten wood is partially decomposed. Material is punky or can be easily kicked apart. It can be important to distinguish between sound and rotten large diameter woody fuels because their moisture retention and combustion characteristics are very different. (See B.2.a.(3) and B.4.d.(3), this Chapter.)

c. Soil litter and organic layers.

(1) Litter. Litter is the top layer of the forest floor, typically composed of loose debris such as fine twigs, and recently fallen leaves or needles, little altered in structure by decomposition. Litter can also include loose accumulations of debris fallen from rangeland shrubs, and dead parts of grass plants lying on or near the surface of the ground. Some surface feather moss and lichen layers are also considered to be litter because their moisture response is similar to dead fine fuel.

(2) Organic layers.

(a) Duff. Duff is the partially decomposed organic material of the forest floor that lies beneath the freshly fallen twigs, needles and leaves. It is equivalent to the fermentation and humus layers of soil.

(b) Organic soils. Soils that are essentially composed of deep layers of organic matter form wherever production of organic matter exceeds rates of decomposition. They frequently develop in poorly drained areas where plant material partially decomposes in water or in saturated environments. Organic soils can be extensive in wetlands and in cool, moist climates (Buol et al. 1973). The amount of incorporated mineral material can vary significantly. The organic content of these soils can burn if soil moisture content is low enough.

d. Live fuels. Live fuels are living vascular plants that may burn in a wildland fire. Live fuels include trees, shrubs, grasses and grass-like plants, forbs, and ferns. Because of seasonal variation in their moisture content, the flammability of their foliage can vary significantly. Herbaceous plants, i.e., grasses and forbs, can cure, changing from a live fuel to a dead fuel. The leaves and older needles of trees and shrubs dry and fall from the plants, adding to the surface litter layer at the end of the growing season. Living plants may contain a large component of dead material, such as dead branchwood in older shrubs. This increases their flammability and the likelihood that the entire plant will be consumed by fire. Live fuel moisture cycles are described in greater detail in section B.5. of this chapter.

e. Fuel strata. Fuels have been classed as surface, ground, aerial, or ladder fuels. These terms are described in Chapter II.B.1.a., this Guide.

f. Total fuel vs. available fuel. Total fuel is the total amount of fuel present on a site. Available fuel is the amount that can burn, under a given set of conditions. The amount of available fuel depends on fuel size, arrangement, and moisture content (Brown and See 1981). Fuel size can affect availability if there are inadequate amounts of smaller sized fuels to burn and transfer enough heat to larger size fuels to raise them to ignition temperature. Standing tree boles are not considered to be available fuel because they are extremely unlikely to burn in wildland fires, except for smoldering in punky snags.

There can be long-term changes in fuel availability. The total amount of fuel within a stand increases as plants grow, while the distribution of fuel within a stand changes when snags or branches fall or foliage drops. There can also be short term changes in availability due to fuel moisture content. At any given point in time, the most important factor affecting fuel availability is its moisture content, because this determines whether fuel can ignite, and whether it can sustain combustion.

2. Factors Regulating Dead Fuel Consumption.

a. Relationship to physical and chemical properties of fuel. The key fuel properties that affect fire behavior were described in Chapter II.B.3.b. The following discussion explains how some of these factors influence fuel consumption.

(1) Fuel size. Fuels less than 1/4-inch diameter are almost completely consumed by fire over a wide range of burning conditions. Most branchwood between 1/4-inch and 3 inches is also consumed (Martin et al. 1979). A fair prediction of the consumption of large diameter dead woody fuels is possible if the average preburn diameter is known (Reinhardt et al. 1991).

(2) Fuel continuity. Fuel continuity relates to the proximity of individual pieces of fuel and also of different fuel strata. It affects fire spread, how much of an area ignites, and how much fuel is consumed. Breaks in fuel continuity contribute to patchy fire spread and may result in patchy fuel consumption.

(a) Forests. Large diameter fuels in local accumulations are more likely to be consumed than if these fuels are scattered. Anderson (1983) found that large downed woody fuels need to be within a distance of about 1.5 diameters of each other for interactive burning to occur.

(b) Rangelands/grasslands. Fuel consumption in range and grass types can be closely related to fuel continuity. If fuels are sparse, windspeed may not be adequate to spread the fire, limiting the amount of fuels that are ignited and consumed.

(3) Quality. Wood may be sound, rotten, or partially rotten. In north Idaho, consumption of rotten wood was greater than that of sound wood, even though moisture content of rotten wood was higher. Consumption of completely rotten pieces was higher than that of partially rotten pieces (Reinhardt et al. 1991).

(4) Heat content. The heat content of wood is about 8,000 Btu per pound (Albini 1976). Pitch adds to the flammability of wood because its heat content is about 15,000 Btu per pound (Carmen (1950 in Byram 1959). Pitchy fuels can burn at a much higher moisture content than those without pitch. A damp pitchy stump that is ignited is often completely consumed by fire.

(5) Fuel moisture content. The major effect of moisture on small fuel consumption is simply whether fuels are dry enough to ignite. Eighty to 90 percent of fine woody forest fuels are consumed wherever fire spreads (Brown et al. 1985). This is also true for fine rangeland fuels. The proportion of large diameter woody fuels consumed is more strongly influenced by their moisture content than by any other factor (Reinhardt et al. 1991).

b. Relationship to phase of combustion. The phase of combustion during which dead woody fuels are consumed is related to their size.

(1) Small diameter fuels. Fine fuels tend to be consumed during flaming combustion. However, the arrangement of small woody fuels sometimes does not provide enough mutual heating during the flaming state for complete consumption to occur. Blackened branches may burn off or fall into the ash and generate enough mutual heat for more flaming combustion to occur. Eventually the amount of heat that is generated decreases and can no longer support flaming, and the remaining consumption of these small pieces occurs by smoldering and glowing combustion (Norum 1992).

(2) Large diameter fuels. While the surface layer of woody fuels may initially support flames, most of the consumption of large woody fuels, both sound and rotten, occurs in the smoldering and glowing phases of combustion. Glowing combustion in large woody fuels commonly lasts for 10 to 20 times longer than the flaming phase (Anderson 1983).

3. Factors Regulating Consumption of Duff and Organic Soils.

a. Moisture content. As is true for large diameter woody fuels, moisture content is the most important variable influencing consumption of duff and soil organic layers. Duff and soil temperatures remain below the boiling point of water (100 C.) until all moisture is evaporated (Hartford and Frandsen 1992). Heating of organic layers to the high temperatures required for ignition cannot occur while moisture is present.

Specific relationships have been observed between duff moisture and duff consumption. At less than 30 percent duff moisture content, duff layers burn on their own once ignited, a threshold level observed in the southwest U. S., Pacific northwest, and the northern Rockies (Brown et al. 1985). At 30 to 120 percent duff moisture content, the amount of consumption of duff depends on the amount of consumption of associated fuel. When duff moisture content is greater than 120 percent, duff essentially will not be consumed.

Similar relationships have been found for organic soils. Peat is a deposit of slightly or non-decayed organic matter, while the organic content of muck is markedly decomposed (Buckman and Brady 1966). Peat burns well when its moisture content is below about 30 percent (Craighead 1974 in Hermann et al. 1991). Blocks of organic soil from south central Florida sustained smoldering up to 135 percent moisture content (McMahon et al. 1980 in Frandsen 1987).

Wet pocosin muck does not burn, but once the water table has lowered, these soils can ignite and sustain combustion. However, the moisture limits for ignition are not known (Frandsen 1993). Research is presently being conducted to determine factors affecting consumption in pocosin organic muck (Frandsen et al. 1993).

b. Surface fuel consumption. Heat generated by consumption of surface fuels can dry, preheat, and then ignite the duff layer. The amount of consumption of large diameter fuels was related to duff reduction and mineral soil exposure in north Idaho (Brown et al. (1991), and western Oregon and Washington (Little et al. 1986; Ottmar et al. 1990, Sandberg 1980).

c. Preburn duff depth. Duff consumption was strongly related to preburn duff depth in the northern Rocky Mountains (Brown et al. (1991); in jackpine (Stocks 1989 in ibid.); Alaska black spruce (Dyrness and Norum 1983); white spruce/subalpine fir (Blackhall and Auclair 1982 in ibid.); and southwestern ponderosa pine (Harrington 1987), but not in deeper duff layers of the Pacific Northwest (Sandberg 1980; Little et al. 1986). **d. Inorganic content.** Mineral soil becomes mixed with soil organic layers by freeze-thaw cycles, insect and small animal activity, overland flow, windthrow, and management actions, particularly skidding of logs (Hartford 1989). Mineral material affects combustion of organic layers because it absorbs some of the heat that would otherwise preheat combustible materials (Frandsen 1987). The greater the amount of mineral material in the organic layer, the lower the moisture content of the organic layer had to be before it would burn (ibid.). If the ratio of mineral particles to organic matter (mass ratio) was greater than about 4 to 1, smoldering did not occur (ibid.).

e. Phase of combustion. Almost all consumption of duff and organic soils occurs during the smoldering and glowing phases of combustion. Combustion can continue for hours, days, and in the case of pocosin soils, for weeks after ignition, if organic layers are dry (Frandsen 1993).

4. Dead Fuel Moisture. Fuel moisture has a significant effect on fuel availability and fuel consumption because it suppresses combustion. Part of the heat produced by the combustion of wood is used to drive off moisture in adjacent woody fuel. If the moisture content is high, the heat generated may be insufficient to dry these fuels and heat them to ignition temperature, and the fire will not continue to burn.

Fuel moisture is the ratio of the weight of moisture in the fuel to that of the dry weight of the fuel. The formula for fuel moisture and its effect on fire behavior is described in Chapter II.B.4.b.(2), this Guide. Moisture effects on woody fuel consumption were discussed in B.2 a. (5), this Chapter, and on duff and organic consumption in B.3.a. The following discussion describes the most important factors affecting moisture content of dead woody fuels, litter layers, and duff and organic layers. Live fuels and their moisture cycles are discussed in B.5., this Chapter.

a. Wetting and drying process. Water in fuels can be present in liquid or vapor form.

(1) Liquid water. Liquid water comes from rainfall, snowmelt, or condensation. It can be present both on the surface, and within cell cavities (Schroeder and Buck 1970). At the fiber saturation point (about 30 to 35 percent of the fuel dry weight), the cell wall holds as much water as it physically can, but no liquid water is present within cell cavities (McCammon 1976).

Liquid water is readily absorbed by fuels through their surface, filling cell cavities and intracellular spaces (Schroeder and Buck 1970). In liquid water, molecules travel with different speeds and directions. A water molecule at or near the surface of a layer of water can attain a high enough speed after colliding with another molecule to escape from the liquid water into the air. By this process, called evaporation, a liquid water molecule becomes a water vapor molecule (ibid.). Evaporation is the primary drying process when fuels are saturated. It decreases in importance as fuels dry below the fiber saturation point.

(2) Water vapor. The following discussion is derived from Schroeder and Buck (1970), except where noted. Water present in the atmosphere in the form of a gas is called water vapor. That part of the atmospheric pressure due to the presence of water vapor is called vapor pressure. The maximum amount of vapor that the atmosphere can hold when it is saturated depends on the air temperature. Water vapor molecules move from an area of higher concentration to one with lower concentration until vapor pressure is equal.

Water molecules in fuels can be bound to cellulose molecules or held by capillary action in tiny openings in the cell wall (Simard 1968). Molecules closest to the cell walls are held the most tightly. Successive layers of water molecules are held with progressively weaker bonds until the cell walls become saturated. At less than saturation, water vapor moves between a fuel particle and the atmosphere if the vapor pressure of the layer of water in the fuel does not equal the vapor pressure of the atmosphere. If the vapor pressure within the outer layer of fuel is greater than that of the atmosphere, moisture escapes to the atmosphere, and fuel moisture content decreases. If the vapor pressure of the atmosphere is greater than the vapor pressure within the outer layer of fuel, the fuel takes water vapor from the atmosphere, increasing the fuel moisture content.

b. Equilibrium moisture content. Equilibrium moisture content (EMC) is the "value that the actual moisture content approaches if the fuel is exposed to constant atmospheric conditions of temperature and relative humidity for an infinite length of time" (Schroeder and Buck 1970). The EMC determines the amount of water vapor that a specific piece of wood can hold (Simard 1968). A unique EMC exists for each combination of atmospheric temperature and relative humidity, with the associated vapor pressure (Schroeder and Buck 1970). If fuel moisture content is greater than EMC, vapor diffuses out of the fuel, and the fuel becomes drier. If fuel moisture content is less than the EMC, water vapor transfers into the fuel particle and the fuel becomes wetter.

Atmospheric temperature and relative humidity are never constant and tend to vary diurnally. Equilibrium moisture content also varies. Because fuels usually take up and release moisture at a slower rate than the temperature and humidity changes, the actual fuel moisture content lags behind the equilibrium moisture content. The greater the difference between the equilibrium moisture content and the fuel moisture content, the more rapidly vapor diffusion occurs, and the more rapidly the fuel particle exchanges moisture with the atmosphere. As a particle approaches equilibrium moisture content, the exchange occurs more slowly. Fuel moisture content never reaches equilibrium moisture content, because other physical processes prevent a complete exchange of vapor (Schroeder and Buck 1970). A fuel that is gaining moisture stabilizes at a lower moisture content than a fuel that is drying (Simard 1968). Van Wagner (1987 in Viney 1991) noted a 2 percent lower EMC for wetting fuels compared to drying fuels.

c. Timelag theory.

(1) Timelag principle. Drying and wetting of unsaturated dead woody fuel has been described by the timelag principle. A timelag has been defined as the length of time required for a fuel particle to reach approximately 63 percent of the difference between the initial moisture content and the equilibrium moisture content.[(1)]

(2) Timelag period. Under standard conditions, defined as 80 F. and 20 percent relative humidity, the length of time that it takes a fuel particle to reach 63 percent of EMC is a property of the fuel and is referred to as the timelag period (Schroeder and Buck 1970).

(3) Timelag classification. The proportion of a fuel particle exposed to weather elements is mathematically related to its size. Small diameter fuel particles have large surface area to volume ratios. Moisture levels in these fine fuels can change rapidly with changes in temperature and relative humidity. Large diameter fuel particles have small surface area to volume ratios, and their moisture content changes very slowly in response to changes in temperature and relative humidity. Time lag thus increases with increasing fuel diameter.

(a) Dead woody fuel timelag classes. Downed dead woody fuels have been grouped into size classes that reflect the rate at which they can respond to changes in atmospheric conditions (Lancaster 1970). The classes relate to an idealized surface area to volume ratio and an average timelag that represents each fuel class. Classes relate to the theoretical length of time required to reach 63 percent of EMC.

i. 1-hour timelag fuels - less than 1/4-inch diameter (less than 0.6 cm).

ii. 10-hour timelag fuels - between 1/4-inch and 1-inch diameter (0.6 to

2.5 cm).

iii. 100-hour timelag fuels - between 1-inch and 3 inches diameter (2.5 to 7.6 cm).

iv. 1000-hour timelag fuels - between 3 and 8 inches (7.6 to 20.3 cm) diameter.

(b) Forest floor timelag classes. There is a loose correspondence between these timelag classes and forest floor litter and duff, although the deeper the duff layer, the more approximate is the relationship. The corresponding classes assigned for fire danger rating purposes were (Deeming et al. 1977):

i. 1-hour timelag fuels - dead herbaceous plants and uppermost layer of forest floor litter.

ii. 10-hour timelag fuels - layer of litter extending from just below the surface to 3/4 of an inch below the surface.

iii. 100-hour timelag fuels - forest floor from 3/4 inch to 4 inches below the surface.

iv. 1000-hour timelag fuels - forest floor layer deeper than 4 inches below the surface.

(4) Timelag of other fine fuels. Weathered aspen leaves, tree lichen *(Alectoria jubata)* and some cheatgrass fuel beds were shown to act as

1-hour timelag fuels (Anderson 1990). The surface layer of lichens and mosses that carries fire in Alaska responds as a 1-hour fuel to temperature/relative humidity changes (Mutch and Gastineau 1970). However, conifer needle litter of some species belongs in the 10-hour timelag category (Anderson 1990), despite its high surface area to volume ratio. Other factors such as surface covering influence the rate at which fuel moisture changes in response to environmental conditions.

d. Fuel properties that affect dead fuel moisture content.

(1) Surface covering. The presence of a surface coating of organic material can limit movement of water, whether liquid or vapor (Simard and Main 1982). Dead woody fuel with bark gained and lost moisture at two-thirds the rate of the same diameter fuels without bark (Simard et al. 1984).

Moisture exchange in recently cast conifer needle litter is inhibited by a coating of fat, waxes, and cutin deposits (Anderson 1990). Anderson (ibid.) noted timelags of 5 to 34 hours for recently cast conifer needle litter, rather than the expected timelag of 10 hours. Weathering causes the breakdown and removal of needle coatings that slow vapor transfer. Timelags of 2 to 14 hours were measured in weathered conifer litter, which is still slower than the timelag of less than two hours expected for that diameter of particle (ibid.).

(2) Composition. The material of which a fuel is composed affects its structure, porosity, ability to gain or lose atmospheric moisture, and the movement of vapor within the particle. Composition and fuel moisture response properties vary significantly among dead woody fuels, deciduous leaf litter, grass litter, and coniferous needle litter (Simard and Main 1982).

(3) Amount of decomposition. Woody fuels that have been affected by weathering and decomposition often develop deep cracks that increase their surface area to volume ratio. Both liquid water and vapor can enter or leave the fuel through these splits in the wood, increasing the rate of moisture exchange. There may be few naturally occurring forest fuels that are actually 1000 hour timelag fuels, because almost all large pieces of wood have cracks that effectively increase their wetting and drying rates (Miller 1988, personal observation).

The cell structure in highly decomposed wood, such as rotten logs, has broken down, and moisture can travel more easily through this material than through solid wood. The moisture content of rotten wood can be very different from that of sound wood of the same diameter.

(4) Thickness and density of litter or duff layer. Because litter and duff layers have porosities of 70 to 90 percent, air can diffuse through them at 60 to 80 percent of the diffusion rate in free air (Fosberg 1975). The particles of organic matter in these layers exchange moisture with the atmosphere in the void space in the litter and duff layer (ibid). The moisture within the voids seeks equilibrium with the external atmosphere (ibid.). Van Wagner (1979) observed that the drying environment within a 3 centimeter (1.2 inches) deep needle litter layer was less favorable at the bottom than at the top because the lower part of the layer was farther from the drying surface. Anderson (1990) noted that moisture diffusion rates were slower if litter fuel beds were deeper or more densely packed. The lower part of a litter and/or duff layer can become matted and tightly bound by fungal strands (Harrington and Sackett 1990). The wetting and drying response of this layer is likely to be slower than that of the more loosely packed material nearer the surface because of slower rates of water vapor diffusion.

e. Soil moisture effects on fuel moisture content. Afternoon moisture content of eucalyptus *(Eucalyptus spp.)* leaf litter placed on wet soils was much higher than litter placed on dry soils (Hatton et al. (1988). Moisture appeared to diffuse upwards from the wet soil, increasing the relative humidity environment of the leaf litter, causing higher litter moisture content (ibid.). The biggest effect of the wet soil was noted in early mornings, probably because wet soils made more moisture available to condense on dead leaves. Litter on the dry soils was dry enough for the litter to burn throughout the night, while litter on wet soil plots was too wet to burn.

Active surface fire behavior occurred throughout many night time burning periods during the 1988 fire season in Yellowstone National Park. Night time moisture recovery of lodgepole pine litter was much slower and reached lower maximum levels than expected (Hartford and Rothermel 1991). The limited amount of surface litter moisture recovery was partially attributed to the lack of moisture in the air and in the soil that could contribute to an increase in night time litter moisture content (ibid.).[(2)]

f. Effect of weather factors on fuel moisture content.

(1) Precipitation duration. Wood absorbs water as long as the surface is wet, so precipitation duration is usually more important than precipitation amount in determining moisture content of dead woody fuels (Fosberg 1971 in Simard and Main 1982). The rate of diffusion of liquid water into wood is usually less than the rate at which precipitation occurs, so much of the rain water drips off before it can soak into the wood.

(2) Precipitation amount. The amount of precipitation is more important than the duration of precipitation in determining the moisture content of duff, organic soils, and accumulations of organic materials that occur beneath isolated trees and shrubs. Duff layers and organic soils retain much of the precipitation that falls, allowing it to slowly soak into the fuel particles (Simard 1968 in Simard and Main 1982).

(3) Temperature. Temperature affects both the humidity of the air and its vapor pressure, and thus the equilibrium moisture content. Higher fuel temperatures decrease relative humidity in the microclimate near the ground (Rothermel et al. 1986), which also decreases the EMC. Higher fuel temperatures increase the tendency of bound water vapor to diffuse away from the fuel, thus drying it further (Schroeder and Buck 1970). Fuel temperature is affected by slope, aspect, time of day, cloud cover, canopy cover, sun angle, and the albedo of the fuel.

(4) Relative humidity. Relative humidity has a significant effect on moisture content of small diameter fuels because water vapor can readily penetrate into or escape from the center of small fuel particles. Diurnal changes in relative humidity have little effect on the moisture content of large diameter fuels because their large volume prevents rapid movement of moisture molecules between the surface of the fuel and its center. Relative humidity can have a major effect on large fuel moisture content if there is a long period of time without precipitation.

(5) Wind. Wind has its most important drying effect on woody fuels when liquid water is evaporating because it removes any layer of water vapor that may be adjacent to the fuel. Wind has a greater effect on wet fuel particles that are above the surface, causing them to dry more rapidly than material on the ground (Simard and Main 1982). When fuel is below the fiber saturation point and most vapor loss is by diffusion, the effect of wind becomes less important as the fuel becomes drier

(Schroeder and Buck 1970). Wind has a more significant drying effect on small diameter fuels than on large diameter fuels, duff, or organic layers.

g. Relationship of topography to fuel moisture content.

(1) Fuel moisture tends to vary with topographic position. Fuels are less directly exposed to sun on north slopes than south slopes so their moisture content tends to be higher. Temperatures are generally cooler and humidities higher at upper elevations, so fuel moistures are usually higher than at lower elevations.

(2) Topography partially determines the strength of any night time inversion layer that forms. If a steep inversion and temperature gradient forms, fuel moisture recovery can be fairly high because of low temperatures and high relative humidities.

If an inversion forms in a valley, a thermal belt may form at the top of the inversion layer. In this belt, temperatures are warmer than at lower elevations within the inversion, and warmer than at higher elevations because temperature decreases with altitude. Higher night time temperatures, lower relative humidities, and lower fuel moistures occur within the thermal belt than at other locations along the slope. Fires can remain active throughout the night within the thermal belt, while activity is limited below the inversion layer (Schroeder and Buck 1970).

5. Live Fuel Moisture.

a. Effect of live fuel moisture on fire. Live fuels can either be a heat sink or a heat source in a wildland fire, depending on their moisture content. If live fuel moisture levels are high enough, they absorb some of the heat produced by associated burning fuels without themselves igniting, and thus do not contribute to the progress of the fire. If live fuel moisture is low, the combustion of dead fuels readily produces enough heat to desiccate and ignite the live fuels, which then add to the total amount of heat released by the fire (Burgan and Rothermel 1984). Live fuels can thus retard, stop, or contribute to fire spread.

b. Factors regulating live fuel moisture.

(1) Internal factors. Moisture content of living plants is controlled largely by species morphology and physiology. The amount of water in plant tissue, and thus its moisture content, relates closely to events during a plant's seasonal growth cycle (plant phenology). For a given species, the maximum and minimum moisture content values and the average values during different parts of the growing season are controlled more closely by the plant structure and its adaptations to the general climate of the area, than by daily weather. Seasonal timing of drying for specific deciduous shrub, forb, and grass species were found to be similar between wet and dry growing seasons, although moisture levels were generally higher in the wet season (Brown et al. 1989).

(2) Site factors. Site conditions can cause differences in moisture content within the same species, possibly because of physiological conditioning or even a genetic adaptation to the site (Reifsnyder 1961). Differences in foliar moisture content within a single species were related both to differences in substrate and the amount of shading provided by a forest canopy (Blackmarr and Flanner 1968).

(3) Climatic variation. Climate affects such factors as the timing and length of the growing season, the length of the green-up period, and the existence of seasonal periods of cold- induced dormancy or drought or heat-induced quiescence.

c. Differences among species groups. There are characteristic differences in seasonal moisture patterns for groups of species. Deciduous leaved woody plants tend to have higher moisture content values than evergreen leaved plants, and the seasonal pattern of moisture changes tends to vary more. Coniferous trees have entirely different foliar moisture patterns than deciduous trees. Herbaceous species moisture levels can be higher or lower than that of associated shrubs, depending on the species present and the time of year. There are differences in average maximum and minimum moisture values among species within any group, depending upon the morphology of the species, and the relative amount of new and old growth on the plant.

d. Deciduous leaved shrubs. The general pattern for deciduous leaved shrubs is for moisture to rapidly increase to a peak level soon after bud break and begin to decrease after all new seasonal growth has occurred. Moisture then slowly declines for the remaining part of the growing season until leaves cure.

Data from Alaskan aspen stands illustrate variation in moisture content levels among deciduous species, as well as variability due to site differences (Norum and Miller 1981). Maximum spring moisture content of leaves and small twigs of highbush cranberry *(Viburnum edule)* (Illustration III-1, page III-24) was about 325 percent, but its moisture content dropped to about 225 percent by midsummer where it remained for most of the growing season. Rose *(Rosa acicularis)* on that same site in that same season had a spring maximum value near 375 percent, but its moisture content decreased to about 175 percent where it remained until fall curing. Maximum moisture levels for that same species of rose on a drier aspen site were less than 250 percent and persisted at about 165% for much of the growing season. Blueberry *(Vaccinium uliginosum)* (Illustration III-2, page III-25), a smaller stature deciduous species on a black spruce site nearby, had spring maximum moisture value of less than 250 percent and spent most of the summer at about 125 percent moisture content (Norum and Miller 1981).

For all of these species, moisture content did not significantly decrease as fall coloration appeared on the leaves. Moisture content began to drop markedly as the abscission layer formed at the bases of the petioles and cut off water transport to leaves, when obvious drying and browning of the leaves occurred (Miller 1981).

e. Evergreen leaved shrubs. The general pattern for broad-leaved evergreen shrubs is more complex than for deciduous species because evergreen shrubs sometimes retain old leaves for several years. They tend to have lower spring maximum values and much lower growing season average values than deciduous species. Values increase from an overwintering minima as new growth is added in the spring, or at other times of the year when precipitation triggers growth after a dry season. Values decrease significantly after new growth ceases. Some evergreen leaved species develop ephemeral leaves in late winter and early spring. Average foliar moisture content drops significantly as these seasonal leaves cure and drop from the shrub.

A typical profile for sagebrush *(Artemisia tridentata)* moisture content would be a rise from early to late spring from about 150 percent to about 250 percent, with a subsequent decline to 60 percent or less in mid to late summer (Schmidt 1992). Riedel and Petersburg (1989) found that the lowest summer levels for sagebrush moisture (Illustration III-3, page III-26) were reached one month earlier in one year than the previous summer. Sagebrush flammability has been related to threshold levels of moisture content. (See II.C.2., this Guide.)

In the Alaskan interior, maximum foliar moisture content levels for Labrador tea *(Ledum decumbens)* were only 145 percent, and that peak value occurred about a month after the maximum moisture values were reached in associated deciduous shrub species (Norum and Miller 1981). Moisture content of new leaves of chamise *(Adenostoma fasciculatum)* in California were at 125 percent in late May, dropped to about 60 percent in early September, and rose to about 90 percent when the plants again became physiologically active in early December (Dell and Philpot 1965). Maximum moisture levels for galberry foliage *(Ilex glabra)* averaged about 140 percent in North Carolina, while minimum values were about 100 percent (Wendel and Story 1962). Maximum values for redbay *(Persea borbonia)* foliage were about 120 percent, while fall and winter minima were around 60 percent (ibid.).

f. Herbaceous plants. Herbaceous moisture content can also vary significantly among species. Moisture levels can be much higher at the beginning of the growing season than for other species groups because all of the plant is new tissue. Also, because there is no residual material, all of the plant can become cured, sometimes before the end of the growing season. This is especially notable for grasses and other species in areas with hot, dry summer weather. In north Idaho, moisture content of cheatgrass *(Bromus tectorum)*, an annual grass, for example, was measured to be 150 percent on June 20, but was only 9 percent on July 20 (Richards 1940). All of the plant material, once cured, responds to atmospheric conditions as a dead fine fuel, as reflected by the 9 percent moisture level just cited.

Some species of grasses and forbs in some regions can produce new growth in the fall, after a summer of quiescence, thus causing fall green-up and associated increase in moisture content. Green-up is caused by renewed growth of perennial species and germination of seeds.

Some herbaceous species do not cure and dry out during the summer, rather only begin a significant amount of curing as frost occurs in the fall. In north Idaho, moisture content of fireweed *(Epilobium angustifolium)* plants was 426 percent on June 20 and 241 percent on September 10 (ibid.). In interior Alaska, bluejoint reedgrass *(Calamagrostis canadensis)* was first measured at about 400 percent moisture content on May 27 when the plants had about 1 to 1-1/2 feet of leaf growth. Moisture content of plants declined to about 260 percent by June 30 and was about 200 percent on August 28, just before the first frost (Norum and Miller 1981). In north central Michigan, large leaved aster *(Aster macrophyllus)* was measured at about 420 percent moisture content at the beginning of June, and the lowest moisture level observed for the rest of the summer fluctuated around 300 percent (Loomis and Blank 1981).

Data for most herbaceous species show only a slow decrease in moisture levels after early growing season maxima. However, in western Wyoming, grasses and forbs had some increase in moisture content in response to mid-summer rain. By September, however, the drying trend was not altered by rainfall (Brown et al. 1989).

g. Coniferous trees. Moisture content of coniferous foliage also varies significantly with season but the pattern is quite different than that shown by deciduous and herbaceous species. Coniferous species retain their needles for several years; the number of years is a species characteristic.[(3)] For most species, the lowest level of moisture content of needles formed in previous

years occurs in late spring, during about the same time period in which buds expand, and new needles and twigs are formed. Moisture content of old needles increases during most of the growing season to a maxima during late summer and/or early fall (depending on species and region).

Moisture content of new needles is very high as buds break, needles grow, and stems elongate, but starts dropping significantly about the same time as the new terminal bud on the end of the current year's growth forms. The moisture content of new foliage drops to about the same level as that of old foliage late in the growing season. In the southeastern U.S., conifers may flush more than once during the growing season. The moisture cycle in older foliage may be different from that of conifers that grow in climates with winter cold and/or shorter growing seasons.

The difference between low and high moisture values for 1-year-old black spruce *(Picea mariana)* foliage in Alaska varied from 28 to 40 percentage points on different collection sites (Norum and Miller unpublished). Seasonal lows occurred in June and seasonal high values in August (Illustration III-4, page III-27). Seasonal low values for Douglas-fir foliage occurred in mid to late June in Montana, with a peak value reached by early September (Philpot and Mutch 1971; Rothermel 1980). The range between high and low values varied from about 20 to 40 percentage points.

Crown fires were much more prevalent in Douglas fir trees burning in late spring and early summer experimental fires than on those sites burned in late summer and early fall (Norum 1975). Low springtime foliar moisture values may explain the observed difference. The peak of the fire season in boreal latitudes is usually shortly after summer solstice, and the low foliar moisture content of black spruce may be a factor in tree crown ignition. However, the fire season in the western United States generally peaks in August, at a time when moisture content of conifers is increasing to a seasonal high. The readiness of western species of conifers to crown during extreme fire weather is not due to low foliar moisture values, although it may be enhanced by early drying of the oldest needles.

Measured moisture content of live foliage in northeast Oregon, southwest Idaho (Miller 1988), and northwest Wyoming (Hartford and Rothermel 1991) in the extreme fire season of 1988 was not different from normal moisture values for that time of the year. Experimental evidence suggests that conifers can rapidly transport water into their foliage when heated to temperatures that occur in a wildland fire (Cohen et al. 1990). This can delay branch and foliage drying and could inhibit crown heating and ignition. During drought conditions, trees may not be able to transport water into their crowns, increasing their flammability and hence their crowning potential (ibid.).

6. Fuel Succession. Vegetative biomass tends to accumulate over time. However, not all biomass is available fuel. Biomass is all of the vegetation on a site, while available fuel is what can burn. Fuel succession is the change in the fuel complex over the long term, including changes in loading, size distribution, availability, and live to dead ratios. These changes are the net result of the counteracting processes of accumulation and depletion (Brown 1987).

a. Accumulation.

(1) Litter layer. The amount of foliage that is produced each year affects the amount of new litter that accumulates. Because coniferous trees retain their needles for several years, there may be no relationship between the productivity in a particular year and the amount of needle litter added to the surface fuel layer.

(2) Dead woody fuels. Insects, disease, suppression of individual trees in young stands, and death of lower branches of trees can provide a source of dead woody fuels. These events, and the timing of the addition of the fuels, occur irregularly, as branch material is broken or entire trees fall because of wind, and heavy snowfalls (Brown 1975). Fire can kill trees and shrubs, and whatever woody material is not consumed can become surface dead woody fuels.

(3) Duff and organic layers. Material is added to these organic layers as the lower part of the litter layer, and moss and lichen layer, decompose. Rotten woody material is gradually incorporated into the forest floor, and becomes part of the duff layer.

(4) Live fuels. Shrubs, herbaceous plants, and young conifers can establish and/or increase in volume. Branches die, and increase the flammability of trees and shrubs.

b. Depletion.

(1) Litter. Dead plant material can oxidize and essentially disappear during one growing season. It can remain into the next growing season, be compacted beneath additional litter, and decompose enough to become part of the duff layer.

(2) Dead woody fuels. Dead woody fuels physically deteriorate and settle over time, and compactness increases as supporting branches decay (Brown 1975). A more compact fuel bed is less well aerated, and may dry more slowly, have a higher moisture content, and be a more favorable environment for additional decomposition.

(3) Live fuels. Productivity of an understory layer of shrubs and herbaceous plants can decrease significantly as the canopy closes, resulting in a much lower annual addition of litter. A coniferous fuel ladder can grow tall enough that its lower branches no longer provide a bridge between surface and crown fuels.

c. Patterns of fuel succession.

(1) Forests. The generality that downed woody fuels accumulate over time is, in many cases, not true (Brown and See 1981). The amount of forest fuel depends on stand history, whether the stand was visited by insects, disease, wind, and fire, and at what intervals. The size and pattern of disturbance, and amount of fuel that results, can vary with the event, and tree and branch mortality can be compounded by drought. Agee (1993) also relates the amount of forest fuel to stand disturbance. Changes in the amount of fine and coarse woody fuels over time relate to the amount of biomass present before a disturbance, the severity of the disturbance, and successional patterns after the stand is disturbed (ibid.).

(a) Relationship to stand disturbance. When a wildland fire occurs in a forested stand, the severity of the impact on the stand, and resulting amount of surface fuels and rate of their accumulation, can vary (Muraro 1971 in Brown 1975). For example, if a fire occurs in a lodgepole pine stand that burns only in surface litter layers, it can kill or weaken many of the trees but not consume much of the foliage. Surface fuels increase moderately as trees die and fall. A fire in a lodgepole pine stand that burns into the duff layer can consume many structural tree roots. This makes trees susceptible to rapid blowdown, and fine fuels are added to the stand at a much higher rate than after a lower severity fire. A high intensity crownfire in a lodgepole pine stand can burn off many of the fine branches in the tree crowns. If it also burns deeply into the duff layer, most of the trees will fall fairly quickly. Most of the fuel added would be large diameter material. Because downed trees are not supported by small diameter branchwood, they would come into contact with forest floor sooner and decompose more readily.

Whether the young stand of lodgepole pine that establishes after fire has a low or high dead fuel loading also depends on the frequency of fire. The stand that develops after a fire that caused rapid blowdown of trees with a lot of branchwood would have a high loading of dead fuel in all size classes. If a fire occurs in this young stand of trees, much of the crown-stored seed could be destroyed and most of the fuel consumed. A sparse stand of lodgepole pine could subsequently establish that has a much lower loading of dead woody fuel than the previous stand (Muraro 1971 in Brown 1975).

(b) Varying patterns among live and dead fuels. Fuel succession is more complicated if live and dead fuels are involved (Brown and See 1981). There may be an increase in one class of fuel while another is decreasing or becoming unavailable. Dead woody fuel may decompose while an understory of trees establishes. The loading of dead woody fuel may increase while some trees become tall enough to be much less available to surface fire (Brown and See 1981). Early successional herbs, such as bracken fern in western Oregon, can cause a high loading of fine fuels before the canopy closes and shades out these plants (Isaac 1940 in Agee 1993).

(c) Relationship to stand age. There is no clear relationship in the northern Rockies (Brown and See 1981) between stand age and amount of dead woody material. The amount of fuel in young and mature forests cannot be related to age because too many other factors are involved. The only generalities are that downed woody fuel loadings tend to become predictably high as stands acquire old growth characteristics, but loading is unpredictable from age alone in young, immature, and mature stands (ibid.).

In western Oregon and Washington, stand replacing fires generally occur at much longer intervals than they do in the northern Rocky Mountains. Fuels in the 0 to 3-inch (to 7.6 cm) range are usually at their highest levels in early stages of postfire succession (Agee 1993). 1000-hour fuel biomass is highest in mid-successional stages when some stems die because of naturally occurring self-thinning. Biomass of larger logs is greatest in the oldest stands.

(d) Relationship to site productivity. Fuel loading and site productivity are not well correlated (Brown and See 1981). In warm, moist forest types, productivity is fairly high, but fuel may not accumulate because the decomposition rate keeps up with fuel production (ibid.). In cool, dry forest types, productivity tends to be low, but a relatively higher proportion of biomass may accumulate as fuel because decomposition is limited (ibid.).

(e) Relationship to fire exclusion. In many areas of the western U.S., naturally occurring fires used to occur at a fairly high frequency. With the onset of organized fire suppression activities, and the removal of fine fuels by livestock grazing, wildland fires became an infrequent event in many forest types. If fire exclusion has removed several fire cycles from a forested stand, the ecological effect is much more profound than if fire has only been effectively suppressed for one-third of the length of a stand's natural fire rotation. In parts of the southwestern U.S., for example, the exclusion of fire from ponderosa pine stands that previously burned at intervals ranging from 2 to 10 years has resulted in higher loadings of litter, forest floor duff, and in some cases, down dead woody fuels (Harrington and Sackett 1990). While the amount of fuel may not be predictable from age, it is logical to conclude that there is more fuel in stands without understory fire for 80 to 100 years than if underburns had continued to occur at frequent intervals.

(2) Shrublands.

(a) Sagebrush. The percentage of dead stemwood in sagebrush *(Artemisia tridentata)* plants increases with age. However, when modelling the effect of higher proportions of dead branchwood on fire behavior, only a small increase was found (Brown 1982). The total amount of fuel correlates to the height of the stand, but stand height does not correlate well with age (ibid.). The amount of fuel in older stands of sagebrush is greater if the volume and density of shrubs has increased.

(b) Chaparral. Old stands of chaparral have been observed to be more flammable than young stands, and this difference has been attributed to an increasing proportion of dead branch material in older age classes of shrubs (Paysen and Cohen 1990). No correlation was found between the percentage of dead branch material and age of chamise in southern California (ibid.).

Because all leaves and fine branch material in the chaparral canopy tends to be consumed by fire when foliar moisture content is low, a stand with more leaves and twigs has more fuel. For any given site, the amount of biomass tends to increase with age of the stand, and it may be this increase in total biomass that causes the higher flammability observed in old stands. However, because of variability in site productivity and species composition, it cannot be said that every stand of chaparral of a certain age is more flammable than a stand that is younger.

C. Resource Management Considerations

The primary ways to manipulate fire effects on fuels are to modify fuel availability and to change the way an area is ignited and burned.

1. Fuel Availability.

a. Fuel moisture.

(1) Change the prescribed fuel moisture. When planning a prescribed fire, the moisture contents specified in the prescription can be chosen to achieve selected effects on fuels.

(a) Fine fuel moisture. Fine fuel moisture indirectly affects overall fuel consumption by determining which fuels ignite. Fine fuel moisture is defined by specifying different ranges of temperature and relative humidity in the prescription.

(b) Large fuel moisture. In forested areas, the moisture content of large diameter woody fuels is the chief factor affecting the amount of total consumption. Remember that rotten woody material can burn at a much higher moisture content than sound material of an equivalent size.

(c) Duff and organic layers. Consumption of soil organic material is also directly related to its moisture content (Brown et al. 1985).

i. At moisture content greater than about 120 percent, duff will not burn.

ii. At moisture content less than about 30 percent, duff will sustain combustion on its own once ignited.

iii. The amount of consumption of duff between 30 and 120 percent moisture content depends on the amount of consumption of associated fuels. **(d)** Live fuels. By prescribing the moisture content of live fuels in the surface fuel layer, the amount of their flammability and consumption is regulated. The amount of scorching of a conifer canopy may be greater early in the growing season when trees are just becoming physiologically active and foliar moisture content is lower than it is later in the year. Live fuels may be consumed, regardless of their moisture content, if a large loading of dry, dead woody material burns.

(2) Alter the fuel moisture. Use of water or foam changes the moisture and burning characteristics of fuel. These techniques are commonly used to build fireline and protect specific features, such as wildlife trees.

b. Fuel loading and distribution.

(1) Remove the fuels. Less fuel is available, and there is less potential for heat release, if fuels are removed from a site. Fuels can be removed by:

(a) Grazing.

(b) Commercial thinning of forests.

(c) Firewood sales.

(d) Yarding unmerchantable material to a central location during forest harvesting operations.

(2) Change the fuels. If fuel distribution or arrangement is changed, flammability, and the potential for heating, changes.

(a) Crushing. Crushing fuels increases fuel bulk density and can make the rate of burning slower. However, if crushing compacts fuels to a more ideal arrangement, it may enhance combustion.

(b) Lopping and scattering. Cutting and scattering of branches during a logging or thinning operation makes fuel continuity more uniform, but also decreases the potential for concentrated heating where piles of branches would have been located.

(c) Piling or windrowing. Piling or windrowing fuels breaks up the continuity and decreases the likelihood that fire can spread. A fire that starts (or is started) in a pile or windrow has a greater potential for subsurface heating if low moisture content of larger pieces and low amounts of intermixed mineral soil permit a high degree of consumption.

(d) Chaining. Chaining woodlands or shrub dominated areas alters the distribution and continuity. If removal of the trees or shrubs allows more grasses and forbs to grow (or if they are seeded), the flammability of the site will be significantly higher because of the presence of downed woody fuels.

(e) Herbicide. The use of herbicide to kill shrubs and woodland trees results in a large amount of standing dead vegetation. Intermixture of newly established grasses and forbs will result in a highly flammable site.

c. Fuel chemistry. Application of long term fire retardants inhibits fuel ignition and hence fuel consumption.

2. Ignition.

a. Backing vs. heading fires. Backing fires usually result in more complete fuel consumption, particularly of litter and duff layers, than heading fires.

b. Mass firing. Use of mass firing techniques, such as center firing or concentric firing, may result in more complete consumption of fuels, for a given moisture regime, than if a backing or heading fire were used.

c. Ignition devices. Use of ignition devices such as a heli-torch that can apply a lot of fire in a short period of time can result in a fire that causes more woody fuel consumption than if surface ignition were used.

D. Methods To Monitor Fire Effects

Fuels inventory data are collected to facilitate accurate prescription development, to determine if fuel consumption objectives are met, and to relate fuel reduction to fire effects on other resources. Fuel moisture data can determine whether prescribed conditions are met, and document the conditions that correlate with specific amounts of fuel consumption and related aspects of the heat regime of the fire. If smoke emissions are a critical factor in a prescribed fire program, both fuel moisture and fuel consumption data can be used to predict emissions, refine prescriptions, and obtain an accurate estimate of the amount of emissions produced by a particular prescribed fire. While mineral soil is not a fuel, soil moisture data can provide important information for documentation and interpretation of fire effects that are related to sub-surface heating.

1. Fuel Loading. The type and amount of fuels inventory should match the objective for

doing the inventory, because fuels data can be expensive and time consuming to collect. Specific techniques have been developed for inventorying or estimating living and dead biomass in forest and rangeland vegetative types, many of which were developed specifically for assessing fuels. The time of year when sampling is performed can be critical if any component of live vegetation is being measured, particularly grasses and forbs. Sampling performed before the full amount of seasonal growth has occurred can produce serious underestimates in fuel loading. Sampling during the normal fire season, or during the specific time of year when a prescribed fire is planned to occur, is recommended. Agency specific guidance for fuels measurement in forests and in grassland and brush is provided in USDI-NPS (1992).

a. Destructive sampling. Destructive sampling is the clipping, sorting by size category, and weighing of all fuel in a representative area. This is an extremely accurate way to collect fuels data but is also time consuming and expensive. All of the other procedures detailed here derive estimates of fuel loading from specific sets of measurements.

b. Estimating weight of herbaceous fuels. There are many techniques for estimation of weight and production of herbaceous rangeland vegetation because of its use as livestock forage. Most techniques for weight estimation can be placed into one of three categories:

1) clipping and weighing, 2) estimation, and 3) a combination of weighing and estimation (Brown et al. 1982). Details on use of these and related methods can be found in Hutchings and Schmautz (1969), and Chambers and Brown (1983).

c. Estimating shrub weight.

1) Rangeland shrubs. Average height of an entire stand of big sagebrush can be estimated by multiplying 0.8 times the average height of the tallest plants in that stand (Brown 1982). Average sagebrush height and percent can be converted to tons/hectare (tons/acre) (ibid.). Martin et al. (1981) developed estimates for average loading by percent of crown cover for big sagebrush, antelope bitterbrush *(Purshia tridentata)*, snowbrush ceanothus *(Ceanothus velutinus)*, and greenleaf manzanita *(Arctostaphylos patula)*.

2) Forest shrubs. Shrub biomass can be estimated from basal stem diameters for 25 species common in the northern Rocky Mountains (Brown et al. 1982).

d. Live/dead ratio. The live/dead ratio within plants can be obtained by ocular estimation or through more time consuming destructive sampling techniques.

e. Inventory of dead woody fuels and duff.

1) Direct measurement. Brown et al. (1982) provides comprehensive procedures for inventorying downed woody material, forest floor litter and duff, herbaceous vegetation, shrubs, and small conifers. Field sampling methods include counting and measuring diameters of downed woody pieces that intersect vertical sampling planes, comparing quantities of litter and herbaceous vegetation against standard plots that are clipped and weighed, tallying shrub stems by basal diameter classes, tallying conifers by height classes, and measuring duff depth (ibid.). All of these procedures can be completed at one sample point in about 15 minutes. The authors recommend that at least 15 to 20 sample points be located in an area where fuel estimates are desired. Although these procedures apply most accurately in the Interior West, techniques for estimating biomass of herbaceous vegetation, litter, and downed woody material apply elsewhere (ibid.).

Formulas for calculating fuel loading from field measurements are found in Brown (1974). Anderson (1978) provides graphs from which loading can be estimated. The calculation procedures are converted into a computer program listed in Brown et al. (1982). Agency fire management staff may have software that can be used to calculate fuel weights from these inventory data.

2) Photo series. A photo series developed for a specific fuel type in a defined geographic area can be used to obtain an estimate of fuels. The stand of interest is compared to pictures of similar stands in which fuel inventories have been conducted. Precision is intermediate when compared to other methods for obtaining fuels information. Photo series are more accurate for assessing fire potential than for estimating fuel loads (Fischer 1981a). Photo guides are available for natural and activity fuels for coastal and interior forest types in the Pacific Northwest (Maxwell and Ward 1976a, 1976b, 1980); for forest residues in two Sierra Nevada conifer types (Maxwell and Ward 1979); for natural fuels in Montana (Fischer 1981b, c, and d); for thinning slash in north Idaho (Koski and Fischer 1979); and for natural forest residues in the southern Cascades and northern Sierra Nevada (Blonski and Schramel 1981).

Supplementary information on fire behavior and resistance to control were compiled for existing photo guides for Pacific Northwest coastal forest (Sandberg and Ward 1981); for two Sierra conifer types (Ward and Sandberg 1981a) and for Northwest ponderosa and lodgepole pine types (Ward and Sandberg 1981b). There are presently no photo series for the Great Basin or southwest U.S. Fischer (1981a) explains how a photo guide is constructed with enough detail for a field office to prepare a series on specific fuel types.

2. Woody Fuel Consumption. Fuel consumption is measured by comparing prefire fuel loading data with data collected after a wildfire or prescribed fire is extinguished. If a quantitative fuel reduction objective was set, and a related fuel inventory technique selected and performed before the fire, that same inventory must be conducted again. Changes in fuel loading can be less precisely estimated by comparing photo series pairs that match the appearance of the site before and after burning.

In some cases, total downed woody fuel increases after a fire because of the addition of branchwood and boles of trees that fell as a result of the fire. If this has occurred, the observation should be recorded with field data, as it will help interpret fuels data when the project is being evaluated.

3. Litter/Duff Reduction. Techniques for measuring litter and duff reduction are described in Chapter II.D.8., this Guide, "Burn Severity/Depth of Burn."

4. Fuel Moisture.

a. What should be sampled. Categories of fuel moisture that can be related to the heat regime of a fire and to fire effects include the following:

- less than 1/4-inch diameter down dead woody fuels (1 hour fuels)

- 1/4 to 1-inch diameter down dead woody fuels (10 hour fuels)

- 1 to 3-inch diameter down dead woody fuels (100 hour fuels)

- 3 to 8-inch diameter sound down dead woody fuel (1000 hour fuels)

- large diameter rotten down dead woody fuel

- surface litter

- thin duff layer

- upper part of a deep duff layer

- lower part of a deep duff layer

- organic soils

- organic layers beneath isolated trees and shrubs

- mineral soil

- tree foliage

- shrub foliage

- herbaceous plants

Moisture data required varies with vegetation type, expected fire behavior and fire characteristics, the fuel situation on the site, and the objectives for conducting the fire. While not a fuel, mineral soil is included in this list because of its role in regulating heat transfer into soil. (See Chapter V.B.1.a, this Guide.)

b. Where fuel moisture should be sampled. The following discussion is derived from Norum and Miller (1984), and Sackett (1981). Fuel moisture samples should be collected within the proposed burn unit and be representative of the area. Samples should span the range of vegetative conditions, fuel conditions, elevation, aspect, and slope on a prescribed fire site, because these variables can lead to notably different fire characteristics and fire effects. Notably wet and dry microsites should be sampled, along with the areas between them. This also applies to shaded and exposed spots, greater and lesser concentrations of fuel, older and younger stands, and any other within-plot variations that might influence fuel moisture content.

If fuels inside and outside of the prescribed fire unit are notably different, as in the case of a clearcut, fuel moisture outside the burn unit should also be monitored and documented. Differences in anticipated fire behavior within and outside of the intended fire area help to determine the probability of a spot starting a fire outside of the unit and the needed contingency suppression forces in case the fire escapes.

c. The number of samples to collect. Prefire variability in moisture content of fuels can be fairly high. Prefire samples can be used to determine how many samples must be collected to guarantee the needed sampling precision. See Chapter XI.B.1., this Handbook, for an example of how to determine how large a sample size is required.

d. Direct sampling. Detailed discussions of fuel moisture sampling methods and drying procedures are given in Norum and Miller (1984) and Countryman and Dean (1979). While these two publications were designed for specific geographic locations, the general principles involved can be applied to other parts of the country.

(1) Containers. Commonly used containers are aluminum soil sample cans, paint cans, nalgene bottles, and wide-mouth glass jars. Plastic bags, even if they have a tight seal, are not suitable for sample collection. Moisture can escape through small pores in the plastic, especially if the sample sits for a while before processing. Moisture from the sample can condense on the bag, and be lost when the sample is transferred to another container for drying. Use of plastic bags for sample collection can result in underestimation of sample moisture content.

(2) General field procedures. Detailed procedures for collecting fuel moisture samples in Alaska were developed by Norum and Miller (1984). The publication contains many general procedures which can be followed in any part of the country. Some general guidelines include:

(a) If recent rain, frost or dew have left obvious moisture on the surface of the plants, sample moisture content may be overestimated.

(b) Material collected from living plants, leaf litter, and upper duff layers becomes fairly stiff as it dries, and may expand, causing it to spring from the sample containers during the drying process. Material must be loosely packed into sample containers. Stems and leaves of live fuels can be cut into small pieces as they are placed in the sample can.

(c) Samples must be kept cool, dry, and out of direct sunlight until they are processed. Countryman and Dean (1979) recommend placing samples within an ice chest until they can be brought back to the lab for processing. Lunch coolers with a container of ice can also be used. If samples cannot be processed immediately, refrigerate them, still sealed, until they can be weighed.

(e) Live fuels. Guidelines for collecting specific species of plants are given in Norum and Miller (1984), and can be adapted to other species. Plant material sampled should consist only of living foliar material, not dead branches, dead leaves, flowers, or fruits. A consistent manner of sampling is most important, both for each species of plant and throughout the growing season. Plant growth stage at the time of sampling should be noted.

(3) Processing samples. A basic requirement for processing of fuel moisture samples is a top-loading beam or torsion balance scale, capable of measuring to 0.1 gram. If many samples must be processed over the course of a field season, or several seasons, the cost of an electronic balance may be justified because of the time saved and accuracy that is gained.

Several different means exist for determining fuel moisture content in the office or lab once samples have been collected.

(a) Xylene distillation. The xylene distillation method is a laboratory procedure which produces extremely precise estimates of moisture content for both live and dead fuels. However, this method is comparatively expensive and takes a significant time to perform. It will not be discussed further here.

(b) Microwave oven. Microwave ovens have been used successfully to dry dead woody fuels (Norum and Fischer, 1980). McCreight (1981) did not recommend use of a microwave oven for drying live fuels.

(c) Computrac®. The Computrac, Model FS-2A is a moisture analyzer that weighs and dries a small sample and provides a moisture content on a dry-weight basis. Material is dried in an automatically controlled oven chamber, continuously weighed, and moisture content calculated.

Results are obtained within about 10 to 20 minutes for dead woody fuels, or about one hour for live fuels. Fuels can be dried at 95 C. (203 F.), minimizing any loss of volatiles.

While quite accurate, a major disadvantage of the Computrac is the very small size of the sample which can be processed, approximately 3 to 10 grams of material. In order to obtain a representative sample, many samples must be subsequently processed. A second major disadvantage of the Computrac is its high purchase price.

(d) Drying ovens. Detailed procedures for use of a scale and drying oven can be found in Countryman and Dean (1979) and Norum and Miller (1984). Processing of fuel moisture samples in a drying oven has long been the standard for measurement of fuel moisture content. Samples are weighed on a scale to the nearest 0.1 gram, dried in the oven, and then weighed again to determine the amount of water lost. Ovens are customarily set to 100 C. (212 F.) for dead woody fuels, and 80 C. (176 F.) for live fuels. The standard drying time is 24 hours. Major advantages of a drying oven are that many samples can be processed simultaneously, and accurate values are obtained if proper procedures are followed. The disadvantage is the 24 hour delay in arriving at the values for moisture content.

e. Fuel moisture meters. Several brands of fuel moisture meters are presently available that provide a direct measurement of fuel moisture. Most meters work by measuring the electrical resistance between two probes which are inserted into a piece of wood. Most of these meters were developed for testing the moisture content of kiln dried lumber and are most accurate at lower moisture values. Some of these probes are calibrated on a wet weight basis, not a dry weight basis, and will not give answers that can be used as input to fire behavior prescriptions. Most of the probes are less than an inch in length and cannot penetrate deeply enough into large diameter wood to measure its moisture content. However, a fairly accurate measurement of large fuel moisture content can be made by cutting across the diameter of a large piece of woody fuel and inserting the probe into the freshly exposed surface. Because meters were developed to measure moisture content of wood, a fairly dense substance, they cannot give an accurate reading of moisture content within litter or duff layers, or of soil. These meters are not suitable for live fuel moisture estimation because the probes cannot be adequately inserted into the live fuels, and most meters do not operate at high moisture contents.

f. Ways to estimate dead fuel moisture content.

(1) Calculation.

(a) Fine fuels. The moisture content of fine dead woody fuels can be estimated with several different computation models. All models use inputs which describe the environment in which the fuel is located, temperature, relative humidity, slope, and time of year. The most simple but marginally accurate calculation method is available in tabular form in the course materials for S-390, Intermediate Fire Behavior, and S-590, Fire Behavior Analyst. A more accurate estimate can be made using the fine fuel moisture model (MOISTURE) in the BEHAVE system. (See XII.C.1.B.)

(b) Large diameter downed fuels. There is a regionally specific model that accurately predicts the moisture content for large diameter dead woody fuels, the ADJ-Th (Adjusted Thousand Hour) model developed by Ottmar and Sandberg (1985). This model applies to 3 to 9-inch diameter Douglas-fir and western hemlock logging slash in western Washington and Oregon.

(2) Fuel sticks. A standard set of fuel moisture indicator sticks consists of four, 1/2 inch diameter ponderosa pine sapwood dowels spaced one-quarter inch apart on two 3/16-inch- diameter hardwood pins. They do not measure any specific fuel but rather "measure the net effect of climatic factors affecting flammability" (Davis 1959). When completely dry, the sticks weigh 100 grams. Their moisture content can be obtained by weighing them, using any of several types of commonly available scales. Procedures for use of fuel sticks are described in detail in Finklin and Fischer (1990).

Fuel sticks have important limitations. The differing density of the wood of which the sticks are made can cause dowels made from the same board to give different fuel moisture values when exposed to the same environment. Response characteristics of the sticks can change significantly with continued exposure and wood aging. A fuel stick should be discarded after one season's use, and more often if rapid weathering or checking has occurred. A fuel stick must be exposed at least five days before moisture readings will be accurate. Because of the variation in fine fuel moisture content caused by microsite differences, use of only one set of fuel sticks to represent moisture conditions for a prescribed fire may give a very inaccurate estimate.

E. Summary

Fuels are an integral part of most wildlands. At some time after death, or while still alive, all vegetation becomes potential fuel. The single most important factor controlling the flammability and consumption of fuels is their moisture content. The moisture content of dead wildland fuels is regulated by environmental factors, while that of living plants is largely controlled by physiological processes. Other fuel properties can also affect the degree of consumption. All direct effects of fire result from the characteristics of the heat regime of the fire, which is controlled by the manner in which fuels burn. Management of fuels is important because by doing so, the heat regime of a fire is also regulated.

1. 0.63 approximates the value 1 minus 1/e, where e is the base for natural logarithms (Schroeder and Buck 1970). This value is used to describe fuel moisture relationships because the shape of the drying and wetting curve as a function of time is approximately logarithmic.

2. The presence of unweathered organic coatings that limited vapor movement in and out of the most recently cast needle litter was another likely cause of the slow moisture response (Hartford and Rothermel 1991).

3. Conifers of the Larix genera (larches and tamarack) have deciduous needles, and their moisture content will not be discussed here.

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CHAPTER IV - AIR QUALITY

By Larry Mahaffey and Melanie Miller

A. Introduction

Wildland fires produce smoke, an air pollutant. Smoke that is a result of human activities is subject to legal restrictions imposed by state, Federal, and local governments. Prescribed fire is a planned event, and Federal land managers have a mandate to manage prescribed fire smoke. Land managers must have a clear understanding of the regulations and processes that must be complied with to manage smoke. The liability for downwind effects is the responsibility of the prescribed fire managers who produced the smoke.

The National Environmental Policy Act (NEPA) is the law that establishes fundamental environmental policy for the U.S. and provides the process for considering the full range of impacts in planning land use activities. Agencies have the responsibility to disclose possible air pollution impacts from land management projects. The Federal land manager is required to conduct NEPA analysis if the project includes a "significant" amount of burning, may have impacts on sensitive vistas or visibility, or is located near a public roadway.

The 1977 Clean Air Act (CAA) mandates the protection of human health and the prevention of significant deterioration of air quality, and establishes acceptable levels of emissions. States are charged with the responsibility for protecting air quality. States write State Implementation Plans (SIPs) to interpret and enforce the Clean Air Act, including the identification of Designated Areas (DA), principal population centers or other areas requiring protection of air quality.

Designated Class I Areas include specific National Parks, Wilderness Areas, and certain Indian reservations. A goal for Federal Class I Areas is to prevent any future impairment of visibility and remedy any existing impairment of visibility that results from human-caused air pollution. The 1990 Amendments to the Clean Air Act specify that individual States must consider smoke from wildland fires in their SIPs. Requirements for prescribed fires can be established by States in the SIP that are more stringent than those in the CAA.

Anyone using prescribed fire must consider smoke management. Smoke management requirements and procedures vary because of the different amounts of fuel burned, fuel type, topography, meteorology, and presence of smoke sensitive areas (Mathews et al. 1985). The following questions can help a manager determine the level of smoke management that is needed, and whether an increased emphasis on smoke management is required.

1. Is the public informed of agency resource objectives for using prescribed fire?

2. Is the amount of acreage treated with prescribed fire predicted to change significantly?

3. Does topography or meteorology cause poor smoke dispersion?

4. Will prescribed fires cause or contribute to increased levels of air pollutants?

5. Will smoke from prescribed fires result in public health and safety problems or complaints?

6. Are Best Available Control Measures (BACM) used to manage prescribed fire smoke?

The effects of smoke on the airshed and the public and the opportunities to reduce these impacts will be discussed in this chapter. Managers have the responsibility to do the best job possible to control and mitigate the impacts of smoke that result from their actions or treatments.

B. Principles and Processes of Fire Effects

1. Combustion Process.

a. Chemistry of combustion. The following is a summary of information provided by Byram (1959). Wood is a chemically complex substance, composed primarily of cellulose and lignin, of which carbon, hydrogen, and oxygen are the primary constituents. When wood burns in a completely efficient manner, it combines with atmospheric oxygen, and produces water, carbon dioxide, and energy. Some of the water that results from combustion is evaporated from the fuel, but a larger proportion is a product of the chemical reaction.

b. Phases of combustion. The four phases of the combustion process are described in Chapter II.B.2, this Guide. Some strategies for smoke management rely on manipulation of the amount of fuel consumed in each combustion phase. The types of emissions and factors regulating their production will be discussed with respect to the phases of combustion. For a more complete discussion of the phases of combustion, see Sandberg et al. (1978).

(1) Pre-ignition phase. During the pre-ignition phase, gases, vapors, tars and charcoal are produced. The proportions and amounts vary widely according to the conditions under which pyrolysis occurs. If rapid heating occurs during pyrolysis, less charcoal, a lot of tar, and highly flammable gases are produced. Slow heating during the pyrolysis process results in the production of more charcoal, little tar, and lower amounts of flammable gases (Sandberg et al. 1978).

(2) Flaming phase. The following is from Ryan and MacMahon (1976 in Sandberg et al. 1978). The principal chemical by-products of flaming combustion are carbon dioxide and water. However, some pyrolyzed substances cool and condense without passing through the flaming zone. Other substances are only partially oxidized as they pass through the flames, and many combustion by products are produced. Low molecular weight organic compounds may remain as gases and are dispersed by wind. Tar droplets and particles of soot result from the cooling and condensation of compounds with higher molecular weights. Visible smoke consists of mostly tar, soot, and water vapor.

(3) Smoldering phase. The lower temperatures of the smoldering phase allow some gases to condense as visible smoke. Smoldering fires produce at least twice the emissions of flaming fires. Heat release is inadequate to loft the smoke as a convection column, so smoke stays near the ground and may persist in relatively high concentrations. Most of the smoke produced consists of tar droplets less than one micron in size (Johansen et al. 1985).

(4) Glowing phase. During the glowing phase, combustion is fairly efficient. Carbon monoxide and carbon dioxide are released, but no visible smoke is formed (ibid.).

c. Combustion efficiency. If combustion of fuels in wildland fires was 100 percent efficient, the burning of one ton of wood would release 3,670 pounds (1,665 kilograms) of carbon dioxide and 1,080 pounds (490 kilograms) of water (Sandberg and Dost 1990). All of the carbon in the fuel would oxidize to carbon dioxide. However, the combustion of fuels in wildland fires is not a completely efficient process. The most important reason for incomplete combustion is that wind cannot deliver enough oxygen to the combustion zone to mix efficiently with all of the flammable gases produced (Ryan and McMahon 1976 in Sandberg et al. 1978). There are differences between heading fires and backing fires in the proportion of time spent in the different combustion phases listed above (ibid.).

(1) Heading fires. A heading fire is one in which the flaming front moves ahead rapidly. The fire may be pushed by the wind, move upslope, or be influenced by both factors. These fires burn with relatively high fireline intensity, moving quickly from one fuel element to another. The main combustion zone moves before most fuel elements are completely consumed by fire. The flames continue ahead, leaving behind a large area of smoldering fuel (ibid.).

(2) Backing fires. A backing fire burns into the wind or downslope. Because the flames move more slowly, a higher proportion of fuel is consumed in the flaming zone of the fire, leaving less fuel to smolder after the flaming front has passed.

(3) Smoke production. For a given fuel bed and set of burning conditions, a heading fire causes more total smoke production than a backing fire. A heading fire generally results in more fuel consumed in the smoldering phase of combustion than does a backing fire, and smoldering fuels produce more smoke than fuels burned in flames. A backing fire is a more efficient fire because more fuel is consumed in flaming combustion, and less smoke production results.

d. Fuel properties that affect smoke production. Fuel properties that affect smoke production are those that influence the phase of combustion in which fuel consumption occurs, and the total amount of fuel consumed. These factors are discussed more completely in Chapter III. Fuels.

(1) Fuel particle size, arrangement, and continuity. The smaller the size of the fuel particle, the more quickly it can ignite and be consumed. The arrangement of fuel particles affects the amount of oxygen that reaches them. More tightly packed fuel, such as a bed of juniper or spruce needles, burns less efficiently, and produces more smoke than a loosely packed fuel bed, such as one of ponderosa pine needles. Fuel continuity is a factor because if fuel particles are too widely spaced, sustained ignitions cannot occur; flames are unable to ignite adjacent fuels.

(2) Fuel loading. A site with large amounts of fuel can generate more smoke than a site with little fuel. The size class distribution of the fuel is also important, because the proportion of fuel in each size class affects the proportion that may be consumed in flaming versus smoldering combustion. Smaller diameter fuels, such as loosely packed grass litter, fine branchwood, and live moss and lichens burn almost entirely in flames with little residual smoldering. In contrast, large diameter downed woody fuels such as those found in logging slash are rarely consumed in flaming combustion, and thus have higher potential to emit large amounts of residual smoke.

(3) Fuel moisture. The moisture content of the different size classes of fuel affects smoke production because it influences fuel availability and combustion temperatures. Extremely dry fuels burn rapidly and completely, while wet fuels burn slowly or not at all. Any moisture released from the fuels absorbs some heat energy from the fire, limiting combustion temperatures (Ryan and McMahon in Sandberg 1978). If larger size classes of fuels have a high moisture content, most or all of the heat released by flames will be expended evaporating water, and little consumption of large diameter fuels occurs. Fuel moisture, its role in combustion, and its relationship to past and present atmospheric conditions, is discussed more completely in Chapter III, Fuels.

2. Emissions. Emission products from fires vary greatly, depending upon the type of fuel, fireline intensity, fuel moisture, wind, and temperature of the fire.

a. Combustion products. Hundreds of different compounds are emitted in the smoke from wildland fires. More than 90 percent of the mass of smoke emitted from wildland fires consists of carbon dioxide and water. Carbon in the fuel is also converted to particulate matter, carbon monoxide, aldehydes, and hydrocarbons, as well as complex organic materials (Johansen et al. 1985). Nitrogen oxides and hydrocarbons produced by the fire can react together in the presence of sunlight and produce ozone and organic oxidants. Ozone production occurs in the top of a smoke plumes where there is more light, and in downwind areas where smoke is less dense (Sandberg and Dost 1990).

Because most of the adverse effects of smoke are related to the amount of smoke produced, fire managers need to know how much smoke is generated. The answer can be estimated from two numerical expressions: emission factor and emission rate.

b. Emission factors. An emission factor is the mass of contaminant emitted to the atmosphere by the burning of a specific mass of fuel, and is expressed in pounds per ton in the English system or grams per kilogram as the metric equivalent (Johansen et al. 1985). An emission factor can be calculated for a single fire, or a single combustion stage of one fire, or it can be a statistical average for a geographical area or a set of similar fires (Sandberg and Dost 1990).

(1) Carbon dioxide. The carbon dioxide emission factor for prescribed fires ranges from 2,200 to 3,500 pounds per ton of fuel consumed (1098 to 1747 g/kg) (Sandberg and Dost 1990). The combination of carbon in the fuel with atmospheric oxygen during combustion results in the production of a greater weight of carbon dioxide than the original weight of the fuel. Carbon dioxide is a "greenhouse gas", i.e., it may have an effect on the global radiation budget and may be a factor in potential global climate change. However, carbon dioxide is also released when wood and other organic matter decays. Logging removes forest fuels from sites and can reduce the amount of carbon dioxide that would be released if the site burned. Fire suppression is not an effective way to mitigate this carbon dioxide release from many wildland fuels because in most cases, suppression only postpones burning. Decomposition by fire has occurred for millions of years in most of the vegetation types in western and northern North America. In the absence of fire, fuels tend to accumulate, ignition eventually occurs, and more carbon dioxide may be released than would have occurred under a natural fire regime. More fuel may be present, and fuel consumption may be more complete.

(2) Particulate matter. Particulate matter is the most important category of pollutants from wildland fire, because it reduces visibility and can absorb and transmit harmful gases. Particles vary in size and chemical composition, depending upon fireline intensity and the character of the fuels. Proportionately larger particles are produced by fires of higher fireline intensity (longer flames) than are found in low intensity and smoldering combustion fires (Ward and Hardy 1986 in USDA Forest Service and Johns Hopkins University 1989). Particulate matter emission factors for forest fuel types range from 4 to 180 pounds per ton (2 to 90 g/kg). For prescribed burning of logging slash, particulate production ranges from 18 to 50 pounds per ton (9 to 25 g/kg) for broadcast burning and 14 to 30 pounds per ton (7 to 15 g/kg) for piled slash. The amount of particulate released when burning sagebrush/grass fuel types averages 45 pounds per ton (22.5 g/kg), mixed chaparral ranges from 24 to 30 pounds per ton (12 to 15 g/kg), and emission factors for pinyon-juniper (slashed) range from 22 to 35 pounds per ton (11 to 17.5 g/kg) (Hardy 1990). The exact amount depends on the fuel type, the fuel arrangement, and the manner of combustion.

Emission factors for particulate matter less than 2.5 microns in diameter ($PM_{2.5}$) range from 9 to 32 pounds per ton (4.5 to 16 g/kg) for prescribed fires in the Pacific Northwest, averaging about 22 pounds per ton (11 g/kg). Emission factors are highest during the inefficient smoldering combustion stage and lowest during flaming combustion.

The amount of smoke produced depends on the total amount of fuel consumed. For example, even though the emission factor for sagebrush is higher than that for chaparral or pinyon-juniper, total smoke production from burning sagebrush is often lower because the total amount of fuel on a sagebrush site is generally less than on a chaparral or pinyon-juniper dominated site.

(3) Other emissions. Emission factors are available for other products of combustion such as the invisible gases. Emission factors for carbon monoxide range from 70 pounds per ton (35 g/kg) during flaming combustion to 800 pounds per ton (399 g/kg) for some smoldering fires.

Volatile organic compounds are a diverse class of substances containing hydrogen, carbon, and other elements such as oxygen. They include methane, polynuclear aromatic hydrocarbons (PAH's), and aldehydes and related substances. PAH's are not free in the environment as vapor, but are incorporated in fine particulates that are respirable. Methane and aldehydes are emitted as gases. Emissions for volatile organics vary from 4 to 50 pounds per ton (2 to 25 g/kg) of fuel burned, about half of which is commonly methane.

c. Emission rate. An emission rate is the amount of smoke produced per unit of time (pounds/minute or grams/second). The portion of the total amount of combustible fuel that a fire will consume for a given set of conditions is called available fuel, and is usually measured in tons per acre (kg/sq m) for forest fuels, and pounds per acre (kg/ha) for rangeland fuels. The land manager can make better estimates of emission rates from a prescribed burn if the amount of fuel consumed in each combustion phase is known. (See B.1.b.)

Fuel consumption rates are expressed as area burned per unit of time: acres per minute. Combustion rates can be calculated whether line-type ignition is used for backing or heading fires or area-type ignition is used in natural or activity fuels.

In order to estimate the emission rate, the following variables are required: available fuel (tons/acre), the combustion rate (acres/minute), and the emission factor (pounds/ton). The emission rate (pounds/minute) can be calculated by the following equation:

Emission Rate = Available Fuel x Combustion Rate x Emission Factor

The emission rate is used as an input to models that predict air pollutant concentrations. Such models can be used to assess the impact of smoke on visibility sensitive areas such as highways, cities, airports, and parks (Johansen et al. 1985).

3. Human Health Risk from Smoke. There is a growing awareness that smoke from wildland fires can expose individuals and populations to hazardous air pollutants. Concern is increasing over the risk to firefighters and the general public from exposure to toxins, irritants, and known carcinogens in smoke. A rigorous risk assessment is needed to address this increasingly sensitive issue. Although there is a low probability that public health is at risk, fireline workers are more likely to be harmed. Firefighters can be exposed to high levels of lung toxins such as aldehydes, acids, and particulates; to carcinogens such as polycyclic aromatic hydrocarbons, formaldehyde and benzene; and to carbon monoxide. These exposures may be at high levels for short periods, or at low levels for weeks at a time. The amount of some hazardous components of smoke, such as formaldehyde and respirable particulate matter, is well correlated to the amount of carbon monoxide (Reinhardt 1989). Relatively inexpensive devices for measurement of carbon monoxide (CO dosimeters) may provide a practical means to help recognize and prevent exposure of firefighters to dangerous levels of smoke.

The most likely effects of smoke on health are the aggravation of existing diseases or increased susceptibility to infection. Those most susceptible to exposure to air toxins include very young children and individuals with chronic lung disease or coronary heart disease. The effects of smoke on human health are discussed in detail in Sandberg and Dost (1990), Dost (1991), and the comprehensive study plan prepared by the USDA Forest Service and Johns Hopkins University (1989). The following discussion is summarized from these sources.

a. Criteria pollutants. The National Ambient Air Quality Standards are a set of goals established by the Environmental Protection Agency for acceptable levels of six air pollutants that are potentially harmful to public health. These criteria pollutants are particulate matter, carbon monoxide, sulfur dioxide, nitrogen dioxide, ozone, and lead.

(1) Particulate matter. The size class distribution of particles produced by forest fires is bimodal. Most of the particles have an average diameter of either 0.3 micrometers or greater than 10 micrometers (USDA Forest Service and Johns Hopkins University 1989). The proportion of particles in wood smoke less than 2.5 micrometers in diameter ranges from 50 to 90 percent (Sandberg and Dost 1990). Particles less than about 10 micrometers in diameter are able to traverse the upper airways (nose and mouth) and enter the lower airways starting with the trachea (Raabe 1984). As the particle size decreases below 10 micrometers, increasing proportions of particles are able to enter the trachea, and penetrate to the deeper parts of the airways prior to deposition. It should also be noted that once such particles reach the lower airways, it is likely that they will deposit on surfaces in the deepest parts of the lungs, the "pulmonary" zone--that part of the respiratory tract most sensitive to chemical injury (Morgan

1989 in Sandberg and Dost 1990).

National air quality standards assume that the components of particulate matter are essentially the same, regardless of location and source. However, the constituents of particulate matter vary widely. The particulate emitted by burning of vegetation has a much different composition and effect than that present in urban areas (Dost 1990). Particulate matter from vegetation fires consists mainly of condensed organic compounds. In urban areas, particulates contain compounds that rarely occur in rural vegetation smoke, such as masonry dust, fly ash, and asbestos. Also, other compounds present in urban air lead to a variety of chemical reaction products not likely to be found in association with wildland smoke. Smoke from wildland fuels is not as environmentally damaging as urban smoke.

(2) Carbon monoxide. Carbon monoxide is a product of combustion that is rapidly diluted at short distances from a fire and therefore poses little or no risk to community health (Sandberg and Dost 1990). However, carbon monoxide can be present at high enough levels near a fire to pose a hazard to firefighters, depending upon the concentration, duration, and level of activity of the firefighters at the time of exposure. Carbon monoxide is a chemical asphyxiant that interferes with oxygen transport in blood. Pilots exposed to carbon monoxide have developed headaches, fatigue, decreased concentration and impaired judgement. Data also suggest that long-term exposure to low levels of carbon monoxide produce accelerated arteriosclerosis, increasing the risk of cardiovascular diseases such as heart attack and stroke (USDA Forest Service and Johns Hopkins University 1989).

(3) Oxides of sulfur and nitrogen. Because forest fuels contain minute amounts of sulfur and somewhat higher levels of nitrogen, it is expected that these criteria pollutants are formed when wildland fuel is burned. Increased levels of oxides of sulfur have never been measured near wildland fires. Some oxides of nitrogen form, but the amount produced by forest burning is not significant enough to be of concern (Sandberg and Dost 1990).

(4) Lead. While a serious problem in urban pollution, lead is not a natural constituent of smoke from wildland fuels. It is assumed that lead may be only a minor component of wildland fire smoke when it has been deposited onto fuels from atmospheric sources, such as contaminated urban air that has moved into wildland areas (Ward and

Hardy 1986 in Sandberg and Dost 1990).

(5) Ozone. Much of the following discussion is summarized from Dost (1990). At lower altitudes, ozone is a common component of air, but at low enough levels to have an insignificant impact on human health. At high concentrations, ozone is a respiratory irritant, and can have a significant impact on individuals who already have serious respiratory impairment. The atmospheric chemistry of ozone formation is quite complex. It can form in the presence of atmospheric hydrocarbons generated in large quantities by the combustion of vegetation. The photochemical reactions that create ozone occur in areas of a smoke column that are penetrated by ultraviolet wavelengths of sunlight, particularly in the upper part of the plume. Ozone is therefore not likely to be a pollutant of concern to people in the immediate vicinity of a fire, although fire crews working at high elevations may find increased levels of this substance (USDA Forest Service and Johns Hopkins University 1989). Ozone may pose a problem in downwind areas affected by smoke, particularly in urban areas where ozone concentrations from other sources may already be high.

b. Non-criteria pollutants. Ambient standards have not been specifically identified by the Environmental Protection Agency for all components of smoke, although some of these substances can have negative effects on human health. The following discussion, taken largely from Dost (1990), describes the two groups of volatile organic compounds most likely to impact human health.

(1) Aldehydes. Aldehydes are classed as irritants, and some are potentially carcinogenic. The two aldehyde compounds found in smoke that are most likely to pose health problems are formaldehyde and acrolein.

(a) Formaldehyde. Formaldehyde is a very common atmospheric contaminant, found in association with building materials, textiles, preservatives, and medical activities. It has been measured as a by-product of burning wood, although little is known about its production in wildland fires. Formaldehyde is probably the most abundantly produced aldehyde, and is likely responsible for nose, throat and eye irritation in firefighters exposed to smoke. At higher concentrations, it may cause a reflexive decrease in breathing rates. Formaldehyde may not only thus contribute to mucosal irritation commonly experienced by firefighters, it also may interfere with their ability to obtain adequate oxygen at times when energy is most needed. Formaldehyde is rapidly transformed in the body to formic acid, a known toxin with a very slow removal rate. Formaldehyde also may be present in decreasing amounts in the smoke plume downwind, being slowly removed by chemical reactions.

(b) Acrolein. Acrolein is formed by few natural processes other than combustion. It has been measured in emissions from fireplace smoke, and studies suggest that greater amounts are produced in inefficient fires. Acrolein is a more potent irritant than formaldehyde. It has similar effects as formaldehyde to the respiratory system, but may also have severe toxic effects on cells. Individuals exposed to acrolein may have a decreased ability to repel respiratory infections. Acrolein is degraded by sunlight, and it is assumed that it slowly dilutes downwind with other components of smoke in the plume. If initial concentration levels are high, acrolein could be a significant irritant at a considerable distance from its source.

(2) Polynuclear aromatic hydrocarbons (PAH). Polynuclear aromatic hydrocarbons are a class of products that have been detected in wood smoke. These benzene containing compounds are incorporated as fine particles that are respirable. Some PAH compounds have carcinogenic properties. The likely risk to the general public of developing cancer because of exposure to these chemicals from prescribed fire is very small because of the rapid dilution of these products in the smoke plume (Sandberg and Dost 1990). However, very little is known about potential effects on firefighters from these compounds due to a lack of research on their production in wildland fires.

C. Resource Management Considerations

1. Control Strategies. When wildland fires occur, managers must consider the impacts on air quality and mitigate adverse effects whenever possible. Wildfire is evaluated through the Escaped Fire Situation Analysis, and prescribed fire through the Environmental Assessment and Prescribed Fire Planning process. There are strategic and tactical measures that can limit the amounts and mitigate the impacts of smoke from fires. The mitigation of adverse impacts can be accomplished through the selection and implementation of an appropriate control strategy. Managers can use these strategies to allow the burn to take place and yet reduce the risk of adverse effects of smoke. Clear resource management objectives and careful monitoring and evaluation of smoke impacts are keys to successful smoke control.

Managing smoke from wild or prescribed fires requires a daily prediction of smoke accumulations and whether they will reach unacceptable levels. Choice of suppression strategies and tactics must include a consideration of smoke effects on safety and visibility.

a. Avoidance. Avoidance is a strategy that considers meteorological conditions when scheduling burns to avoid incursions of smoke into sensitive areas. Burning should occur on days when weather conditions allow the transport of smoke away from populated areas. Smoke may not be such a limiting factor in sparsely populated areas, but any downwind effects should be considered when burning. The wind direction during both the active burning period (flaming stage) and the smoldering period must be considered. At night, downslope winds can carry smoke toward smoke sensitive areas, or allow valley bottoms to fill with smoke. Residual smoke emitted during the smoldering stage is especially critical.

b. Dilution. The dilution strategy controls the amount of emissions or schedules the rate of burning to limit the concentration of smoke in sensitive areas. The concentration of smoke can be reduced by diluting it through a greater volume of air, either by scheduling during good dispersion periods or burning at slower rates (burning narrow strips or smaller areas). However, burning at a slower rate may mean that burning continues into the late afternoon or evening when atmospheric conditions may become more stable. Burn when weather systems are unstable, but not at extremes of instability. The time of day at which ignition occurs is also important. Consider early morning ignitions to take advantage of weather conditions where improved mixing will occur as atmospheric heating takes place. Avoid days with low morning transport wind speed, less than 4 miles per hour (6.5 km/hr). Use firing methods to rapidly build a smoke column to vent smoke up to the transport wind and larger volumes of air. Using mass-ignition or rapid ignition will loft the column up and away from the unit, allowing for better dispersion and reduced emissions during the smoldering phase. Generally, a burn early in the day encounters improving ventilation; an evening burn encounters deteriorating ventilation.

c. Emission reduction. Emission reduction is an effective control strategy for decreasing the amount of regional haze and avoiding smoke intrusions into Designated Areas (DA's) (Sandberg et al. 1985). It reduces the smoke output per unit area, and is a concept applicable in both forest and rangeland areas. Most emission reduction techniques are based on limiting the consumption of larger fuels and soil organic layers. Large fuel consumption can be reduced by physically removing or scattering the larger fuels or burning when the larger fuels and duff are too wet to carry fire. Burning when the larger fuels or duff are wet will produce fewer emissions and allow rapid extinguishment of the fire. When windrowing and piling debris, allow fuels to dry before piling, and avoid mixing dirt into the pile. Emissions can also be reduced by use of a backing fire that results in more fuel consumption in the flaming stage, producing less smoke (Sandberg and Peterson 1987).

2. Techniques to Minimize Smoke Production and Impacts. Some smoke management techniques have application to both wildfire and prescribed fire situations, while others apply specifically to prescribed fire management.

a. All fire situations.

(1) Be sure that each burning operation has clear and concise management objectives that consider the impacts of smoke.

(2) Ensure that burn prescriptions and ignition plans provide for optimal smoke dispersion for the specific circumstances of the fire.

(3) Use the best weather data available to ensure adequate smoke dispersal. This includes obtaining spot weather and transport wind forecasts from the National Weather Service, taking weather at the burn site for several days prior to ignition, and validating the fire prescription and spot forecast with onsite weather observations. Wind speed and direction over the area can be checked by release of a helium balloon, or by observing the smoke from a test fire.

(4) Burn when conditions allow rapid dispersion. The atmosphere should be unstable so smoke will rise and dissipate, but not so unstable that control problems result.

(5) Burn when fuel moistures are higher and consume only those fuels that are specified in the treatment objectives. Higher duff moisture shortens the smoldering phase, thereby reducing residual smoke and particulate production.

(6) Mass ignition allows burning to occur with higher fuel moistures.

Higher temperatures generated by mass fire cause smoke to rise to a greater height above terrain than if a line ignition is used.

(7) Use a backing fire. The slow rate of spread and long residence time result in a higher fraction of fuel consumption in the flaming stage of combustion rather than in the smoldering stage. Since total smoke production per unit of fuel burned is considerably less during flaming combustion, backing fires favor lower total smoke production.

(8) The volume of smoke in a geographic area must be considered when making management decisions about prescribed burns, prescribed natural fires, or wildfires.

b. Prescribed fire.

(1) Burn other than in the "traditional" late summer and fall season. The impact on the air resource can be spread over a longer period, thereby reducing the possibility of a heavy smoke load on a particular day. Be careful of night burns because predicting smoke drift is more difficult, although night burning can be successful if properly planned and implemented.

(2) Burn fuel concentrations, piles, landings, and jackpots outside of the prescribed burning season. This increases the number of units that can be burned without overloading the airshed on days with good dispersal conditions.

(3) Public criticism of a burn program can be decreased by limiting its impact on recreational users. Avoid burning on days when smoke may affect Class I Areas and heavily visited recreational areas, or on holidays when many visitors may be using public lands.

(4) Using prescribed natural fire requires close monitoring of fuel loadings, fuel moistures, normal weather patterns, and down wind receptors in the area that may be affected by smoke drift.

(5) For prescribed natural fires, daily certification that the fire remains in prescription must include an assessment of smoke dispersal.

3. Participation in State and Local Smoke Management Programs. State and some local air quality agencies have mandatory smoke management programs. Programs are tailored to the needs of local and regional prescribed fire managers, while working to minimize adverse impacts of smoke.

a. Comply with air pollution and smoke management regulations. Know the regulations for your State and local area when developing the prescription. Details on State and local laws and regulations can be obtained from agency fire management or air quality staff.

b. Be pro-active in protecting air quality. Take part in the development (or update) of the State Implementation Plan that contains rules that govern prescribed burning. Working with State and local air quality agencies provides an opportunity for field input and some control over the future of prescribed fire.

D. Methods to Monitor Fire Effects

This section suggests methods for monitoring smoke effects that are practical for management purposes. Although there may be few State regulations that require monitoring of prescribed fire smoke, there are stewardship principles and ethical reasons that make monitoring a compelling aspect of a smoke management policy. As the first step, managers must develop and maintain an awareness of air quality monitoring techniques. Monitoring allows the evaluation of program adequacy and the effectiveness of communication with local air quality personnel. Implementation of air quality monitoring does not require having an elaborate array of monitoring instruments or hiring a monitoring contractor to evaluate fires.

While no readily available operational smoke monitoring techniques accurately predict the effect of a specific fire on air quality, understanding principles of air quality monitoring can result in better smoke management decisions. Some states are currently charging fees for burning, such as one fee to register each acre and an additional fee to burn the acre. This money is used to support the smoke management program and provide monitoring services for agencies doing prescribed burning. Local fire management officers should determine the proper level of monitoring and incorporate it into the burn plan. They should develop an objective method to monitor and evaluate the effectiveness of their smoke management efforts. Monitoring practices can range from simple to very complex programs as determined by managers or by the states. Agency specific guidance that identifies smoke management monitoring techniques and frequencies is provided in USDI-NPS (1992). The following are some practical procedures for monitoring and modelling smoke.

1. Visual Techniques.

a. Visual estimation. Visual estimation is the most common smoke monitoring method in use. Although most visual methods are subjective and limited, they are still very useful. When burning near smoke sensitive areas, a spotter on a hill away from the fire can watch where the smoke goes and relay information to the Burn Boss.

b. Aircraft tracking. Aircraft tracking of smoke plumes can be used to verify the source and trajectory of the smoke. It is used by some regulatory agencies to detect violations of air quality/smoke management regulations. This procedure provides a means to observe the loading of the airshed and to determine if additional burning should be limited.

2. Instrumentation.

a. Nephelometer. A nephelometer is an electronic device that measures the amount of particulate in a sample of air. This optical device measures the amount of light reflected from particles in the enclosed sample space. A nephelometer can be useful for safety monitoring, such as by measuring the amount of smoke on a highway. The machine could be programmed to flash lights as a warning when visibility is poor.

b. Filter sampler. Filter samplers draw a known volume of air through a filter. The filter is weighed before and after the sampling period, and the weight of particulate per volume of air can be calculated.

3. Computer Models.

a. SASEM. The Simple Approach Smoke Estimation Model (SASEM) is a screening model developed by the Bureau of Land Management and approved by the States of Wyoming and Arizona for estimating smoke impacts from prescribed fires. This model calculates emissions, and uses the emission figure to calculate down-wind concentrations of particulates. Estimated particulate loadings are compared quantitatively against ambient air quality standards to see if standards may be exceeded. (See Chapter XII.E.1., this Guide.)

b. TAPAS. The Topographic Air Pollution Analysis System (TAPAS) is a system of models for predicting the dispersion of air pollution over flat or mountainous terrain. Data on topography, wind speed, and direction are used to model plume direction and speed of smoke. Documentation and more information is available from the air quality staff, National Biological Survey, Environmental Science and Technology Center, Fort Collins, Colorado.

c. TSARS. The Tiered Smoke/Air Resources System (TSARS) is a group of computer programs that allows smoke management to be performed in a series of increasingly advanced levels of proficiency (Riebau et al. 1991). Tools with varying degrees of sophistication are available to model smoke production from wildland fires, producing results with different degrees of resolution. The level of analysis conducted can be matched to the complexity of the problem, or the expertise of the person using the model. Simple to use tools which produce easily interpreted results can be used at field levels, while more central offices in an agency would have access to more elaborate techniques. The components of the TSARS system were all derived from existing models, but have been modified for fires as the emission source and to have a consistent appearance to the user. See Chapter XII.E.2. for a more complete discussion of TSARS.

d. PUFF. PUFF is a plume trajectory model for multiple fires being developed for the Pacific Northwest. PUFF uses input on emission production, and models smoke dispersion for a specific grid of atmospheric temperature and pressure. Atmospheric conditions are derived from a National Weather Service model. These data can be input automatically from a computer that is operated by a private contractor. For more information, contact the U. S. Forest Service, Global Environmental Protection Project, Forestry Sciences Laboratory, Seattle, Washington.

E. Summary

The effects of smoke on health, air quality, and regional haze is very important to all land managers. They must recognize the need to manage smoke from wildland fires using the Best Available Control Measures. Every manager must determine the level of smoke management necessary to provide the least impact on the public, both in terms of health and visibility. The effects of smoke on firefighters also must be considered when managing wildland fires. If federal agencies do not take a rational, voluntary approach to smoke management, a mandatory approach may be provided that makes it more difficult to meet resource management goals and objectives.

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CHAPTER V - SOILS, WATER, AND WATERSHEDS

By Dr. Bob Clark

A. Introduction

Fire, either wild or prescribed, may have a wide range of effects on the soils, water, and watershed resources of forestlands, shrublands, grasslands, and wetlands. The wide range of effects is due to the inherent preburn variability in these resources, and to fire behavior characteristics, season of burning, and prefire and postfire environmental conditions such as timing, amount, and duration of rainfall. Further, effects of fire on some soils and watersheds are poorly documented and poorly understood. Thus, generalization about effects of fire on these resources is likely to be more risky and overstated than with any other managed resource.

This chapter, in keeping with the theme of this Guide, will discuss principles and processes that contribute to the effects of fire on soils, water, and watersheds. Where applicable, the opportunity to manage or influence these processes will be discussed. Also, methods for monitoring the effects of fire on these resources will be described, recognizing fire management investigation and monitoring needs for management purposes are not normally required to the degree and precision that are required at the research level. Finally, other factors are critical to "correct" prediction and evaluation of fire effects on soils and water; Chapter III.D. in this Guide is particularly relevant.

Excellent texts that provide basic information on soil taxonomy, properties, physics, and hydrology include those written by Soil Survey Staff (1975), Pritchett (1979), Hanks and Ashcroft (1980), and Branson et al. (1981). The ecology of the soil nitrogen cycle is well described in (Sprent 1987). Soil descriptions used in this chapter follow the nomenclature of the Soil Survey Staff (1975) unless otherwise noted. References to litter layers are described in Youngberg (1981). Important terms are identified in the Glossary.

B. Principles and Processes of Fire Effects

1. Soils. Most information about the effects of fire on soils is from forested land

and chaparral; also, much information is predicated on the effects of wildfire, not prescribed fire. By extrapolation, this situation has frequently led to the conclusion that fire is always detrimental to soils, including shrubland and grassland soils. However, in a history of fire research, Schiff (1962) indicated that researchers began documenting about five decades ago that in addition to negative effects, fire occasionally had beneficial effects on soil, and often had no measurable effect; further, the negative effects often were short-lived. These data are not meant to imply that the effects of fire are unimportant, because any negative effect, however small, can have substantial postfire consequences. The effects of fire on the soil resource are induced by soil heating, by removal of the protective cover of vegetation, litter and duff, or by the concentration of plant material substances in the soil. These effects are described in detail in Chandler et al. (1983), Wells et al. (1979), Wright and Bailey (1982), and other references listed in the Bibliography.

a. During the fire.

(1) Soil heating. The magnitude of the heat pulse into the soil depends on fuel loading, fuel moisture content, fuel distribution, rate of combustion, soil texture, soil moisture content, and other factors. The movement of heat into the soil is not only dependent upon the peak temperature reached, but even more so upon the length of time that the heat source is present. Because fuels are not evenly distributed around a site, a mosaic of soil heating occurs. The highest soil temperatures are associated with areas of greatest fuel consumption and the areas that have the longest duration of burning. In forested areas, high subsurface soil temperatures usually occur beneath fuel accumulations, with the highest temperatures most likely to be found in association with consumption of large piles of dry harvest residue or windthrow, or very thick duff layers. Because the pattern of soil heating varies significantly around a site, with differences in both the amount and duration of soil heating, a range of fire effects on soils can occur on one burned area.

Duff and soil moisture contents are critical regulators of subsurface heating. In a controlled soil heating experiment, the heat load into wet duff and mineral soil averaged 20 percent of the heat load that penetrated dry duff and mineral soil (Frandsen and Ryan 1986). Peak temperatures were more than 1000 F (538 C) greater where duff and soil were dry. DeBano (1977) estimated that about 8 percent of the heat generated in California chaparral fires is absorbed and transmitted downward into the soil. In general, "lightly burned" forests will cause maximum soil surface temperatures between 212 and 482 F (100 and 250 C) and the temperature 0.4 to 0.8 inches (1.0 and 2.0 centimeters) below the surface will not exceed 212 F (100 C) (Chandler et al. 1983). In "moderately burned" areas, surface temperatures are typically in the 572 to 752 F (300 to 400 C) range and may be between 392 and 572 F (200 and 300 C) at the 0.4 inch (1.0 centimeter) depth (Chandler et al. 1983). A "severely burned" area may result in surface temperatures approaching 1400 F (760 C) (Chandler et al. 1983).

In contrast, rangelands support considerably lighter fuel loadings and frequently result in fires of shorter duration that produce less subsurface heating. Rangeland fires typically result in soil surface temperatures from 212 to 730 F (100 to 388 C) although extremes to 1260 F (682 C) have been reported. The highest surface temperatures are probably associated with local accumulations of loosely arranged litter and intense winds created by the fire (Wright and Bailey 1982). 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 (DeBano 1979), is instrumental in eventual effects of fire on plants (see Chapter VI.B.1.c., and VI.B.2.c., this Guide) and in physical, chemical, and biological effects on soils.

Less is known about heat effects on wetland soils. Due to the high water content of wetland soils, penetration of heat generated by a surface fire can be significantly less than in mineral soils. Since many wetland soils are composed of significant amounts of organic materials, and organic matter has a lower thermal diffusivity than mineral soil, penetration of heat can be further reduced. However, organic soil layers can become dry enough to burn. Significant amounts of heat can be generated when organic soils burn, particularly in drought situations when the fire burns deeply into organic layers.

(2) Postfire temperature increases. Soil temperature may increase after a fire because of the removal of vegetative cover, consumption of fuels, thinning or removal of the litter and/or duff layer, and the enhanced "black body" thermal characteristics of charred material on the surface. This is of great significance in Alaska where permafrost (permanently frozen soil) is present. The soil layer above the permafrost thaws each summer, and is called the "active layer." Soil temperatures usually increase after a fire because fire removes the overstory vegetation, blackens the surface, and consumes some of the layer of moss, lichens, and semi-decomposed organic matter that insulated the soil from summer warmth. Soil temperatures were 9 to 11 F (5 to 6 C) greater at depths of 4 to 20 inches (10 to 51 centimeters) after fire in a black spruce/feathermoss stand in interior Alaska (Viereck and Foote 1979). Eight years after this fire, the depth of the active layer had increased from about 18 inches to 72 inches (46 to 183 centimeters) (Viereck and Schandelmeier 1980). The depth of the active layer eventually stabilizes, and then decreases to its original thickness. The length of time before this occurs depends upon the rate at which new vegetation grows and shades the soil surface, and how long it takes for a soil organic layer to develop that has the same insulating properties as the organic layer that was removed by the fire.

Under similar moisture regimes, warmer soils increase the rate of decomposition, and nutrient availability to postfire vegetation. Within physiological limits, higher soil temperatures also improve growing conditions for plants. In Alaska, deeper annual soil thawing increases the depth of soil available for rooting. This makes additional nutrients, especially nitrogen, available to plants, simply because they are not in frozen soils (Heilman 1966; 1968). Postfire vegetation productivity generally increases significantly after fire on permafrost sites (Viereck and Schandelmeier 1980), although the duration of this effect is undocumented.

b. Physical effects. Heating may cause changes in some physical properties of soils, including the loss or reduction of structure, reduction of porosity, and alteration of color. Most frequently, however, the important consequences include the reduction of organic matter, exhibition of increased hydrophobicity (nonwettability), and increased erosion due to the loss of protective plant and litter cover. Organic matter and hydrophobicity are discussed below under the heading "c. Direct chemical effects."

Erosion by wind (aeolian), water, or gravity often, but not always, increases following fire. The severity and duration of the accelerated erosion depend on several factors, including soil texture, slope, recovery time of protective cover, the amount of residual litter and duff, and postburn precipitation intensity. Raindrop splash, sheet and rill erosion, dry ravel, soil creep, and even mass wasting can occur. In extreme cases, such as steep, chaparral sites in the San Gabriel Mountains of southern California, erosion rates of more than 150 tons of debris per acre have been measured after wildfires (Krammes and Osborn 1969). It is reasonable to assume that hydrophobicity contributed to the extreme erosion rates reported from these areas (DeBano 1979). Extreme rates, up to 165 tons per acre, have also been reported following severe wildfires on timbered and chaparral sites with 40 to 80 percent slopes in Arizona (Wright and Bailey 1982). More commonly, erosion rates, even on steep slopes, range from about 23 to 52 tons per acre on granitic, sandstone, and shale-derived soils, and 7 to 10 tons per acre on limestone-derived soils (Wright and Bailey 1982). It is unclear from the literature how much of the soil movement is attributable to fire, because preburn soil movement or soil movement from unburned "control" areas is seldom reported.

Excessive erosion may not occur for several years after burning (Wright and Bailey 1982) because root systems of top-killed shrubs can maintain soil stability. Mass wasting apparently occurs when root systems begin to decay. If this occurs, it is reasonable to assume that rapid reestablishment of soil-stabilizing, deep-rooted shrubs (rather than shallow rooted grasses) is critical, especially on steep slopes. It has also been reported that coarse-textured soils are more erodible than fine-textured soils (Wright and Bailey 1982). This may explain why little soil movement occurs, even on steep slopes, following prescribed fires on sites with fine-textured Mollisol soils in Wyoming and elsewhere. In Alaska, however, fine textured permafrost soils tend to be much more erosive than coarse textured permafrost soils. Coarse textured soils usually have a low water content, while fine textured soils may contain as much as 50 percent ice. Postfire erosion on ice-rich permafrost soils occurs much more frequently where firelines have been constructed than on sites that have burned, because fires are seldom severe enough to completely remove the organic layer (Viereck and Schandelmeier 1980).

c. Direct chemical effects. Several chemical changes in soils may occur as a direct result of fire, including an increase in pH on some sites; the formation of water repellant soil layers, hydrophobicity, on some sites; and reduction in organic matter.

(1) Organic matter. The reduction of incorporated organic matter is critical if it occurs, on arid, semi-arid, and forested sites, because organic matter is a basic reservoir of the site nutrient (especially nitrogen) budget (Sprent 1987; Harvey et al. 1987). Organic matter helps regulate the hydrologic cycle and the carbon/nitrogen ratio, provides a site for nitrogen fixation by N-fixing bacteria, and maintains soil structure porosity and the cation exchange capacity. The amount of soil organic matter consumed by fire depends on soil moisture content, amount and duration of heating, and amount of organic matter available for combustion or distillation. For example, peat soils may burn extensively, whereas fire rarely affects most rangeland soils. Similarly, saturated soils rarely loose any organic matter whereas substantial losses may occur in dry soils.

(2) Hydrophobicity. The hydrophobicity phenomena is most common in the chaparral soils of southern California. Although not completely understood, the process by which hydrophobic soil layers form has been described in some detail by DeBano (1981). Organic compounds in litter, probably aliphatic hydrocarbons, are distilled during combustion, migrate into the soil profile, and condense on soil particles, forming a water repellant layer. The phenomena is most severe in dry, coarse textured (sandy) soils that are heated to 349 to 399 F (176 to 204 C). It is least severe in wet, fine textured soils where temperatures remain below 349 F (176 C). It also appears that high temperatures, above 550 F (288 C), destroy the compounds. These data suggest that fires that heat soils to an intermediate range of temperature are more likely to cause the formation of a non-wettable layer than fires that only heat the surface of the soil, or those that cause deep penetration of high temperatures; and that certain plant communities, such as those containing chaparral species, are more likely to be affected.

It is important to recognize that hydrophobicity occurs naturally, in the absence of fire, on forestlands, shrublands, grasslands, agricultural lands, and even golf greens around the world (DeBano 1969a, 1969b, 1981). In addition to the potentially severe problem in southern California, it has also been reported, although less severe and of shorter duration, in every western State except Alaska, New Mexico, and Wyoming (Branson et al. 1981, DeBano 1969a). There are several reported "benefits" of hydrophobicity, including evaporation control and water harvesting (DeBano 1981). One additional, unreported benefit occurred in central Oregon where precipitation limited reestablishment of lodgepole pine *(Pinus contorta)* following a severe wildfire. The presence of hydrophobic layers beneath large burned logs channeled water to inter-log areas, providing adequate soil moisture for seedling establishment.

(3) Acidity/alkalinity. pH is a standard measure of acidity or alkalinity, with 7.0 (i.e., the concentration of H^+ ions is 10^{7} equivalents per liter) being neutral on the pH scale of 1 to 14. The scale is logarithmic, so that water or soil with a pH of 5 is ten times more acidic than water or soil with a pH of 6. "Pure" water is neutral, although "pure" rainfall may have pH values between 5.4 and 5.6 due to absorption of CO_2 that reacts to form one or more weak acids. Understanding the pH concept allows understanding of the mechanisms by which fire alters soil pH and thus the soil nutrient regime.

The combustion process releases bound nutrients, many in elemental or radical form. Certain positive ions, collectively called cations, are stable at typical combustion temperatures and remain onsite after burning. They are subsequently washed into the soil where they exchange with H^+ ions; the resulting increase in H^+ ions in solution increases the pH. Nutrient availability is related to soil acidity (c.f., Tisdale and Nelson, 1975). Elements critical for plant growth, such as nitrogen and phosphorus, become more available to plants after a fire that raises the pH of an acidic soil. Fire can significantly enhance site fertility when it raises the pH on cold, wet, acidic sites.

Fire-induced increases in soil pH are widely reported (Chandler et al. 1983, Wright and Bailey 1982). Most cases of increased pH occurred on forest soils where the initial pH was acidic, and a large amount of organic material burned. Increases in

nutrient availability may be highly significant. Rarely do arid or semi-arid soils, which are typically alkaline, exhibit increased pH after burning. Those that do are near neutral initially, may increase a few tenths of a pH unit, then return to preburn pH levels within a year or two after burning. Little effect on the soil nutrient regime occurs.

d. Soil biota. Soil fauna are variously affected by fire (Ahlgren 1974, Chandler et al. 1983, Daubenmire 1968a, DeBano 1979, Mueggler 1976, Wright and Bailey 1982). Aboveground, soil-related herbivores and carnivores usually suffer drastic, but temporary declines, and may be eliminated by "clean" fires (Wright and Bailey 1982). Sub-surface animals respond differently, depending on both amount and degree of soil heating, the size of preburn populations, and the specific organism in question. One study of Douglas-fir *(Pseudotsuga menziesii)* residue reduction burning found that the bacteria *Streptomyces* were not affected by burning but mold populations were significantly reduced. In contrast, prescribed burning in jack pine *(Pinus banksiana)* resulted in greatly increased *Streptomyces* populations that were still increased into the third postburn growing season. Even where bacterial populations immediately decrease after burning, they typically increase dramatically following the first significant postburn rainfall (Chandler et al. 1983).

Fire induced changes in the soil environment may favor one soil microorganism to the detriment of another. Reaves (et al. 1990) reported that growth of populations of species of *Trichoderma*, a soil fungus, was encouraged in soils sampled from a ponderosa pine *(Pinus ponderosa)* site that had been burned by prescription. In a laboratory study, these fungi inhibited growth of *Armillaria ostoyae*, one of several species of *Armillaria* responsible for serious root diseases in coniferous forests and plantations.

(1) Soil moisture content. Fire-caused mortality of soil microorganisms can be related to the amount of moisture in the soil when a fire occurs. *Nitrosomonas* and

Nitrobacter, two bacteria groups critical to nitrification (Huber et al. 1977, Sprent 1987), are killed at 284 F (140 C) in "dry" soil but at 167 and 122 F (75 and 50 C), respectively, in "wet" soil (Chandler et al. 1983). This suggests that this sensitivity to heat may be critical to the recovery of low-nitrogen ecosystems.

Water in soil increases the rate of conductance so that elevated temperatures are reached more quickly in surface layers, especially in coarse soils. Therefore, the premise that soil temperature cannot exceed 212 F (100 C) until all moisture is evaporated is academic with respect to certain organisms that have lethal thresholds below the boiling temperature of water. It is important to note that susceptibility to the heat pulse is usually dependent on time-temperature interaction rather than peak temperature alone. If nitrogen fixing bacteria are a concern, the best treatment may be to burn when soils are wet or moist because they restrict the heat pulse to deeper soil layers.

(2) Mycorrhizae. Fire can have a significant, although indirect, effect on soil mycorrhizae. Mycorrhizal fungi form a symbiotic relationship with roots of most higher plant species of both forests and rangelands. The fungal strands absorb water and nutrients (particularly phosphorus) from the soil and translocate them to the roots of the host plant. The host plant provides photosynthetic products to the fungi. The presence of mycorrhizae can lengthen root life and protect them against pathogens (Harley and Smith 1983 in Perry et al. 1987), and can be critical for the establishment of some species of trees. Most mycorrhizal roots occur in surface soil horizons, particularly the organic soil layer, and decaying wood, especially large diameter decomposing logs. If fire removes most of the organic matter on a forested site, productivity may be significantly reduced for many years (Harvey et al. 1986). If fire kills all species of plants that sustain mycorrhizal associations, spores of these fungi may die after several years. It may then be difficult for desired species of plants to reestablish, either by natural regeneration, planting, or direct seeding.

An important mechanism for reintroduction of mycorrhizal fungi on burned forested areas is dispersal by chipmunks *(Tamias spp.)*. These animals eat fruiting bodies of mycorrhizal fungi in adjacent unburned areas, and spread spores in burned areas when they defecate (Maser 1978b and McIntyre 1980 in Bartels et al. 1985). Downed logs provide important travel lanes and home sites for chipmunks. Therefore, the presence of residual logs after a wildland fire enhances the reestablishment of mycorrhizal fungi, both by enhancing habitat for chipmunks, and providing suitable microsites for mycorrhizal infection and growth.

Little is known about the ecology of mycorrhizae in rangelands (Trappe 1981). Most plants of arid and semi-arid rangelands are mycorrhizal, and many of these same relationships may be true, particularly the association of mycorrhizae with organic matter.

e. Soil nutrients. Nutrient changes occur during combustion. Two distinctions germane to the discussion of nutrients include total site nutrient budgets vs. soilborne nutrients, and total nutrients vs. available nutrients. For example, sites with large volumes of woody material have considerable portions of site budgets bound in organic matter, in forms unavailable to plants. When this material burns, a large amount of nutrients may remain on the site in ash, may leave the site in fly ash or via overland flow, or may volatilize and leave the site in gaseous form. In any event, if bound nutrients leave the site, the site budget decreases but the soil reservoir may remain unchanged (Owensby and Wyrill 1973). Second, even though part of a nutrient's budget may be removed, the remaining portion may be converted into a different, more available form. This latter case is common with nitrogen, which volatilizes at low temperatures; when volatilization occurs, the site budget decreases, but usually the ammonium (NH_4^+) form increases. The ammonium form is directly usable by plants. It is also converted to nitrite by *Nitrosomonas*, then to nitrate, which is also directly usable by plants, by the *Nitrobacter* group. The net result is that while the total amount of nitrogen on a site decreases, the amount of available nitrogen frequently increases.

Postfire nitrogen accretion occurs by such means as fixation by heterotrophic bacteria and symbiotic fixation by nodulated plant roots. On forested sites, many "nitrogen fixers" such as alder *(Alnus spp.)* and ceanothus *(Ceanothus spp.)*, provide rapid recovery of nitrogen (Farnsworth et al. 1978, Raison 1979, Rodriguez-Barrueco and Bond 1968). Bacterial fixation in decomposing wood can also provide an important postfire nitrogen source. In contrast, on chronically nitrogen-deficient sites, including many semi-arid rangelands (Whitford 1986), nitrogen may be depleted by burning too frequently. Burning at intervals of less than 5 to 8 years depleted nitrogen on tobosagrass *(Hilaria mutica)* sites (Sharrow and Wright 1977).

Nitrogen is often the growth-limiting factor on many sites, and is therefore of major interest. Sulfur also volatilizes at low temperatures and its loss also may be important (Tiedemann 1987). Most of the remaining nutrients typically increase or remain unchanged after burning (Chandler et al. 1983, Mueggler 1976, Wright and Bailey 1982).

The cations released to the soil during combustion may be substantial where fire consumes heavy fuel loads on forest sites. However, this so-called "ash effect" is probably minimal on most rangelands. A rangeland site supporting 1,000 pounds (454 kilograms) of completely consumed vegetation per acre that contained 1 percent calcium, would only add about 10 pounds (4.5 kilograms) of calcium "fertilizer" per acre. Most vegetation contains about 3 to 6 percent cations.

2. Water. Wildland fire may affect both water quality and water quantity. The effects are summarized in Chandler et al. (1983), Tiedemann et al. (1979), Wright (1981), and Wright and Bailey (1982).

a. Water quantity. Plants, especially phreatophytes, transpire enormous quantities of water. It follows that breaking the soil-plant-atmosphere continuum should result in a net reduction in water loss. This concept has been applied, with mixed results, in attempts to increase water yield from watersheds (Branson et al. 1981, Davis 1984, Sturges 1983). For example, conversion of shrublands to grasslands has been thought to increase off-site water yield. More recently, Hibbert (1983) suggested that such practices in areas with annual precipitation less than about 15 to 20 inches (38 to 51 centimeters) will probably not result in increased off-site flow. It may be difficult to increase off-site water yield by any practical means in areas where evapotranspiration greatly exceeds precipitation.

There are no conclusive studies that clearly demonstrate that fire causes long-term increased water yield (Settergren 1969). Temporary (for a few years) increases may occur following large, "clean" fires because although direct evaporation may increase, water detention by litter and debris, and transpiration, both decrease. However, the effect is quickly reduced as vegetation and litter return. Demonstration of the "increased yield" is difficult because the effect is often temporally shorter than natural variation in climatic events, and because increased evaporation from the soil surface may compensate for reduced transpiration. There is good circumstantial evidence that greater accumulations of snow may occur following fires that remove some tree cover because of decreased interception of snow by the canopy. However, if the burned area exceeds about four times the height of surrounding cover, snow accumulation may decrease due to wind scour (Haupt 1979). In contrast, water quality may be dramatically affected by fire.

b. Water quality. The literature is replete with evidence of fire-induced changes in water quality, including increased sedimentation and turbidity, increased stream temperatures, and increased concentrations of nutrients resulting from surface runoff (Buckhouse and Gifford 1976, DeByle and Packer 1972, Feller and Kimmins 1984, Helvey et al. 1985, Nissley et al. 1980, Richter and Ralston 1982, Striffler and Mogren 1971, Tiedemann et al. 1979, Wright et al. 1976). The implication is clear: wild and prescribed fires, on forestlands, shrublands, and grasslands, have the potential to decrease on and off-site water quality, and should be mitigated. Effects may be short or long-lived. In a study on 26 to 28-inch (66 to 71 centimeters) annual precipitation rangeland with Mollisol and Inceptisol soils, Wright et al. (1976) found that level areas were unaffected, but adverse effects lasted for 9 to 15 months on moderate (8-20 percent) slopes and 15 to 30 months on steep (37 to 61 percent) slopes. Wright et al. (1976) further found that the average sediment yield was less than 0.01 tons per acre during the first six months after burning from the level sites but was about 10-fold greater on moderate slopes and 100-fold greater on steep slopes.

Mesic, forested sites revegetate much more quickly, but also may be exposed to greater, and often more intense, rainfall. In a study following a wildfire on a ponderosa pine site in central Washington, where annual precipitation is about 23 inches (58 centimeters), Helvey et al. (1985) found that annual sediment yields increased as much as 180-fold above prefire levels. The yields were still 12-fold greater after seven years. A carefully controlled study on a larch *(Larix occidentalis)*, Douglas-fir, and Engelmann spruce *(Picea engelmannii)* site in western Montana (DeByle and Packer 1972) found that sediment returned to preburn levels after about four years. Erosion rates in the Montana study remained below 0.01 tons per acre per year throughout the study.

Methods for mitigating accelerated sedimentation due to fire have not been fully developed. Sedimentation may be reduced by the protection of steep slopes, retention of wide buffers along water courses, rapid revegetation, the presence of residual fuel and duff, and the exclusion of use until recovery.

Fire may induce sudden changes in water chemistry. Such changes probably result from nutrients that are carried into water courses from burned areas. Typically, several forms of nitrogen, phosphorus, and most cations show increases in stream water after burning (Tiedemann et al. 1979). Chemistry is most often altered during the first few storms following fire. Changes include increases in bicarbonates, nitrates, ammonium, and organic nitrogen (Chandler et al. 1983). These nutrients usually are not hazardous to humans but may contribute to eutrophication or algal blooms. Water quality typically returns to preburn levels within one to two years. Some fire retardant chemicals used during fire suppression may be toxic to aquatic animals; the addition of these chemicals near or in water courses should be avoided until specific consequences are clarified.

Stream temperatures also often increase after fire occurs. Usually the temperature increase is due to the removal of overhead protective vegetation rather than direct heat flux from the fire. Elevated stream temperatures are detrimental to most cold water fish species. Therefore, protection of streamside vegetation, and quick revegetation of burned areas, are critical to stream rehabilitation.

C. Resource Management Considerations

1. Expertise on Interdisciplinary (I. D.) Teams. Expertise in soils and hydrology is required on interdisciplinary teams.

a. A soil scientist, knowledgeable about fire effects, should be assigned to interdisciplinary teams involved with fire prescription development, site selection, emergency fire rehabilitation projects, and wildland fire suppression activities.

b. A hydrologist should be assigned to the I.D. team, or at least consulted, if wild or prescribed fire might affect water quality, on or off-site.

2. Statistical Analysis. Statistical analysis is necessary to assess the effects of fire on soils and hydrology.

a. Physical and chemical characteristics of soils typically are extremely variable. Fire effects can vary significantly around a site because of differences in the amount of soil heating. A biometrist or statistician should be consulted before any sampling is undertaken.

b. Adjacent, unburned "control" sites should be used for comparison with burned sites whenever possible to evaluate the effects of fire on soils or water. A biometrist or statistician should be consulted for appropriate sampling and comparison methods. **3. Limited Ability to Extrapolate to Other Sites.** Much of the fire literature describes the effects on soils and hydrology of intense wildfires. Such information should be extrapolated to different regions, soils, environmental conditions, types of fire behavior and characteristics, and to prescribed fires with caution.

4. Variability of Effects. Because fire effects on soils and water are highly variable, consideration should be given to locally documenting effects and relating the effects to fireline intensity, burn severity, fuel, duff, and soil moisture content at the time of the fire, and other appropriate factors.

5. Factors Related to Postfire Erosion. Potential for wind, water, or gravity (especially dry ravel) erosion should be given strong consideration in the timing (i.e., fall vs. spring) of prescribed fires, and in the methods, timing, and species proposed for emergency fire rehabilitation.

a. Delayed recovery of vegetation and slope steepness appear to be important factors in accelerated erosion.

b. The presence of large woody debris and duff after a fire helps to protect the soil from erosion.

6. Management of Soil Heating. The amount of soil heating caused by prescribed fires in forest or woodland areas can be managed.

a. The distribution of soil heating is affected by the choice to broadcast burn, pile burn, or burn windrows. Also, the piling method may be important because machine piles tend to be "dirtier," and hold heat longer, than hand piles.

b. Small diameter, unmerchantable trees (whips) can be slashed just before fire, when they are still green and will not burn well, and thus can contribute little to soil heating.

c. Higher levels of utilization or yarding some unmerchantable material in areas with heavy dead and down fuel loads can decrease the amount of potential soil heating.

d. Burning an area while moisture content of large diameter fuels, lower duff, and soil is high will limit the duration of the fire and the amount of heat penetration into lower soil layers.

e. Rapid ignition techniques (e.g., aerial drip torch) can sometimes be used to shorten the duration of the burn, and thus the amount of soil heating.

7. Leaving Woody Material. When prescribed burning, it is important to leave some coarse, woody debris on the site for nutrient cycling and mycorrhizal function. Agencies may have specific requirements for retention of downed, woody material.

8. Riparian Areas.

a. Buffer strips. When prescribed burning, leave unburned strips of vegetation along riparian areas to serve as slope stability buffers, and decrease the potential for stream sedimentation. Width of buffer strips should be in accordance with applicable agency policy.

b. Season of fire. Riparian areas should be burned, if necessary, in spring when conditions are favorable for rapid recovery of adjacent vegetation.

c. Use of fire retardant. Use of fire retardant chemicals in or near waterways should be avoided. Fire retardant has the greatest impact on small or slow moving bodies of water.

d. Firelines. On erosive soils and/or steep slopes, restrict the location of firelines that lead directly into water courses. Rehabilitate any firelines that were constructed as soon as possible. Replacement of soil and plant material removed during construction is an effective method of fireline rehabilitation.

9. Salvage Logging. Know the potential for soil erosion when considering or planning salvage logging operations after wildfire.

a. Road construction may increase the amount of soil erosion and mass movement. Also, some areas (e.g., Western Oregon) have restrictions to limit "off road" use to minimize compaction. These restrictions may dictate the appropriate logging method.

b. A choice may be made to helicopter log or to not log at all.

10. Need for Closures. It may be necessary to close burned areas to all types of vehicular use, and other uses, for several years because of increased erosion potential.

D. Methods to Monitor Fire Effects

The effects of fire on soils and water are usually extremely variable over time and space due to variations in soil characteristics and plant communities; in the intensity, duration, and timing of postfire precipitation; and in the heat regime of the fire. Many methods used to monitor changes in soils or water quality are time consuming, expensive, and often require elaborate laboratory facilities. Therefore, methods used to monitor fire effects for day-to-day management purposes are usually less extensive and intensive than methods used for research. This section suggests methods for monitoring fire effects that are practical for management purposes. A more complete understanding of soil monitoring techniques can be gained from Black (1965) and Golterman and Clymo (1969).

1. Soil Temperature. Soil temperature, by itself, may not be particularly revealing. However, it may add valuable insight when used in conjunction with other information.

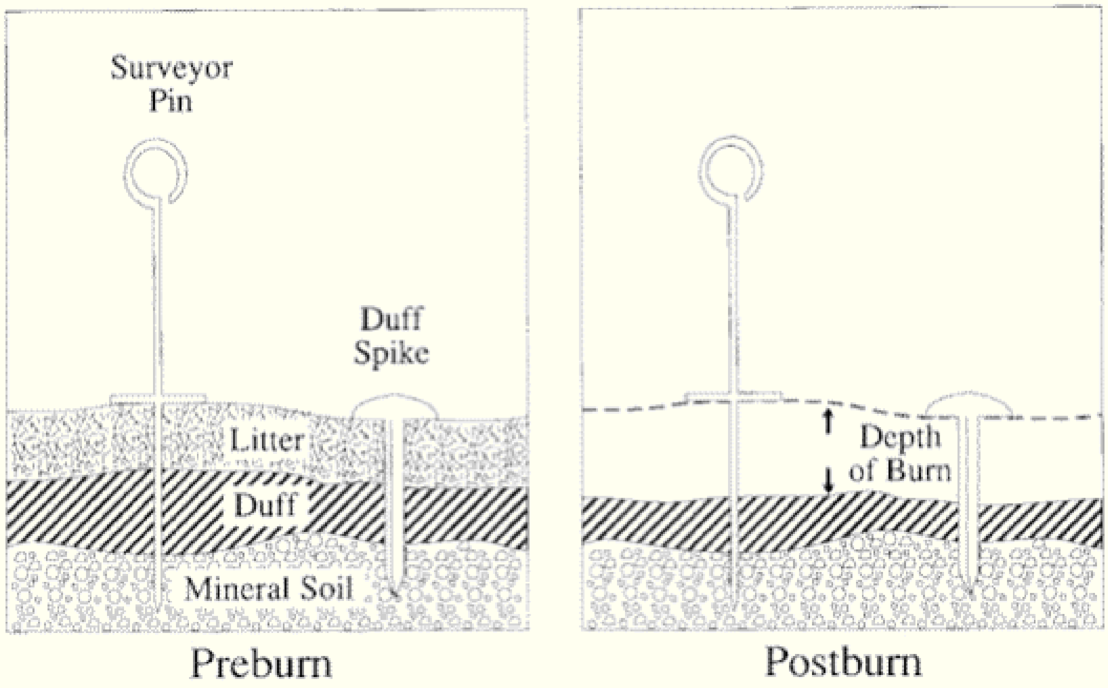
a. Heat sensitive paint. Temperatures can be monitored by placing chips covered with heat sensitive paints at the duff/soil surface and at various depths below the surface. These paints turn color at a temperature specific to each kind of paint, but do not indicate duration of heat. However, it is likely that the presence of high temperatures may increase the certainty of impacts. Temperature sensitive devices and paints are available from many forestry suppliers.

b. Electronic equipment. Extremely accurate data can be obtained through use of sophisticated electronic equipment such as thermistors and thermocouples. A thermocouple can be wired to a strip chart recorder which provides a continuous record of temperature. These are expensive, and can be unreliable unless properly used. Unless research is being conducted on a prescribed fire, their use is probably not warranted. If they are used, advice should be obtained from the research community on appropriate composition, size, number, and placement of thermocouples.

2. Soil Physical Properties. The primary physical effect of fire on soil is the removal of protective cover, which allows accelerated erosion. Erosion can be estimated using predictive models, qualitative guides, or quantitative methods. To isolate fire effects from other effects, burned areas should be compared with adjacent, unburned control areas. The following practical, quantitative methods are suggested for obtaining direct estimates.

a. Erosion.

(1) An appropriate number of "depth-of-burn" pins can be randomly placed onsite before burning to measure duff removal and/or subsequent soil movement. The pins may be of the v-notched survey pin type (with the pin inserted such that the notch is at duff or soil surface level), or the t-bar survey pin type (with the pin inserted such that the cross member rests on the duff or soil surface) (McRae et al. 1979). Bridge spikes also work well. Remeasurements over time will provide an indication of soil movement. Frost heaving and pin disruption by animals may produce erroneous values. Depth of soil removed can be expressed as volume or mass by using appropriate mathematical conversions. This method of estimating soil movement appears to work well for relatively uniform erosion of minor depth, such as wind erosion, but does not work well for irregular soil movement, such as gully erosion, or for mass wasting.



(2) Soil erosion bridges (Blaney and Warrington 1983, Ranger and Frank 1978) are excellent devices for estimating the amount of soil leaving the site, but may require special or additional equipment. Staff soil scientists, or soil scientists from the USDA Soil Conservation Service, universities, or experiment stations should be consulted.

(3) Soil catchments or erosion troughs (Ryan 1982, Wright et al. 1976) are used to collect material after it leaves the site. Commonly, paired watersheds (one member of the pair receives the treatment and the other serves as an untreated control) are used to estimate off-site movement of soil. This method requires the construction of catchments and may not be practical on wildfires or on most prescribed fires.

(4) Other methods are available for estimating accelerated erosion, including models such as the Revised Universal Soil Loss Equation (Renard et al. 1991), radioactive markers (Lance et al. 1986), and photogrammetry (Lyon et al. 1986). These methods are especially useful for research and special management needs, but may not be practical in most fire situations.

b. Hydrophobicity. Hydrophobicity is most easily estimated by the water drop penetration time method (DeBano 1981). A water drop is placed on the sample surface and the length of time to be absorbed is monitored. DeBano suggests that water droplets remaining longer than 5 seconds indicate water repellency. A further refinement of this method uses the surface tension of various ethanol solutions; a water repellency index is then obtained by dividing the critical surface tension into the time (up to 600 seconds) required for water drop absorption. An index value greater than 10 indicates extreme repellency, 1 to 10 indicates moderate repellency, 0.1 to 1 indicates slight repellency, and less than 0.1 indicates wettability.

c. Other physical properties. Fire-induced changes in other physical properties of soils, including soil structure, porosity, and rate of infiltration can be measured using standard methods (Black 1965) for soil analysis. A soil scientist familiar with the methods should perform these analyses.

3. Soil Chemical Properties.

a. Soil water content. Soil moisture content partially regulates the heat pulse into the soil. Because of this importance it is discussed separately here. Several methods of soil water determination are readily available (Roundy et al. 1983) but the gravimetric method (Gardner 1965) is probably the most reliable and commonly used method. An appropriate number of 1 to 100 gram samples are collected in soil cans, weighed (wet weight), oven dried to constant weight (normally at 212 F [100 C]), and reweighed (dry weight). Soil water content is then calculated according to:

soil water content (percent) = (wet weight - dry weight x 100) / dry weight

Drying at temperatures greater than 212 F (100 C) can cause volatilization of soil organic matter, resulting in a loss of materials other than water from the sample, and an overestimation of soil moisture content.

b. Soil pH. The acidity of soil is readily determined using soil paste or aqueous soil suspension and glass electrode pH meters (Peech 1965). Although "standards" are used for meter calibration, it is important to concurrently analyze soil samples from adjacent, untreated soils for comparison, because variations occur among meters and investigators.

c. Soil conductivity. The electrical conductivity of soil caused by the presence of soluble salts is readily determined by using a solu-bridge (Bower and Wilcox 1965). Extracted soil solution is read on the bridge, corrected for temperature, and reported in millimho (unit of conductance). High soil salt content is an indicator that salt-tolerant species should be planted.

d. Other chemical properties. Measurement of other chemical properties of soils that are likely to be affected by the fire require special laboratory equipment and procedures. These are probably beyond practicality for routine fire effects analyses. Bureau soil scientists or soil scientists of other agencies, universities, or experiment stations should be contacted if such analyses are necessary.

4. Water Quantity. Increases in off-site water yield due to burning are most easily determined by measuring changes in streamflow volume before and after burning. If available, paired watersheds should be used. Other agencies, such as the USDA Soil Conservation Service or DOD Army Corps of Engineers, often have gaging stations or use other methods to determine flow volumes on many streams and rivers. In addition, these agencies and many universities and experiment stations often have portable devices that can be used to assess changes in water yield.

5. Water Quality. Several descriptors of water quality can be estimated with minimal investment of time, equipment, training, and personnel, and include turbidity, conductivity, dissolved oxygen content, and temperature. Kits and relatively inexpensive instruments are commercially available for sampling these properties. Experiment stations, universities, and Federal and State water quality agencies can provide assistance.

a. Sedimentation and turbidity. Sedimentation and turbidity reduce the quality of spawning areas and reduce photosynthetic activity. These effects can be amplified by fires that occur near water and may persist for several years or longer. Platts et al. (1983) described general methods for sampling and evaluating stream conditions. Specific procedures for the nephelometric turbidity estimation method are contained in the EPA (1979) publication on water quality evaluation methods.

b. Conductivity. Changes in the specific conductance due to increased ionic composition of the water may provide a useful estimate of the addition of nutrients or contaminants to water. Specific procedures for the use of conductivity meters are found in EPA (1979).

c. Dissolved oxygen. A dissolved oxygen content of about 5 milligrams or more per liter may be necessary to maintain aerobic conditions and support cold water fisheries in Western streams (EPA 1976, Thurston et al. 1979). Because the addition of sediment and nutrients following fire may reduce the oxygen content below acceptable levels, oxygen contents can be monitored to estimate potential impacts on fisheries. Inexpensive kits and meters for measuring dissolved oxygen contents are readily available from chemical and forestry equipment suppliers. The EPA (1979) described specific procedures for using the membrane electrode and modified Winkler methods of dissolved oxygen analysis. A hydrologist, fisheries biologist, or chemist should be contacted for recommendations of preferred methods and equipment in specific locations.

d. Temperature. Reasonably correct estimates of temperature may be important because water temperature, outside of some fairly narrow ranges, can dramatically influence algal blooms, fish survival and reproduction, and a host of other biological activities. However, precise estimates of temperatures in streams can be difficult to obtain because such factors as diurnal variation, angle of the sun, shading, and flow can contribute to error. These factors, as well as thermal layering, can cause equally bad estimates in lakes and ponds. Therefore, depending on the significance of temperature in a particular situation, a temperature sampling scheme should be carefully designed. It should also be noted that data obtained from monitoring and recording devices are usually more reliable than "grab samples" obtained with hand-held mercurial thermometers. Specific procedures for estimating the temperature of water are found in EPA (1979).

e. Other chemical properties. Measurement of other chemical properties of water, such as the concentrations of specific chemicals or nutrients, are probably beyond the practical reach of most land management agencies. Such analyses require special and expensive laboratory equipment and training. If such analyses are necessary, a hydrologist, fisheries biologist, or chemist should be contacted, and appropriate procedures (EPA 1979, Golterman and Clymo 1969, Hem 1970) applied. Analyses can sometimes be completed using inexpensive soil or water testing kits. Results from such testing are not definitive and should remain suspect until confirmed by standard laboratory procedures.

E. Summary

The effects of fire on soils, water, and watersheds are extremely variable. In some cases, such as accelerated erosion, the outcome is reasonably predictable and mitigating measures such as rapid revegetation are necessary. In other cases, such as change in off-site water yield after burning, the outcome is much less predictable because it appears to depend on site-specific characteristics and on unpredictable climatic events. The application of mitigating measures must be based on local experience and local research. In almost all cases, the establishment of a local data base would provide useful information for future events.

CHAPTER VI - PLANTS

By Melanie Miller and Jean Findley

A. Introduction

This chapter discusses the interaction of fire with plants. It explains the basic principles and processes that determine how plants are affected by fire, and the factors that control plant response after the fire. Documentation of burning conditions and fire characteristics provides important information for understanding postfire vegetative response. Use of appropriate techniques when monitoring specific effects of fire on vegetation is necessary to detect changes that occur in the postfire plant community. The goal is to enable managers to predict fire effects on plants based upon knowledge of burning conditions and prefire species and community characteristics, and to interpret the causes for observed variability in postfire vegetative response.

The response of plants to fire can vary significantly among fires and on different areas of the same fire. Both variability in the heat regime of the fire and differences in plant species' abilities to respond affect the postfire outcome. Fireline intensity, burn severity, total duration of combustion, soil heating, time of the year, and time since the last fire all influence mortality or survival of the plants, and thus subsequent recovery. Postfire effects also depend upon the characteristics of the plant species on the site, their ability to resist the heat of a fire, and the mechanisms by which they recover after fire. Plant recovery can be affected by factors that vary with growing season, or age of the plant. Whether the plants that first appear after a fire successfully establish on the site can be influenced by external factors such as postfire weather, postfire animal use, and plant competition.

The inherent abilities of plants to respond to fire depend partially on the fire regime to which the plant community has adapted. For example, a community may characteristically have been subject to frequent, low intensity, low severity understory fires, or the site may have experienced infrequent high intensity fires that killed all standing vegetation. Knowing the "natural" role of fire on a site gives an indication of the type of plant adaptations to fire that may be present.

The most significant sources of heat from most fires are downed dead surface fuels, litter, and duff layers. However, dead branches, leaves, or needles within a plant itself can produce a considerable amount of heat. Old decadent stands of shrubs may produce a more intense fire than a young shrub stand, which may have little dead or dry material and cannot be ignited. The amount of dead woody fuel, thickness of litter and duff layers, and amount of dead material within or around a living plant may be greater than "natural" if fire has been excluded from an environment in which fires used to occur at a moderate to high frequency. In this situation, the impact of fire on the vegetation may be different than it would have been under natural conditions because of the potentially higher temperatures and longer duration of fire that can occur.

B. Principles and Processes of Fire Effects

1. Plant Mortality. Fire-related plant tissue mortality is dependent upon both the temperature reached and the duration of time it is exposed to that temperature. The lowest temperature at which plant cells die is between about 50 to 55 C (122 to 131 F) (Baker 1929 in Wright and Bailey 1982). Some plant tissue may be able to withstand an exposure to 60 C (140 F) for a few seconds, but dies if exposed for about 1 minute. Plant tissue can sustain higher temperatures for greatly decreasing periods of time. Douglas-fir *(Pseudotsuga menziesii)* needles can tolerate temperatures of 70 C (158 F) for only about 0.01 second (Silen 1960 in Martin 1963). Additionally, some plant tissues, particularly growing points (meristems or buds) tend to be much more sensitive to fire heat when they are actively growing and their tissue moisture is high, than when tissue moisture content is low (Wright and Bailey 1982). Thus, plant tissues more readily die after exposure to a specific temperature for a certain length of time when actively growing than when they are physiologically dormant or quiescent, or have finished active growth for the year. Susceptible plant tissue may not be directly exposed to fire heat, because it is protected by other tissues such as bark or bud scales, or is buried in duff or soil. Plant mortality depends on percentage of tissue killed, location of dead tissue, reproductive mechanisms, and species ability to recover from injury.

a. Crown mortality. Both structural and physical characteristics affect the likelihood that the aboveground part of a woody plant will be killed by a given fire. Important crown characteristics include branch density, ratio of live to dead crown material, location of the base of the crown with respect to surface fuels, and total crown size (Brown and Davis 1973). Small size buds are more likely to be lethally heated because of their small mass. Large buds, such as on some of the pines, are more heat resistant. For conifers, long needles provide more initial protection to buds than short needles that leave the bud directly exposed to fire heat (Wagener 1961 in Ryan 1982).

The moisture content of new needles, leaves, and small twigs, the foliar moisture content, fluctuates throughout the growing season. It is highest during the period of active leaf formation and shoot elongation (greenup), subsequently declines to a lower level during the remainder of the growing season, and drops again when foliage cures (Norum and Miller 1984). For conifers, the moisture content of new foliage follows the above pattern, while moisture levels in older needles drop in the spring, and rise again in late spring and early summer (Gary 1971; Chrosciewicz 1986). Moisture content influences foliar flammability because leaves and twigs containing more water require a greater amount of heat to raise them to ignition temperature. Coniferous tree crowns seem to be more susceptible to crown damage in the spring than they are in the fall because tissue moisture of new growth is highest at about the same time the moisture content of old foliage is near its seasonal low and more flammable. The foliage of some shrubs, particularly those with evergreen leaves, contains flammable compounds that allow foliage to burn more readily than if these compounds were not present (Countryman and Philpot 1970; Shafizadeh et al. 1977).

The scorching of a tree crown is primarily caused by peak temperatures associated with the passage of the flaming fire front (Wade 1986). The height above the surface to which crowns are scorched, (crown scorch height), can be estimated from flame length, an output of the Fire Behavior Prediction System, ambient air temperature and windspeed (Albini 1976). (See XII.D.1.a., this Guide, for a more complete discussion of how this model works.) Long-term heating caused by burnout of fuel concentrations after the flaming front has passed can also scorch crowns. The percent of crown volume with scorched foliage is a better indicator of fire impact than crown scorch height because it considers the proportion of live foliage remaining (Peterson 1985). For conifers with short needles, and trees and shrubs with small buds, crown scorch is often equivalent to crown death because of lack of protection afforded the buds (Wade 1986), and the low heat resistance of small buds and twigs.

Crown consumption is the result of the ignition of needles, leaves, and twigs. Needle ignition occurs at about 400F (220C) (Wade and Johansen 1986). For fire resistant conifers with long needles, such as ponderosa pine *(Pinus ponderosa)*, and/or large or well protected buds that are buried in wood such as western larch *(Larix occidentalis)*, crown consumption is often a better indicator of crown mortality than crown scorch. For these species, bud and twig death generally only occurs where foliage is consumed by fire (ibid.).

b. Stem mortality. Trees and shrubs can be killed by lethally heating the cambium, the active growth layer that lies beneath the bark. Bark surface texture can affect its likelihood of ignition, whether stringy and flammable, or smooth. Fire resistance of tree stems is most closely related to bark thickness, which varies by species and with age. The cambium layer of thin barked trees such as lodgepole pine *(Pinus contorta)*, subalpine fir *(Abies lasiocarpa)*, aspen *(Populus tremuloides)*, madrone *(Arbutus menziesii)*, or most of the spruces *(Picea spp.)* is usually dead beneath any charred bark. Heat released in the flaming front, and hence flame length, can be a good indicator of the amount of injury sustained, and even the mortality of thin barked species. External char is not a good indicator of cambium damage on thick barked trees such as ponderosa and Jeffrey pine *(Pinus jeffreyi)*, Douglas-fir, or western larch (Ryan 1982). The cambium beneath thick bark layers is usually only killed by heat released over a long duration, such as from burnout of logs, and deep litter and duff layers, which cannot be predicted by the Fire Behavior Prediction System. The amount of bole damage was a better indicator of postfire survival of Douglas-fir after a series of spring and fall underburns than either scorch height or the percentage of crown volume scorched (Ryan et al. 1988). Once tree cambium is wounded by fire or mechanical damage, it is often more susceptible to additional injury by fire, both because the bark is thinner near the scar, and because of pitch that is often found in association with wounds. A model that estimates tree mortality based on species, and the amount of crown and bole damage is described in XII.D.1.b., this Guide.

Bark thickness, texture, and the presence of wounds or pitch also can affect the likelihood of mortality of shrub stems. However, because of the relatively small diameter of most shrub stems, most stems are girdled by any fire that reaches into their canopy, unless heat is present for only a very short period of time.

c. Root mortality. As with tree and shrub crowns and stems, there are physical and structural characteristics that affect root damage. Structural support roots growing laterally near the surface are more susceptible to fire damage and consumption than those growing downward. Roots found in organic layers are more likely to be lethally heated or consumed than those located in mineral soil layers. This makes shallow-rooted trees more subject to postfire windthrow. Most plants have feeder roots. Tree feeder roots collect most of its water and nutrients, are very small in diameter and are usually distributed near the surface. If most of the feeder roots are located in soil organic layers rather than in mineral soil, they are much more subject to lethal heating and consumption. While this may not always kill the tree, it can place the tree under significant stress. If fire has been excluded for a long time from areas that formerly had a high fire frequency, increased amounts of root (and bole) damage may result from fires smoldering in accumulations of litter beneath trees (Herman 1954 and Wagener 1955 in Wade and Johansen 1986).

Damage to roots and other subsurface plant parts cannot be predicted by the general behavior of the surface fire, nor by any specific descriptors of surface fire activity, such as fireline intensity, flame length, or rate of spread. Temperatures reached in the flaming front may be extremely high, but most of that heat is directed upwards. The mortality of buried plant parts is much more dependent on the total residence time of the fire, the length of time a heat source is present (Wade 1986), not just the length of time flaming combustion occurs. The subsurface heat regime of a fire is influenced by the amount of surface dead fuel, the amount and compactness of litter and duff, and the moisture content of those materials.

Burn severity (see II.B.6.e.), a qualitative measure of the amount of consumption of surface fuels and duff layers, is an indicator of subsurface heating. Soil moisture also retards penetration of heat into soil layers (Shearer 1975; Frandsen and Ryan 1986), thus protecting subsurface plant structures. There is some relationship between heat per unit area, the total amount of heat released in flaming combustion, and root damage, particularly if only a thin layer of combustible fuel is present. However, if moderate to heavy accumulations of surface dead fuel or organic layers exist, their consumption in smoldering and glowing combustion is the best indicator of significant amounts of subsurface heating. (Factors that regulate fuel and duff consumption, and thus burn severity, are discussed more completely in III.B.2. and III.B.3., this Guide.) There may be considerable damage or consumption of roots when little or no damage is apparent in tree or shrub crowns (Geiszler et al. 1984 in Wade and Johansen 1986). The amount of subsurface plant mortality can be indirectly estimated by knowing the location of roots and other buried reproductive structures, and relating this to classes of burn severity.

Plant mortality is often the result of injury to several different parts of the plant, such as crown damage coupled with a high percentage of cambial mortality. Mortality may not occur for several years. Death is often the result of secondary infection by disease, fungus, or insects, because the resistance of plants to these agents is often lowered by injury, and wound sites provide an entry point for pathogens (Littke and Gara 1986). A plant weakened by drought, either before a fire or after wounding, is also more likely to die.

2. Vegetative Regeneration. Sprouting is a means by which many plants recover after fire. Sprouts can originate from plant parts above the ground surface or from various levels within the litter, duff, and soil layers. Where sprouts originate and the depth below the surface at which buried plant parts are found can be species specific characteristics (Flinn and Wein 1977). Heat released in the flaming front of the fire can have a direct impact on mortality of sprouting sites that are above the ground surface. The same factors that control mortality of roots affect mortality of buried reproductive structures of woody plants, grasses, and forbs.

a. Location of dormant buds. Dormant buds are often located on laterally growing stems. Stolons are stems that run at or near the surface of the ground, producing plants with roots at intervals, such as a series of strawberry plants. Rhizomes are laterally growing underground stems located at varying depths in litter, organic, and mineral soil layers. They have a regular network of dormant buds that can produce new shoots and roots. Rhizomes are a structure common to many plants, including blue huckleberry *(Vaccinium globulare)*, thimbleberry *(Rubus parviflorus)*, Oregon grape *(Mahonia spp.)*, common snowberry *(Symphoricarpos albus)*, shiny-leaf spiraea *(Spiraea betulifolia)*, heartleaf arnica *(Arnica cordifolia)*, gambel oak *(Quercus gambelii)*, bittercherry *(Prunus emarginata)* and chokecherry *(Prunus virginiana)*.

Many plants have buds located in the tissue of upright stems, above or below the surface of the ground, such as bitterbrush *(Purshia tridentata)*, bigleaf maple *(Acer macrophyllum)*, rabbitbrush *(Chrysothamnus spp.)*, winterfat *(Ceratoides lanata)*, and mountain-mahogany *(Cercocarpus spp.)*. Bud masses may also be present in branch axils. Paper birch and madrone are species that have root collar buds located in stem tissue at the point where roots spread out from the base of the stem. Lignotubers, burls, and root crowns are names for masses of woody tissue from which roots and stems originate, and that are often covered with dormant buds (James 1984). Dormant buds may be deeplyburied in wood, and may be located far below the surface if the tissue mass is large. Chamise *(Adenostoma fasciculatum)*, serviceberry *(Amelanchier alnifolia)*, scrub oak *(Quercus dumosa)*, tanoak *(Lithocarpus densiflora)*, alder *(Alnus spp.)*, and mallow ninebark *(Physocarpus malvaceus)* all produce sprouts from these buried woody structures.

A caudex is an upright underground stem common in many forb species, such as arrowleaf balsamroot *(Balsamorhiza sagittata)* and lupine *(Lupinus spp.)*, that develops new leaves and a flowering stem each year. Other stemlike reproductive structures are bulbs and corms, which are essentially buried, short thickened stems with a bud or buds covered with fleshy leaves. Some species have dormant buds or bud primordia located along the surface of roots from which new shoots can originate, such as aspen, fireweed *(Epilobium angustifolium)*, and horsebrush *(Tetradymia spp.)*.

b. Postfire sprouting process. Postfire sprouting occurs by a very orderly process. The following discussion describes a likely model of the interactions that control postfire shoot development in woody plants. Similar processes probably regulate sprouting in grasses and forbs. The growth of most dormant buds or bud primordia of woody plants is controlled by a phenomenon called apical dominance. Growth hormones, particularly auxin, a plant hormone manufactured in actively growing stem tips and adjacent young leaves, are translocated to dormant buds, which prevent them from developing into new shoots (Schier et al. 1985). If these plant parts are removed, the source of growth hormones is eliminated. The balance of plant hormones within the buds changes (ibid.). Growth substances in roots, particularly cytokinins, that are translocated to the buds can cause dormant buds to sprout, or stimulate bud primordia to differentiate into shoots (ibid.). Cytokinins may already be present in buds, and a decrease in the ratio of auxins to cytokinins provides the stimulus for bud outgrowth (ibid.). The buds that become shoots are usually those nearest to the part of the plant killed by the fire. If dormant buds are destroyed, new buds may form in wound tissue, called callus, and subsequently produce shoots (Blaisdell and Mueggler 1956). Once new shoots are actively growing, they produce growth hormones that are translocated to other dormant buds that are farther away from the point of damage, suppressing their growth (Schier 1972; Bilderback 1974).

If the organic layer is thinned or disturbed, additional light may reach the tips of rhizomes and stimulate them to grow towards the surface and produce shoots (Barker and Collins 1963; Trevett 1956; Miller 1977). It has also been observed that decapitating a rhizomatous plant causes laterally growing rhizomes to turn upwards and become shoots (Schier 1983). Additional rhizomes often form in response to vigorous aerial plant growth (Kender 1967), which may subsequently produce aboveground shoots. Sprouts from new rhizomes or lateral roots may recolonize areas where all reproductive plant parts were killed by a fire. Plants may sprout soon after a fire, or not until the following spring if the fire occurs after the plants have become dormant for the winter (Miller 1978). Warmer soil temperatures after burning may enhance the amount of sprouting that occurs (Zasada and Schier 1973). The initial energy required to support growth until the sprout is photosynthetically self-sufficient comes from carbohydrates and nutrients stored in the reproductive structures or in adjacent roots (James 1984).

Postfire sprouting ability can vary with plant age. Young plants that have developed from seed may not be able to sprout until they reach a certain age, which varies by species (Smith et al. 1975; Tappeiner et al. 1984). For a given species such as bitterbrush, an older plant may be able to produce few, if any, sprouts that survive (Ferguson 1988). Other factors also may lead to decreased amounts of postfire sprouting in older bitterbrush. These include the higher amount of dead material within old crowns, and the deep organic layer found beneath some old plants that cause increased potential for lethal heating during a fire (Clark 1989; Miller 1988).

Aspen produces sprouts from healthy roots. Decreased amounts of postfire sprouting observed from older aspen stands (Schier 1975) may be because the condition of many of the roots has deteriorated to the point where they cannot sprout (Zasada, pers. conv. 1989). Aspen stands in Alaska can resprout vigorously after fire when they are 150 to 200 years old, perhaps because the incidence of pathogens in Alaskan aspen stands is relatively low (ibid.). In areas such as the Lake States, aspen stands are often killed by cankers by the age of 50 to 70 years (ibid.) An aspen stand that is producing a few understory suckers still has the capacity to sprout after a fire (DeByle 1988a).

Some plants therefore replace themselves, forming a new aboveground stem, but use essentially the same root system as before -- vegetative regeneration. Some plants may spread and develop new individuals from different locations along their roots or rhizomes. Shoots may form their own root system and become separate individuals. This is called vegetative reproduction by some, but these plants are genetically the same as the parent plant, and represent growth of the clone (Zasada 1989). True reproduction only occurs when a new genetic individual is formed, by establishment and growth of a new seedling (ibid.). Sexual reproduction of new individuals of gambel oak occurs when plants establish from seeds. Gambel oak can also regenerate vegetatively, replacing itself by sprouting from a lignotuber, and extending the clone by developing new plants from buds on rhizomes (Tiedemann et al. 1987).

c. Relationship of sprouting to burn severity. A strong relationship exists between burn severity, a measure of the amount of heating at and below the ground surface, and postfire sprouting in forested areas (Miller 1977; Dyrness and Norum 1983; Ryan and Noste 1985; Morgan and Neuenschwander 1988).

(1) A light severity fire occurs under moist fuel conditions, or where little fuel is present. Woody debris is partially consumed but some small twigs and much of the larger branchwood remain. Leaf litter may be charred or consumed, and the surface of the duff layer also may be charred. A light severity fire may kill reproductive parts at or very near the surface such as stolons, or stem buds that are not well protected by bark layers. It has little effect on most buried plant parts and significant amounts of postfire sprouting can occur.

(2) A moderate severity fire occurs when fine and smaller diameter dead fuels and surface litter and organic layers are dry, but large dead fuels and lower organic layers are still moist. Foliage, twigs and the litter layer are consumed. Duff, rotten wood, and much of the woody debris are removed. Logs are deeply charred. This type of fire kills or consumes plant structures in litter and in the top part of the duff layer, such as stolons and shallow rhizomes, and may kill buds on portions of upright stems that are beneath the surface, and buds on the upper part of root crowns. Sprouting occurs from buds in deeper duff or soil layers. Moderate severity fires frequently cause the greatest increases in stem numbers of rhizomatous shrubs (Miller 1976), and of root sprouters, such as aspen (Brown and Simmerman 1986).

(3) A high severity fire occurs when large dead fuels and organic layers are dry. It consumes all litter, twigs, and small branches, most or all of the duff layer, and some large diameter dead, down woody fuels, particularly rotten material. Significant amounts of soil heating can occur, especially near fuel concentrations. This kind of fire can eliminate plants with reproductive structures in the duff layer, or at the duff-mineral soil interface, and may lethally heat some plant parts in upper soil layers. Sprouting can only occur from deeply buried plants parts, which may still be a significant amount for species with deep roots such as aspen, or deep rhizomes such as gambel oak. Killing all belowground reproductive structures usually occurs only where there is a long duration surface heat source, such as beneath a large pile of woody debris that sustains almost complete combustion. Observations show that the concept of burn severity also can be related to fire effects on sprouting rangeland shrubs. The severity of burn relates to the depth of the litter layer beneath a shrub (Zschaechner 1985), and its moisture content when the fire occurs. A light severity fire may scorch litter beneath the shrub crown, but causes little or no damage to reproductive buds buried in stemwood or soil, although it can kill buds at the surface of the soil or those not protected by wood. Most sprouting plants will likely regenerate after this type of fire. A moderate severity fire consumes some basal litter and organic matter, and can kill some reproductive buds. Buds located in deeper litter layers may be lethally heated even if the litter is not consumed, and sprouting of some species may be reduced or eliminated. A high severity fire can consume all litter and organic matter beneath a shrub, and kills all buds and roots in or near the organic layer. This kind of fire favors shrubs with buds and roots buried so deeply in the soil beneath the plant that they do not receive a lethal dose of heat. Fires which occur where there are deep accumulations of litter beneath shrubs and isolated trees, or significant amounts of dead lower branches that burn off and smolder beneath a shrub crown, are more likely to lethally heat roots and reproductive structures than a fire that occurs where there is sparse litter and few dead branches. Reproductive buds of rangeland shrubs that are located on roots or rhizomes at some distance away from the parent plant are not likely to be killed by fire because fuels are often sparse in these locations.

d. Postfire sprouting of grasses. Grass meristems, growing points, are the point where new leaf tissue is formed during the active growing period, and resumes after summer quiescence or winter dormancy. New growth also may occur by "tillering," branching from dormant axillary buds in the plant crown or on rhizomes. Burning of all live leaves may stress the plant, and cause subsequent death. However, a more common cause of death of grasses is the lethal heating of meristems and buds. Sensitivity of meristems and dormant buds to heating relates to their location with respect to the soil surface and the fuel provided by dead grass and shrub litter, and other associated fuels, including shrub canopies. Meristems of some grasses form on long shoots which are elevated above the surface and are readily exposed to fire heat. Their postfire recovery depends upon growth form and whether any basal meristems and buds survive. A detailed description of the vegetative regeneration process in grasses can be found in Dahl and Hyder (1977).

Physiologically active meristems are more susceptible to heat than when they are dormant or quiescent (Wright 1970). Mortality of cool season grasses which green up early in the growing season can be caused by the burning of the litter of associated warm season grasses that are still dormant and hence more heat resistant. A high mortality of perennial grasses also may occur if fire burns in cured litter of annual grasses while perennials are still actively growing.

(1) Response of stoloniferous and rhizomatous grasses. Stoloniferous grasses, those which spread by stolons, such as black grama, are sensitive to heating. Many species of grass are rhizomatous, with meristems and buds buried beneath the litter, duff, or soil surface. Whether rhizomatous grasses are stimulated or killed by fire depends on rhizome location with respect to the surface, whether rhizomes are located in mineral soil or in organic layers, the moisture content of the litter, organic, and soil layers, and the amount and duration of heat generated by the surface fire. Rhizomatous grasses often respond positively to rangeland fires because meristems and buds are usually protected by soil, and a long-term surface heat source over large, contiguous areas is rarely present (Wright and Bailey 1982). In forested areas, grass rhizomes are more likely to be located in litter or duff layers or in association with dead woody fuels, and the potential for lethal heating is higher than it is in rangeland situations.

Litter and decomposed organic matter derived from rhizomatous graminoids such as cattails *(Typha latifolia)*, reeds *(Phragmites communis)*, and rushes *(Juncus spp.)* can accumulate to such thick levels that they completely fill areas of open water in wetlands. Occasional severe fire can have a criticalrole in maintenance of these wetlands. Fires which occur after long dry periods when water levels are low can consume much of this organic accumulation, restoring areas of open water. Prescribed fire is recognized as a management tool for this purpose in the Delta Marshes of Manitoba (Ward 1968). This same role in wetland maintenance for wildland fire has been noted in Alaska (Kelleyhouse 1980), and in the coastal plain of the southeast U.S. (Hermann et al. 1991).

(2) Response of bunchgrasses. The location of meristems and dormant buds of bunchgrasses can be near the surface of the bunch above the level of the soil, or at various depths below the soil surface within or below the bunchgrass litter. Buds and meristems can be readily exposed to lethal temperatures, or be fairly well protected if deeply buried in unburned organic materials or in soil. Fuel and moisture characteristics affect the amount of heat generated. Wright (1971) discusses the relationship between stem coarseness and the rate at which a bunchgrass clump burns. Fine stemmed grasses with a dense clumping of basal stems can burn slowly and generate a fair amount of heat that can be transferred to meristems and buds. Fire tends to pass fairly quickly through coarse stemmed bunchgrasses, which usually have little material concentrated at their base near reproductive structures. Fires tend to burn more rapidly through small diameter bunches in comparison to large diameter bunches, with larger bunches more likely to have enough fuel to release significant enough amounts of heat to affect growing points (Wright and Klemmedson 1965). The amount of surface litter, i.e., the amount of fine fuel, depends on the amount of use by livestock and wildlife, production in this and previous seasons, and the time since the last fire in areas receiving little utilization.

(3) Relationship of moisture conditions and fire behavior to mortality. The moisture content of fine aerial litter, accumulations of basal litter, dead bunchgrass centers, adjacent shrubs and shrub litter layers, and dead woody fuels all affect the amount and duration of heat that the meristems will receive. Mineral soil moisture can control how much heat is received by plant parts located in soil layers. While it is true that moist soil conducts heat better than dry soil, the moisture in surface soil layers must first be evaporated before heating of deeper layers occurs (Albini 1975 in Miller 1977). Moist heat, i.e., steam, may more effectively heat meristems than dry heat, and may be a cause of higher mortality when fires occur where greenup has begun in some plants. However, wet fuel doesn't burn, so the likelihood of a long duration fire under damp fuel and soil conditions that will kill all active bunchgrass meristems and dormant buds is very low, and heat penetration into organic and soil layers is minimal under these conditions (Frandsen and Ryan 1986). If flammable shrubs ignite, dry and preheat adjacent bunchgrass clumps, bunchgrass mortality may be higher than on a similar site with few shrubs that burned under the same conditions (Zschaechner 1985).

For a given litter moisture content, windspeed controls how quickly a fire passes over a plant and the rate at which the litter burns. Fires have been observed in northwest Colorado burning at windspeeds of 10 to 14 miles per hour (16 to 22.5 kilometers per hour) with rates of spread greater than 88 feet/minute (27 meters/ minute) during dry summer conditions. These fires charred only the tops of the crowns of bluebunch wheatgrass and Indian ricegrass plants that were 4 to 5 inches (10 to 13 centimeters) in diameter. Fire may have moved through grass litter too quickly to have a long enough residence time to ignite grass crowns, and little grass mortality occurred (Petersburg 1989).

(4) Relationship of damage to postfire sprouting. A fire may move quickly through a bunchgrass stand with little residual burning. At the other extreme, the dead center of a bunchgrass plant may ignite, smolder, and burn for hours. Conrad and Poulton (1966) developed damage classes for bunchgrasses: 1) unburned, although foliage may be scorched; 2) plants partially burned, but not within 2 inches (5 centimeters) of the crown; 3) plants severely burned, but with some unburned stubble less than 2 inches; 4) plants extremely burned, all unburned stubble less than 2 inches and mostly confined to an outer ring; 5) plants completely burned, no unburned material above the root crown.

Postfire response of a bunchgrass plant can be related to these damage classes, particularly for those species with meristems above the mineral soil surface (ibid.). The highest postfire sprouting potential usually is found in those plants with only some surface litter removed. The amount of sprouting tends to decrease as the amount of basal litter consumption increases, with new shoots most likely to appear from the outside edge of the bunch when little unburned stubble remains. Plant mortality is most likely if all plant material above the root crowns is consumed. Survival and recovery after a specific amount of fire caused damage must also be considered with respect to phenology and other seasonal factors that affect plant response. (See Sections B.4.a.; B.4.c.)

3. Seedling Establishment.

a. Seedbed. Requirements for successful germination and establishment vary for different species. Organic seedbeds, even rotting logs, may be able to successfully support seedling establishment and survival if water is not limiting during the growing season (Zasada 1971). However, moss, litter, and duff are poor seedbeds in many climates because they frequently dry out in the summer, killing the seedling if the root has not yet reached mineral soil. Other attributes of organic seedbeds may also inhibit seedlings (Zasada et al. 1983). For many species, the best seedbed is exposed mineral soil, and microsites where most or all of the organic layer has been removed by fire provide the greatest chance for seedling survival. Soil does not dry out as readily as organic material, and nutrients may be more readily available in ash. Competition from sprouting plants may be reduced.

On hot, dry, exposed sites, seedling germination and establishment may occur more readily if some organic material remains as mulch, especially if the seeds are covered (Clark 1986). Ponderosa pine seedlings are more likely to establish if seeds land on bare mineral soil, and the ungerminated seeds are subsequently covered by litter (McMurray 1988). Allelopathic chemicals, those that inhibit the germination and/or establishment of seeds of plants of other species, are commonly found in the litter beneath certain plant species, including chamise (McPherson and Muller 1969), Utah juniper *(Juniperus osteosperma)*, and singleleaf pinyon *(Pinus monophylla)* (Everett 1987a). Fire can volatilize these chemicals and allow additional seed germination. Some species that must establish from seed may be temporarily eliminated from a burned area because their establishment is not favored by conditions created by a fire. They may require shade, have slow growth of their primary root, or a high water requirement. Some tree species such as pinyon and juniper, may establish a few years after a fire in the shade of plants that established first, but subsequently can grow in full sun (Everett 1987b).

b. Seedbank. The seedbank, the supply of seeds present on a site, is composed of buried seeds, those stored in the tree canopy, and those that are deposited annually. Seeds of some species, such as willow, are very short-lived, and are part of the seed bank for only a short time. Other species have extremely long-lived seeds, and become a fixed member of the soil seedbank, once their seeds are dispersed.

Seed dispersal mechanisms vary. Light seeds may be windblown while heavier seeds may skid across the surface of the snow. Some seeds have wing-like structures which enhance their movement through the air. Seeds with barbs or hooks may be carried by animals. Hard-coated seeds ingested along with their fruit may pass through the bird or animal, with an enhanced likelihood of germination. Seed dispersal from unburned areas depends on the amount of available seed, the distance of the seed source from the burned area, the prevailing wind direction, and the type of seed.

The supply of seeds of a specific species can be greatly influenced by the amount of annual seed production, which can vary significantly (Zasada et al. 1978). Regeneration of conifers may be limited because cone crops are poor during the period of time when exposed mineral soil seedbed is present.

Surviving plants on or near the burned area may not be old enough to produce seed (Zasada 1971; Barney and Frischknecht 1974), or may be too old to produce much viable seed.

The dispersal of seeds from plants occurs at a time that is characteristic for that plant, and can last for different durations of time (Zasada 1986). The time of fire occurrence with respect to seed dispersal can determine whether a species can regenerate promptly. Heat from the fire may kill seeds in the canopy and seeds that have recently been distributed onto the site. Seeds of a certain species may require a period of cold before they can germinate, so seedlings of that species will not appear until the following spring.

Seeds in immature cones in tree canopies may have survived the fire, and may continue to ripen even though the foliage was killed (ponderosa pine) (Rietveld 1976), or the bole was completely girdled by fire (white spruce) (Zasada 1985). Serotinous lodgepole pine cones retain their seeds because of the presence of a resin bond between the cone scales. These cones do not open and release their seeds unless heated to 45 to 50C (113 to 122F), a temperature that melts the bond (Lotan 1976). Numerous lodgepole pine seeds are often released after heating of the canopy during a fire. However, there is considerable variation in the amount of cone serotiny, both on individual trees, and geographically (ibid.). Black spruce *(Picea mariana)* cones are "semi-serotinous", i.e., they open and release their seeds over a period of years (Zasada 1986). Because cones are usually bunched near the top of the tree, some cones are often shielded from fire heat and provide a postfire seed source.

c. Stimulation of buried seed. There may be an enormous reserve of seed stored in litter, duff, and soil. Seed may accumulate on the surface and be gradually buried by litter, or may be cached by rodents and birds (West 1968; Tomback 1986). Seed of some species may remain viable for many years, with dormancy imposed by an impermeable seed coat (Stone and Juhren 1953). Plants such as snowbrush ceanothus *(Ceanothus velutinus)*, raspberry *(Rubus idaeus)*, geranium *(Geranium bicknellii)*, and corydalis *(Corydalis sempervirens)*, as well as many annuals of California chaparral may appear on a site after a fire where they were not apparent before the fire. Seeds of snowbrush ceanothus remain viable for 200 to 300 years (Gratkowski 1962 in Noste and Bushey 1987). Germination of some species is enhanced by fire that can melt or crack the seed cuticle or otherwise scarify the impermeable seed coat (Keeley 1987; Rasmussen and Wright 1987).

Requirements for optimum germination may be very specific, such as redstem ceanothus that has the highest amount of germination after exposure to moist heat at 80 C (176 F) (Gratkowski 1973), explaining its higher germination after high severity fires than low severity fires (Leege 1968). Chemicals leached from charred materials stimulate germination of many species of California chaparral and coastal sage scrub (Keeley 1987). Increased light levels caused by removal of vegetation can induce or enhance seed germination (Keeley 1987). Some species, such as chamise and hoaryleaf ceanothus *(Ceanothus crassifolius)*, produce a certain proportion of seeds that will only germinate after fire treatment, while other seeds from the same plant will germinate under any suitable moisture and temperature conditions (Christensen and Muller 1975a). Annual species may appear from stored seed after a fire, but may disappear in a few years as site conditions change, a common phenomenon in California chaparral (Sweeney 1956; Muller et al. 1968; Christensen and Muller 1975b).

Germination of seeds of chaparral communities is adapted to wildfires that normally occurred during fairly hot, dry, late summer or fall conditions. Seeds of some species of chaparral communities require dry heat to induce germination, but are killed by lower temperatures if they have imbibed moisture. Other seeds require higher temperatures for a longer duration to induce germination than generally occur under spring burning conditions (Parker 1989). If chaparral sites are burned under moist spring conditions, germination of both of these types of seeds is often very much reduced. This is a particular concern for maintenance of a seed bank of chaparral "fire-following annuals" as well as shrub species that can only reproduce from seed (Parker 1987).

d. Dual response plants. Some plants will recover after a fire both by resprouting and by germination of duff stored seeds, while obligate seeders reproduce by seed only. Obligate seeders often have seedlings with better potential for establishment than seedlings from species that sprout (Parker 1984). Other plants have a two-stage response to fire. They sprout from surviving reproductive structures, then produce seed, right onsite, that can readily utilize available seedbed. Fireweed and rabbitbrush, for example, can sometimes gain temporary dominance over a site for these reasons.

4. Factors Influencing Postfire Plant Recovery and Growth.

a. Climate and weather. Different parts of the country have characteristic seasonal distribution of temperature and precipitation. The overall pattern of seasonal plant growth (phenology) relates to climate, such as the time of the year when most growth occurs; occurrence of late summer quiescence, and the onset of winter dormancy. However, the timing and rate of plant development and total amount of growth can vary greatly with seasonal weather (Mueggler 1983). The date when plants begin growth, flowering, and cease growing all relate to seasonal weather (Sauer and Uresk 1976). The average annual occurrence of the wildfire season in various parts of the country is closely related to climate, while the actual timing and severity is related to fuel amount and conditions, and the weather that occurs that year. Generalizations can be made about weather trends and patterns for a particular region, but there are always exceptions. Long-term averages do not reflect the wide range of conditions possible.

(1) Prefire weather. Prefire weather can affect the plant growth stage at the time of burning. The amount and availability of fuel is influenced by weather. Fires in the cheatgrass region tend to be much larger in years with high winter precipitation and spring rain, resulting in high production of fine fuel. Burned acreage in the Sonoran Desert tends to be higher after two winters of above average precipitation that promotes growth of winter annuals, which subsequently dry and provide fuel (Rogers and Vint 1987). The moisture content of heavy fuels and deep litter and duff layers is closely related to temperature and precipitation in previous months, and thus the likelihood that these fuels are available to burn and provide a long-term heat source is weather dependent (Brown et al. 1985). Fire size, and its degree of impact on vegetation, is influenced by fuel availability, and burning conditions at the time of the fire. Drought, anomalous high winds and low humidity, or high summer precipitation all affect the immediate impact of a fire in a particular area in a certain year.

(2) Postfire weather. Postfire weather can affect plant survival. Sprouting plants must produce enough growth to restore food reserves before the next period of high use, and this growth can be enhanced or limited by weather. Without restoration of carbohydrate reserves, the plant may die. Plants which sprout late in the season also can die because they have too little time or energy to harden off for the winter. The amount of autumn rain can determine whether germination of seed of some species occurs in the fall or the following spring. Late summer rains (Thill, Beck, and Callihan 1984) followed by a dry period can cause germination, and subsequent death, of many seedlings. Weather in the following years affects the rate of recovery from burning by influencing productivity. Drought can place additional stress on injured plants, and increase the likelihood that they will die. Postfire weather is a primary factor in determining range readiness for postfire grazing use. The weather cannot be controlled, but it is important to document it. Fires burned on similar sites in different years with the same burning weather may have widely varying results because of differences n prefire and postfire weather. Analysis of these records may explain the reasons for significant variations in postfire response, and the "success" or "failure" of a specific prescribed fire project.

b. Carbohydrates.

(1) Carbohydrate cycle. Carbohydrates are starches and sugars manufactured by plants and used to provide energy for metabolism, and structural compounds for growth (Trlica 1977). Carbohydrates which are manufactured in excess of those used are stored in various parts of the plant, such as roots, rhizomes, root crowns, stem or leaf bases (Cook 1966a), or evergreen leaves. There is a seasonal cycle of depletion and restoration of total available carbohydrates (TAC) that relates to events in the growth cycle of the plant. The most rapid depletion usually occurs during greenup, to support initial development and growth of leaves and shoots. Stored carbohydrates levels also may be lowered during the period of flower and fruit development. Carbohydrates are required to prepare plants for winter, the "hardening off" process, as well as for respiration and cellular maintenance during winter dormancy (McCarty and Price 1942 in Trlica 1977), and quiescence during late summer drought conditions (Hanson and Stoddart 1940 in Trlica 1977). Roots must be maintained even while the aboveground part of the plant is not actively growing. A major depletion also can be caused by heavy or repeated grazing or browsing, as the plant may need to use reserves to support subsequent new growth if inadequate leaf area remains on the plant to provide energy for growth and new tissue formation. "In general, too heavy, too early, or too frequent grazing or defoliation result(s) in declining vigor of vegetation" (Hedrick 1958 in Trlica 1977).

Restoration occurs when production by photosynthesis exceeds demands for growth and respiration. The beginning and length of the restoration period varies. Cook (1966a and b) discusses the seasonal carbohydrate cycle. Some plants rapidly deplete, but then quickly restore carbohydrate reserves, all within about the first month of growth. Squirreltail *(Elymus elymoides)* is a plant that exhibits this V-shaped carbohydrate cycle. U-cycles are shown by plants that deplete carbohydrate reserves over a much longer period of time, and don't begin to make a significant restoration of reserve carbohydrates until later in the growing season. Bitterbrush, a species with a U-cycle, does not show a major increase in its level of reserves until August or September, only accumulating carbohydrates from the period of seed formation until leaf fall (McConnell and Garrison 1966). The timing of highs and lows in the carbohydrate cycle thus corresponds with the growth states of the plant. The cycle can differ from year to year because the timing of phenological events can vary significantly among years (Sauer and Uresk 1976; Schmidt and Lotan 1980; Turner and Randall 1987). The amplitude of the cycle will vary with growing conditions, the amount of growth produced, and the amount of carbohydrate used for other physiological processes, all of which affect the amount left for storage.

(2) Relationship to fire. Energy and material for initial plant regrowth after a fire depend on the availability of reserve carbohydrates. The biggest negative impacts from burning may occur during the lowest point in a plant's annual carbohydrate cycle, usually during the early seasonal growth period. The low survival of chamise sprouts after spring prescribed fires has been attributed to low winter and spring carbohydrate reserves because of high spring demand for growth, flowering, and fruiting (Parker 1987). For other species, the effects are most negative if the plant is burned late in the growing season when reserves are being rapidly replenished because the plant uses a considerable amount of stored carbohydrates to sprout, but does not have enough time to restore reserves before winter dormancy (Trlica 1977; Mueggler 1983).As a result of burning during an unfavorable growth period with respect to stored carbohydrates, a plant or any sprouts that it produces may die during the next long period of carbohydrate demand, such as summer quiescence or winter dormancy. If the plant survives, its productivity in the next few years may be greatly reduced. An additional consideration when burning old stands of woody plants is that energy reserve levels of these plants may be low, because annual production is low, and/or much of the plant's carbohydrate production is used to maintain old plant tissue. Meager amounts of sprouting observed from old bitterbrush plants may be partially due to low levels of stored root carbohydrates.

The degree of dependency for regrowth that a plant has on carbohydrate reserves after fire depends on whether any photosynthetically capable material, such as sheath leaves on stubble, survived. If some plant tissue that can photosynthesize remains or rapidly regenerates, newly grown leaf material soon manufactures all of the carbohydrates that the plant needs for growth and respiration. However, the initial spurt of growth after a fire likely requires use of some stored carbohydrates, even if only for a day or two. Evidence from clipping and grazing studies has shown that the recovery of grass plants is more related to the removal of growing points than to the carbohydrate level at the time of defoliation (Caldwell et al. 1981; Richards and Caldwell 1985). However, fire may have a greater impact on grass plants than severe defoliation because a majority of the carbohydrates used to initiate regrowth are derived from the basal portion of the older tillers, and these may not survive a fire.

c. Postfire plant competition. Plant competition occurs when growth and reproduction of one plant is hampered by the presence of another, or, when the resources of a site required by one plant are reduced by another (Harris 1977). Plants compete the most for whatever is in shortest supply - particularly water, nutrients, and light. Whether competition occurs and the degree to which it occurs depends on the species present on the site, the number of plants present, and the site conditions (Samuel and Depuit 1987; Brand 1986). Simultaneous requirements for limited resources such as water and light can place individuals in competition with each other. Whether certain species compete depends on the timing of germination and growth, germination and establishment requirements, rate of growth, and requirements for water and nutrients. Some species have an innately high ability when in a seedling state to compete with seedlings of other species (Samuel and DePuit 1987). The ability of a plant to respond to changes in the supply of nutrients or water varies by species (ibid.). Some species can take better advantage of changes in the postfire environment than other species can, which may give them a competitive advantage.

Fire affects plant competition by changing the numbers and species of existing plants, altering site conditions, and inducing a situation where many plants must reestablish on a site. In a postfire situation established perennial plants that are recovering vegetatively usually have an advantage over plants developing from seed because they can take up water and nutrients from an existing root system while seedlings must develop a new root system (ibid.). Natural regeneration of shrubs may severely limit growth of naturally occurring or planted conifers because of competition for light or moisture (Stein 1986; Haeussler and Coates 1986). If perennial plants are few, or their postfire survival is low, and a seed source is present, seedlings may establish and dominate the community for varying periods of time. Certain species may be favored, such as ceanothus (Parker 1984), because of the sheer volume of seeds on a site. Cheatgrass *(Bromus tectorum)* has such a great postfire advantage over seedlings of most native grasses because roots of cheatgrass seedlings can grow at much cooler soil temperatures than those of most native perennial grasses, and can proliferate much more rapidly at warmer soil temperatures than can roots of natives. Cheatgrass seedlings can deplete soil moisture in the spring before other species get their roots down into the soil profile (Thill, Beck, and Callihan 1984).

Grass seeded for postfire erosion control in forested areas may easily overtop conifer seedlings. In chaparral areas, seeded grasses compete with sprouts and seedlings of native plants (Barro and Conard 1987). Litter from seeded grasses may increase the flammability of these sites to higher levels than would occur if only native vegetation recovered on the site (Cohen 1986 in Barro and Conard 1987). A second fire after a short time interval might kill all seedlings of native species, often before they have produced much seed, decreasing the number of seeds in the soil seed bank. Conversely, if seeded crested wheatgrass establishes on a cheatgrass site after it burns, the amount of litter, and fire frequency, can decrease.

A lack of fire can also increase plant competition. One hundred year old stands of juniper usually have very low cover of shrubs and grass (Barney and Frischknecht 1974), probably because of juniper's superior ability to extract soil water, as well as the inhibitory effect of juniper litter on germination and establishment of seedlings of shrub and herbaceous species (Everett 1987b). Herbaceous production in the vicinity of sagebrush plants decreases as sagebrush cover increases, because of root competition (Frischknecht 1978). Young stands of conifers that develop in the absence of fire beneath mature overstories of ponderosa pine compete for moisture and nutrients with the mature trees (Wyant et al. 1983), weakening them and making them susceptible to insects and disease.

d. Animal use. If burning occurs in close association with heavy use of the plant community by livestock or wildlife, either before or after the burn, plant recovery may be delayed or prevented. Heavy postfire use of perennial plants in the first growing season after a fire is likely to cause the most harm, particularly in arid and semiarid range communities (Trlica 1977). Depending upon the plant community and its production capabilities, some use after the first full growing season may not have a negative impact, and may even be desirable, as in tobosagrass communities. Two full growing seasons of postfire rest are necessary before plants can sustain much utilization in the Intermountain west after wildfire (Wright and Bailey 1982). A longer recovery period is necessary if weather has been unfavorable for growth, or if establishment of plants from seeds is required to completely revegetate the site. Desert plants required more than seven years of recovery after moderate defoliation (Cook and Child 1971 in Trlica 1977), and some shrubland sites may require this long a period of postfire rest if recovery of browse species is desired. See Chapter IX.B.2 and B.3 for additional discussion on this topic.

5. Plant Productivity. Fire can affect postfire plant productivity. Short-term decreases can be caused by plant mortality, reduction in basal area of grasses, forbs, and shrubs, changes in species composition to less productive plants, and reduced availability of soil nutrients. Increases are caused by fire induced vegetative reproduction and regeneration, fire enhanced seedling germination and establishment, improvements in the soil nutrient regime, and increases in soil temperature. Warmer soil temperatures often result in earlier greenup on burned areas, particularly in grassland and rangeland environments.

Removal of thick layers of litter and organic matter in tall grass, wetland, and boreal environments increase soil temperature and nutrient availability, enhancing plant growth (Vogl 1973 and Hulbert 1969 in Young 1986). An occasional fire is very important for rejuvenating cold, nonproductive forest sites in interior Alaska (Yarie 1983), and this is likely also true for many tundra sites. Where permafrost is present, many nutrients are tied up in frozen organic layers, and are unavailable to plants (Heilman 1966; 1968). Fire's removal of insulating organic matter and the blackened surface cause deeper annual soil thawing, and a greater depth and higher temperature of the rooting zone. Soil acidity decreases and rates of nutrient cycling increase. Vegetatively regenerating plants and seedlings use these nutrients, significantly enhancing growth. Eventually, organic matter accumulates and becomes an effective soil insulator, causing a decline in both growing season soil temperatures and associated plant productivity.

There may be a significant decrease in productivity during the initial postfire recovery period, then an increase in production after one or several years. Some conifers have reduced growth the first growing season after the fire, but show increased growth rates in subsequent years caused by the removal of competing trees (Reinhardt and Ryan 1988a). Total productivity may not change, but can shift among classes of plants on the site, such as from conifers that are killed by a fire to shrubs, grasses, and forbs (Volland and Dell 1981). Total site productivity may actually decrease, but production of shrubs, grasses, and forbs often increases over prefire levels (Harniss and Murray 1973; Dyrness and Norum 1983). On sagebrush sites, total prefire productivity may not be reached until sagebrush again dominates the site (Bunting 1985), because its deep root system can allow it to utilize site resources that are physically unavailable to other plants.

The length of productivity changes depends on the ecosystem, the degree of change caused by burning, and the resulting amount of change in species composition in the postfire plant community. A low intensity, low severity fire may have little effect, while a shift from an old coniferous forest to a shrubfield may result in long-term changes in plant production. Site productivity in the first few years after fire will likely be higher if a significant amount of vegetative regeneration occurs, than if plants on the site must reestablish from seed. Sprouts can obtain nutrients and carbohydrates for initial growth from the parent plant while a seedling often has access to only a small nutrient reserve in seed, and may initially grow fairly slowly. Seedling establishment and growth are much more dependent on site conditions and postfire weather. Snowbrush seedlings grow slowly until age 4 or 5, but then grow rapidly until about age 10, while sprouts of snowbrush may grow from 1 to 2 feet (0.3 to 0.6 meters) per year from the time growth is induced (Peterson 1989). Exceptions to this general rule do occur. Obligate seeders, plants that must regenerate from seed, can be adapted for making rapid growth on burned or disturbed sites (Parker 1984).

Greatly increased amounts of flowering and fruiting may occur, including a significantly enhanced output of grass seed and berries (Daubenmire 1975; Young 1986; Christensen and Mueller 1975b). Changes in production are caused by the same factors that increase vegetative productivity: warmer soil temperatures, improved nutrient availability, and removal of senescent, woody material that requires a lot of energy to maintain. For a given species, flower and fruiting generally occur sooner on sprouts than on plants that develop from seed. For some species, flower buds are formed on the previous year's growth, so it takes two growing seasons for flowers and fruits to appear. Increased levels of fruit or seed production may only persist for a few years of burning. Improvements in forage amount and availability, and increases in flowering and fruiting are key reasons for wildlife and livestock attraction to newly burned areas.

C. Resource Management Considerations

Fire effects on plants cannot be understood unless their survival and reproductive strategies with respect to fire are understood. Some plants resist fire by characteristics such as thick bark or buds that can withstand scorching temperatures. A site can be repeatedly burned, and many of these plants survive. Plants may have their surface parts completely consumed, but endure the fire because belowground reproductive structures typically survive. Some plants are almost always killed by fire, and their seedlings cannot tolerate immediate postfire conditions. It can be said that these species avoid fire, because they are only found on sites that are fire-free for long periods of time (Rowe 1983).

Plants can be divided into four basic groups with respect to postfire revegetation of a site (Stickney 1986), as defined by their source and time of establishment. Survivors are species with established plants on the site that can regenerate after a fire. Colonizers are species that establish on the site from seed. Residual or onsite colonizers originate from seed that is present on the site at the time of the fire. Off-site colonizers develop from seed that is carried from off the site. Secondary off-site colonizers develop from off-site seed, but not until site conditions are mitigated by the plants that established first. Initial establishment of a plant is only the first step, because its long-term survival and productivity is affected by competition with other plants and by weather.

The following management considerations summarize key elements to consider with respect to predicting, observing, and interpreting the effects of fire on plants. They are derived from information explained in greater detail in the text of this chapter, as well as in Chapter II. Fire Behavior and Characteristics, and Chapter III. Fire Effects on Fuels.

1. Plant Mortality.

a. Relationship to fire behavior, fire characteristics, and fuels.

(1) Flame length relates to the amount of crown scorch and canopy consumption.

(2) Dry concentrations of down, dead woody fuels can ignite and provide a long-term heat source that can damage a tree crown, tree stem, roots, or buried reproductive structures.

(3) The amount of heating that results from combustion in the flaming front of a prescribed fire can be regulated. Ignition methods and techniques must be selected with consideration for fuel conditions, weather, and slope steepness and concavity.

(a) The width of the flaming zone can be manipulated by controlling the number of lines of strip headfires that are ignited at once (Norum 1987), and the spacing between them.

(b) Regulating the interval between lines of strip headfires controls flame length, because the shorter the interval between lines, the shorter the flames (ibid.)

(c) Use of rapid ignition techniques can greatly increase the rate of heat release and decrease the duration.

b. Crown scorch height.

(1) The height to which tree crowns are being scorched is often not obvious during ignition of a prescribed fire.

(2) Scorch height can be estimated from current weather, and observed flame lengths, using the graphs in Albini (1976, pages 63 to 66).

(3) If scorch height is too high, then ignition can be altered to lower flame lengths, or the fire may be curtailed until more moderate burning conditions occur.

(4) Too high a scorch height can indicate that the fire prescription may require modification to reduce scorch heights, such as by prescribing increased fuel moistures, or lower air temperatures when the fire is ignited.

c. Mortality of crowns.

(1) Dormant buds have varying degrees of sensitivity to fire heat. Sensitivity relates to size, the presence of protective bud scales or needles, and whether they are physiologically active or dormant.

(2) Foliage flammability and sensitivity to scorching temperatures varies seasonally, especially because of changes in foliar moisture content.

(3) Foliage flammability varies by species according to branch density, the presence of lichens, presence of flammable compounds, retention of ephemeral or evergreen leaves or needles, and the proximity of the crown base to the surface of the ground.

d. Mortality of tree stems and cambium.

(1) Thick barked species are more resistant to fire heat than thin barked species.

(2) Duration of heating is generally more important than peak temperature in determining damage to thick barked trees and shrubs.

e. Mortality of roots and other buried reproductive plant parts.

(1) Potential for heating to lethal temperatures relates to the plant part and its location.

(a) Depth of roots or reproductive structures below the surface.

(b) Whether plant parts are located in litter, soil organic layers, or mineral soil.

(2) The potential for heating relates most closely to the duration of heat released during the consumption of accumulations of dead woody fuels or deep litter and duff layers. Duff reduction relates to its moisture content (Norum 1977; Brown et al. 1985). See Chapter III.B.3.a. for moisture content guidelines for consumption of organic soil layers.

(3) Moist soil retards the penetration of heat and protects buried plant parts.

2. Postfire Sprouting.

a. Process. The physiological processes that control postfire sprouting are essentially the same for trees, shrubs, forbs, and grasses.

b. Species specific characteristics. The type of plant part on which dormant buds are located, the subsurface distribution of reproductive structures, and the depth below the surface from which new shoots can develop are species specific characteristics.

c. Relationship to burn severity. Sprouting is closely related to burn severity because the number of postfire sprouts relates to the number of reproductive buds or bud primordia that survived the fire. A species may be enhanced or harmed depending on how deeply lethal temperatures penetrated below the surface, and the characteristic depth of its reproductive structures.

d. Spread from adjacent areas. On sites where all reproductive structures were killed, sprouts may develop from rhizomes or roots that colonize the area from adjacent, less severely burned areas.

e. Bunchgrasses. Bunchgrass species also have reproductive buds located at characteristic depths below the surface, and with respect to accumulations of dead basal material.

(1) Moisture contents of basal litter, dead centers of plants, and soil are critical for determining the amount of consumption of a bunchgrass plant.

(2) There is a potential for additional heating of bunchgrasses from burning of adjacent shrubs, with the amount of heat related to shrub species, density, and flammability.

(3) The amount of consumption of a bunchgrass plant of a particular species can be related to its potential for postfire sprouting, because it relates to the amount of physical damage to growing points and dormant buds.

3. Postfire Reproduction by Seed.

a. Seed ecology. The likelihood that a species will reestablish from seed depends upon its seed ecology.

(1) Germination and establishment requirements.

(2) Whether its seed is sensitive to heating or is stimulated by heat or chemicals leached from charred materials.

(3) Length of period of seed viability.

b. Seed source.

(1) Distance from living, seed producing plants.

(2) How much seed in organic and soil layers survived the fire.

c. Timing of fire.

(1) Production of current year's crop of seeds.

(2) Age of plants on or near the site, whether they were old enough to produce seed.

d. Soil seedbank. Some species of plants may establish from duff or soil stored seed and produce a significant amount of biomass.

(1) The length of time that a species persists depends on its habitat requirements and how the site conditions change as plant succession proceeds.

(2) Different plant communities have characteristically different species and numbers of seeds in their seed bank, that also vary in longevity.

e. Relationship to burn severity.

(1) The amount of bare mineral seedbed created.

(2) The amount of heat stimulation or mortality of specific species.

4. Carbohydrates.

a. Plant phenology. Plant growth stage is related to the level of stored carbohydrates that provide energy for initial postfire vegetative regrowth.

(1) The amount and timing of high and low levels, and rates of recovery, of stored carbohydrates varies by species, and with conditions in a particular growing season.

(2) Recovery of a plant may be most affected if a plant is burned during a low point in its carbohydrate cycle, or when there is not enough time for the plant to rebuild stored carbohydrate levels before the next period of high demand.

b. Animal use. Prefire and postfire use of a site by livestock and/or wildlife must be evaluated and managed, particularly important if heavy utilization has occurred.

5. Postfire Plant Productivity.

a. Tree growth. Postfire productivity of surviving trees relates to the amount of injury to crowns, stems, and roots.

b. Sprouting woody plants. Rapid recovery of perennials may occur by postfire sprouting if reproductive structures were not killed.

c. Seed and fruit production. Seed and fruit production generally increase much more quickly from plants that regenerate vegetatively, than from plants that must establish from seed.

6. Direct Seeding and Planting.

a. Postfire rehabilitation considerations.

(1) The requirement for seeding is determined by the specific situation on the burned area and the management objectives for the area. Factors such as erosion control, native species restoration, limiting establishment of annual exotics, and meeting wildlife habitat requirements are major considerations in the decision whether or not to do postfire rehabilitation.

(2) The likelihood of survival of native species should be assessed before artificial reseeding is planned. The percentage mortality of individual plants should be estimated, and likely methods of recovery determined, such as vegetative regeneration or plant establishment from stored seeds. (See B., this chapter.) Reseeding is not necessary where recovery of native plants will occur.

(3) Prefire species composition may be determined by inspection of adjacent unburned areas. A seed mixture of species adapted to the site results in the highest likelihood of establishment, as well as the greatest long-term diversity and productivity. Grass, forb, and shrub mixes have been successfully seeded on some Federally managed rangelands.

(4) Seeded grasses may compete with other desirable species.

i. Seeded grasses can interfere with the establishment of native plants, and limit the future seed bank of those species.

ii. Seeded grasses can provide significant competition to planted trees and shrubs.

iii. Where postfire erosion is a significant threat, seeding annual or short-lived perennial grasses may allow greater recruitment of native plants than seeding long-lived perennial species.

b. Need for rapid replanting. It is often necessary to plant tree seedlings on productive sites as soon as possible after fire because of potential competition from naturally regenerating shrubs, grasses, and forbs. Rapid reseeding of rangeland sites is required if an objective is to establish perennial species on a site dominated by annual exotics.

c. Prescribed fire considerations.

(1) Residual logs and duff can enhance site productivity by providing sites for mycorrhizal infection, and nitrogen fixation, both of which are beneficial to establishment and growth of tree seedlings. (See V.B.c.(4) and (5), this Guide.)

(2) Residual downed logs and shade from standing dead trees provide shade that can aid establishment of planted seedlings or natural regeneration on dry forest habitat types.

7. Effect of Postfire Weather. Postfire weather has a significant effect on the rate and amount of postfire vegetative recovery.

8. Need for More than One Treatment.

a. Dual treatment. A site may require burning after mechanical, chemical, or manual treatment to kill residual target species or seedlings developed from residual seed.

b. Maintenance burning. Repeated burning at regular intervals may be necessary to prevent reinvasion of the site by seedlings of undesired species. The desired burning interval is related to the natural regime that fire used to play in the vegetation community on the site.

D. Methods to Monitor Fire Effects

A variety of monitoring methods have been employed to study vegetative attributes and their changes over time. Those methods most appropriate for postburn studies will be reviewed here. Monitoring schemes chosen to evaluate the effects of fire on both individual plants and plant communities must be sensitive to the responses observed from the perturbation of burning. If the fire has been planned, methods selected by the observer to evaluate changes in the vegetative component must necessarily follow the objectives of the fire so that the vegetative responses can be properly evaluated. Preburn measurements are critical, and thoughtful establishment of almost any preburn study will provide valuable information.

Specific attributes of vegetation or plant communities affected by fire may be expressed in the generally accepted terms of cover, density, frequency of occurrence, weight, species composition, number, height, vigor, growth stages, age classes, and phenology. Plant mortality, injury to trees, and burn severity, all a direct function of burning, also merit consideration because they directly or indirectly relate to postfire effects. Definitions of each of these attributes will be given as monitoring methods appropriate for each are discussed.

In addition to selection of the most appropriate methods of study, two other considerations are vital for a successful monitoring program: control plots must be established outside the area of the fire so that universal factors that may be influencing results, such as climate or insect infestations, can be separated from effects of the fire itself; and, timing of studies must be planned so that plants have reached maximum growth, and repeat studies should be taken as near to the same time as possible as the initial studies were conducted. For planned burns, control plots must be in similar vegetative communities as the area to be burned in order to make valid conclusions regarding fire effects.

Standard references discussing both the philosophy and methodology of vegetation sampling include Cook and Stubbendieck (1986), Greig-Smith (1983), Mueller-Dombois and Ellenberg (1974), Pieper (1973), and Brown (1954). A rangeland monitoring guide describes in detail the more commonly used techniques for monitoring range trend, some of which may be valuable in monitoring fire effects (USDI-BLM 1985a). Additional guidance for prefire and postfire vegetation monitoring is found in USDI-NPS (1992).

The matrix in Table VI-1 relates the specific effects of fire on vegetation to measurable vegetation and site attributes, so that appropriate methods of study can be most efficiently chosen for the effects to be measured. In designing any sampling scheme, the community type being sampled must be considered in determining which methodology to be employed, as well as size and shape of plots to be used. Chambers and Brown (1983) outline appropriate quadrat sizes and shapes for specific methodologies in a variety of vegetation types. It is important to work with qualified personnel to design valid sampling schemes and methods of analysis. (See Chapter XI, this Guide, for a discussion of sampling and statistical analysis.)

1. Cover. Cover refers to the area on the ground covered by the combined aerial parts of plants expressed as a percent of the total area. Specifically measured are either basal cover, which is the vertical projection of the root crown on to the ground, or foliar cover, which includes the projection of all plant parts vertically on to the ground. Cover of litter, rocks, or any other physical parameter on the ground may be determined using cover measurements.

Points and point frames are also used to measure cover (Chambers and Brown 1983; Floyd and Anderson 1983), and are particularly suited to dense or rhizomatous vegetation where intensive sampling is desired. A disadvantage of using points is that an extremely large number of points must be collected to obtain a representative sample of the population. Specific methods include vertical and inclined point frames, points along line transects, and pace transects. First hit only or all hits through the various canopies may be recorded. Usually, aerial canopy is used for trees, shrubs and broadleaf perennial forbs, and basal crown is used for grasses and single-stemmed forbs. Where vegetation is identified by layer, cover may exceed 100 percent. Cover using quadrat frames has been employed by Daubenmire (1959). The method uses canopy coverage classes for each species within a given frame.

Table VI-1: Vegetation and Site Attributes Useful for Evaluating Selected Fire Effects.

FIRE EFFECTS

ATTRIBUTES	Mortality	Reproduction		Productivity	Structure
		Sprouts	Seedlings		
Cover	I*			I	D
Density	D**	D	D		
Frequency	I	I	I		
Weight	I			D	
Species Composition	D		D		I
Number	D				D
Height				I	D
Crown scorch	D/I***	I***		I	
Crown consumption	D***	I***		I	D/I
Stem char	D/I***	I***			
Burn severity	D/I***	D/I***	I	I	D/I

* I = Indirect Relationship
** D = Direct Relationship
*** = Depends upon species

2. Density. Density is the number of plants or parts of plants per unit area, although older literature may use the term to refer to the attribute of cover. Density is highly useful and frequently used for evaluating effects of fire on mortality and reproduction. It is generally straightforward, easily measured and readily understood. Density is not particularly useful in describing community structure and the relationship of species importance to one another, but can be highly valuable in tracking response of individual species to fire. For example, the methodology may measure the number of seedlings, shrubs, or trees per unit area. Response of rhizomatous or suckering plants, such as Western wheatgrass or aspen, may be measured in terms of stems or ramets per unit area with this methodology also. Sprouting shrubs are often tracked using density measurements before and after controlled burning.

Quadrats used to sample density may be small frames or large plots, depending on size and abundance of the species studied. Species to be monitored must initially be defined; size of plots will follow so that the physical sampling does not become cumbersome. Belt transects to measure density of shrubs before and after burning are frequently used in rangeland situations.

3. Frequency of Occurrence. A quantitative expression of the presence or absence of individuals of a species in a population is termed frequency of occurrence, or simply frequency. It is the ratio between the number of sample units that contain a species and the total number of sample units. Its sensitivity in accurately reflecting the population parameters is a direct function of the size of the quadrat used for sampling. Because this method does not measure any plant or plant community attribute directly, its usefulness in fire management has been somewhat limited. Frequency is an integrator that encompasses plant size and shape, density, distributional patterns, number, and a host of other physical attributes. Because of the often severe nature of fire's effects on a plant community, wide swings in frequency values may present problems in both sampling and interpretation of results.

Quadrat sizes used to sample frequency will vary based on vegetative characteristics and the size and distribution of the species being sampled. Frequency values between 20 and 80 percent are considered necessary to both describe the plant's occurrence and detect change over time. Smith and others (1986) describe a nested frequency configuration to sample more than one species in a specific series of transects. If a single frame is used, it is critical that the quadrat size used to collect the initial set of frequency data be used on all subsequent data collections so frequency values are comparable. Statistical analysis cannot be conducted if quadrat sizes have been changed during the course of monitoring.

4. Weight. A measure of the mass of some aspect of an ecosystem may be defined as weight. As used in both ecological and fire literature, the term needs considerable redefining to be understood and thus useful. The term biomass refers to the total weight of living plants and animals above and below ground in a given area at a given time. Because total biomass is obviously beyond normal capabilities to measure, aboveground plant biomass becomes a standard reference in describing a weight aspect of plant communities. It is the total amount of living plants above the ground in a given area at a given time. A virtually synonymous term, standing crop, may be used as both a time and weight indicator of biomass, and refers to the total amount of living plant material in aboveground parts per unit of space at a given time, with particular emphasis on the specified date. Vegetation samplers often seek to measure peak standing crop, that being the maximum amount of living tissue when accumulation is greatest. Aboveground phytomass, which includes dead attached parts, is a standard expression for all organic plant parts in a specified area.

Aboveground phytomass data are useful for fire managers particularly for writing prescriptions and understanding fire behavior. The fuel load, or aboveground phytomass, carries the fire; knowing the precise fuel load prior to burning not only contributes to designing a successful fire, but permits evaluation of observed fire behavior and results of the prescription. In addition, an objective of many prescribed burns is to increase the yield of specific species, or groups of species such as grasses, on a site. Weight data must be collected in order to evaluate success in meeting the objective.

Methods employed to sample weight include clipping of quadrats, estimates, double sampling, use of height-weight curves, and use of capacitance meters. Dense, uniform vegetation requires fewer quadrats for clipping studies. The vegetation is clipped to a specific height in a specified size quadrat, air or oven-dried, and weighed. For total aboveground plant phytomass, everything but unattached litter is clipped. Litter also may be added if total fuel loading is desired. For aboveground plant biomass, also referred to as current year's production or current annual growth, all dead material must be removed from the material clipped. Weights may be estimated in the quadrats, or a double sampling scheme may be employed whereby some plots are clipped and some are estimated, and the actual weights from the clipped plots used to adjust estimated weights in the nonclipped plots. For shrubs, regressions of crown volume, stem lengths, or stem diameters have been used to estimate current annual production. Height-weight curves have been developed by researchers, particularly for individual grass species, based on the relationship of plant height to weight in the various segmented portions. Capacitance meters, which require recalibration for each site and sampling date, rest over vegetation to be sampled and, using the difference between the dielectric constant of herbage and air, estimate weight of underlying vegetation.

Actual clipping or sampling yields good information that can provide more than weight data alone. Fuel moisture content can be calculated; species can be sampled individually or lumped into categories; and botanical composition by weight can be obtained. The attribute is useful not only in determining changes in productivity on an area, but may provide information on kinds and amounts of wild and domestic animal use that may be expected in a given area.

5. Species Composition. Species composition is a term relating the relative abundance of one plant species to another using a common measurement. It is defined as the proportion (percentage) of various species in relation to the total on a given area. It may indicate the relative importance or influence of one species to another in a specific physical setting and is a reflection of structure and hence wildlife habitat. Composition can be determined by weight, cover, number, or other basic variables, and should be reported as such, e.g., percent composition by weight. Generally, it is an attribute arrived at indirectly. The attribute sampled may have been cover, but in order to understand the relationship of species to one another, the relative percents by species are calculated based on cover measurements. In a burn situation, data may be somewhat misleading if caution is not exercised in interpretation. For instance, the entire shrub or tree component may be eliminated, resulting in a dramatic increase in the herbaceous component on a percent composition basis, although no real increase in number of plants or volume of plant material produced may have been realized.

6. Number. Number is the total population of a species or classification category in a delineated unit and is a measure of its abundance. The attribute is most valuable when dealing with small numbers or particulars. In fire situations, actual counts may be important to know for a scarce resource that may be affected by the fire. Threatened or endangered species may be counted in their entirety, or numbers of snags before and after prescribed burns may be noted. Because the attribute does not involve a sample but rather the entire population, no sampling techniques except sheer counting can be described.

7. Height. Height is the vertical measurement of vegetation from the top of the crown to ground level. In herbaceous vegetation, it may be an indirect indication of productivity.

(See 4. Weight) Changes in height and hence changes in structure are some of the most important vegetation characteristics used in determining suitability of areas for various kinds of wildlife. Methods used for measuring height include the Biltmore stick, clinometer, Abney level, and Relaskop.

8. Vigor. Vigor relates to the relative robustness of a plant in comparison to other individuals of the same species and may vary with site, climatic conditions and age. It can be a subjective assessment of the health of individual plants in similar site and growing conditions based on general observations, or it can be more completely defined with some kind of "measurement" of vigor, e.g., references to seed stalk production per plant or unit area, number of tillers produced per plant or unit area, number of leaves or stems, and so forth. Vigor also can be reflected by the size of a plant and its parts in relation to its age and the environment in which it is growing. To be a useful attribute to measure for fire effects, definition of what is to be measured must be made prior to the fire, so that the term maintains a modicum of objectivity. The phenological phase of the species under observation must be the same during each evaluation in order to accurately assess and compare vigor. No matter how carefully measurements are standardized, vigor is considered subjective and is based generally on indirect measurements which may or may not relate to the actual vigor of the plant.

9. Growth Stages and Age Classes. Growth stages are the relative ages of individuals of a species usually expressed in categories. Examples of such categories are seedlings, juvenile (young), mature, and decadent plants. Age classes define in more discrete units the ages of individuals, such as 0 to 5 years, or 6 to 20 years. Age classes may be difficult to determine in herbaceous vegetation, succulents, and any vegetation that does not produce definable growth rings. Both density and frequency measurements outlined above may be made within the parameters of growth stages or age classes, so that the observer may catalogue postburn changes in reproductive capabilities of a site or in effects of the fire on the diversity reflected in different age structures.

10. Phenology. Phenology refers to the timing of various growth and reproductive phases of vegetation. It is based on yearly growth patterns of individual species. A wide variety of phases may be described and then traced for individual species as the growing season progresses (West and Wein 1971). For example, recording the time of initiation of spring growth may be valuable to assess the effects of fire on early growth before and after burning. Other phenological phases frequently recorded include time of blooming, time of seed set, initiation of new terminal bud (signalling the end of seasonal stem or leader growth) and time of dormancy. Mechanics of tracking phenology simply involve delineating the growth phases one is most interested in and then charting them as the season progresses.

11. Injury to Trees. As described earlier in this chapter (B.1.a.), percent crown scorch and percent crown consumption can be good predictors of mortality for many tree species. These can be assessed by estimating or measuring (as with an Abney level) the total length of the tree crown and the length of crown scorched or consumed. Monitoring of damage to tree stems may be needed to better understand the cause of tree death. Height of stem char is measured on all sides of the tree. Depth of char might also be a useful measure of injury on thick barked trees. If bark was consumed or is sloughing off, this should also be noted.

12. Plant Mortality. The cataclysmic effects of fire frequently result in mortality of vegetation. Conversely, in many situations it is of great value to know if plant mortality has been slight following fire.

a. Tagged individuals. A quick and easy way to assess mortality is to tag individuals prior to the fire. Pieces of tin, numbered metal tags, metal stakes, and other nonflammable materials should be used adjacent to the individuals to be checked postburn. A mapped layout of the plot will permit rapid relocation following the fire. It may be necessary to monitor mortality for several years because it may take that long for injured plants to die. Mortality of aboveground portions of shrubs, grasses, forbs, and some species of trees can be visually determined. For many nonsprouting species, death of the main stem, such as of a big sagebrush, is readily apparent, and indicates that the entire plant is dead.

Some species on the burned area are capable of producing vegetative regrowth from buried plant parts. A plant that appears to be dead immediately after the fire may sprout the following growing season. In some cases, new growth must be excavated to determine if a plant has vegetatively reproduced or if seedlings have established.

b. Chemical tests. Chemical tests can be performed to assess the death or survival of individuals of important species.

(1) Tetrazolium. Tetrazolium tests were developed to determine the degree of seed viability, but have been a standard mortality test for range situations. This chemical tests for hydrogen (dehydrogenase) that is released by plant tissues during respiration. Strong, healthy tissues develop a red stain; dead tissues remain their original color. Detailed procedure for testing grass tissue and grass seeds are given in Stanton (1975). The basic procedure is to soak the tissue of interest in a one percent tetrazolium solution, and place it in the dark. Results show up in 5 to 6 hours or overnight. Although this test has long been used, there are associated problems. Procedures for seeds vary by species, and are best performed by experienced analysts. Any sample being tested must be put in a closed container in the dark, because bright sun can affect tetrazolium and cause the same color change as occurs in the presence of dehydrogenase. Results take several hours to appear. On dark tissue, such as Idaho fescue meristems, the color change may not be visible. The interpretation of results can be a problem, because red stain may indicate something other than active metabolism.

(2) Orthotolodiene-peroxide. The chemicals orthotolodiene and peroxide are used sequentially to test for the enzyme peroxidase, found in most living plant cells. This test has been successfully used on trees, and should work well on shrubs. While no documentation of its use on grasses has been found, peroxidase should be present. However, it is not known if peroxidase is present in sufficient quantities in dormant or quiescent grass tissues to stain blue. The basic procedure is described in Ryan (1983). For trees and shrubs, a piece of cambium is extracted with an increment borer (the preferred approach), or exposed by scraping away the bark. An eyedropper is used to cover the sample with a one percent orthotolodiene solution, and then peroxide is applied. Live tissue will turn bright blue within a few moments. A reddish purple color, followed by the appearance of a blue color, also indicates life. A greenish blue color probably means dead tissue. After using this technique for a while, the colors that indicate dead or live cambium become readily recognizable. This test is preferable to the tetrazolium test because it can be used in the field and provides almost immediate results. However, caution is necessary because orthotolodiene has been found to be carcinogenic in laboratory studies. Gloves should be worn as a precaution.

(3) General comments. Metabolic by-products being tested for may not break down until a few days after a fire, even though the plant is dead. The proper location on the plant must be tested to determine mortality. On coniferous trees with living foliage, the cambium should be checked, and also the roots, if much heating occurred at the base of the tree. Trees and shrubs sprout from different locations on stems, root crowns, and roots, and it is these sprouting sites which should be tested to indicate whether the shrub may sprout. Grass crowns should be tested where the buds and reproductive meristems are found.

More than one test may be necessary per plant, because plants can survive some amount of fire damage. Tree cambium requires a test on all sides, and a shrub at several sprouting sites. Unburned meristems and buds of bunchgrasses should be tested at both the center and edges of each plant, and at different depths below the surface if the buds occur below the ground surface.

13. Burn Severity. Burn severity (discussed in II B.6.e), while not an attribute of vegetation, is an exceptionally good predictor of fire effects on vegetation. It indirectly measures the heat pulse below the surface, and provides an indicator of fire impacts on buried plant parts. Burn severity classes can be developed that apply to the type of vegetation and soil organic layer characteristics on the site being investigated. The degree of burn severity can be assigned to one of five classes, including "unburned", "scorched", and "light", "moderate", and "high" severity. Definitions for the latter three classes can be based upon the information in section B.2.c. in this chapter. Burn severity can be described as a percentage of area on plots of a specific size, or related to specific inventory points. Although a qualitative measure, this descriptor can be related to plant mortality, and amount and mode of reproduction, such as by rhizome sprouting or seed germination.

Burn severity classes have been developed for bunchgrass plants. (See VI.B.2.d.(4)) Monitoring the relationship of these classes to postfire mortality or production of specific species can provide a valuable tool for predicting postfire grass response when considering emergency fire rehabilitation, or developing prescriptions for prescribed fire use.

14. Moisture Conditions. The heat regime of a fire depends on the amount and condition of the fuel on the site, how it burns, and the duration of burning. In order to build a database that can be used to predict plant response to fire, moisture conditions at the time of a fire must be documented, because moisture levels are a key regulator of heat release during a fire. See Chapter III.B. and III.D. for a more complete discussion of fuel moisture content and how it is measured.

15. Postfire Weather. Vegetation response to fire can be dramatically affected by postfire weather, particularly in regions with arid or semiarid climates. Knowledge of postfire weather, especially precipitation, can often explain much of the measured or observed variation in postfire effects.

E. Summary

Plant response to fire is a result of the interaction of the behavior and characteristics of a fire with the characteristics of a plant. Plant community response is a product of the responses of all plants on a burned area. The response of an individual species of plant, or plant community, can vary among fires or within different areas of one fire. This is because of variation in fuels, fuel moisture conditions, topography, windspeed, and structure of the plant community itself, causing the heat regime of a fire to vary significantly in time and space. The immediate effects of fire can be modified by postfire weather and animal use. Fire can cause dramatic and immediate changes in vegetation, eliminating some species or causing others to appear where they were not present before the fire. Monitoring techniques that are used to detect trend in vegetative communities are often not appropriate, either because they are not sensitive enough to detect the changes that have occurred, or provide statistically inadequate samples. Fire effects on plants, and plant response to fire treatments are predictable if the principles and processes governing plant response are understood. If burning conditions, the fire treatment, and vegetation response are properly monitored, the fire effects that are observed can be interpreted, and our ability to predict fire effects on plants will increase.

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CHAPTER VII - TERRESTRIAL WILDLIFE AND HABITAT

By Loren Anderson

A. Introduction

Fire effects on terrestrial wildlife and their habitat are addressed in this chapter. Too many variables are involved with fire, wildlife and wildlife habitat to allow a "cookbook" approach. The underlying ecological relationships of vegetation and wildlife are briefly described. A discussion on how fire may subsequently influence those relationships through effects on food, cover, water, and space follows. Considerations for managing and monitoring fire effects on wildlife habitat are also offered.

There is an increasing literature base available regarding fire-wildlife relationships. Four publications the reader may find of particular interest are: Effects of Fire on Birds and Mammals (Bendell 1974); Effects of Fire on Fauna (Lyon et al. 1978); Fire: Its Effects on Plant Succession and Wildlife in the Southwest (Wagle 1981) and Fire in North American Wetland Ecosystems and Fire-Wildlife Relations: an Annotated Bibliography (Kirby et al. 1988). This last document includes an extensive bibliography of all literature on fire-wildlife relations indexed in Wildlife Review 1935 to 1987.

The subject of wildlife, habitat and fire is burdened with generalizations and ambiguities. The definition of wildlife varies from a taxon of invertebrates to an entomologist, a Life Form (Thomas 1979) or Guild (Short 1982) to an ecologist, or a full curl bighorn ram *(Ovis canadensis)* to a hunter. Evaluation of the quality of habitat varies depending on the perspective. Fire is frequently given the anthropomorphic rating of "good" only because fire is considered a natural phenomenon. There is a segment of the public which confuses the fact of fire with the effect of fire and envisions only death, destruction, and loss. Cool burn, hot burn, spring burn, and fall burn are terms that have definitive meaning only to the person using them. Generalizations regarding fire effects on vegetation also can be misleading. Species such as bitterbrush *(Purshia tridentata)* are frequently credited with being so severely harmed by fire that they should be given complete protection. Ultimately, however, many of them are dependent on fire or some similar disturbance (Bunting et al. 1984). Diversity and mosaic are two other commonly used terms that frequently generalize to the point of being meaningless. The widely held assumption that increased edge is always beneficial is not uniformly accurate (Reese and Ratti 1988). As there are no averages in nature (Vogl 1978), there are no generalizations that can stand alone. Terms must be clearly defined and qualified appropriately.

Oversimplification of fire effects commonly occurs when data, knowledge, time or initiative are lacking. Conversely, there is the attitude that if it is not complicated and difficult to understand, it is invalid (Szaro 1986, Vogl 1978). Providing for (as with prescribed fire) and assessing fire impacts on wildlife habitat consistent with general ecological concepts can assist in addressing many complex relationships in a more simplified manner while still retaining a level of validity (Vogl 1978).

B. Principles and Processes of Fire Effects

Adhering to ecological principles and processes is recognized as essential in preserving functional systems (Dubos 1972, Wilson 1985). More accurate effect prediction and assessment, increased assurance of success with prescribed fire, overall cost, and production efficiency are a few of the other benefits derived from thinking and acting as "ecologically" as current knowledge allows (Allen 1987, Chase 1988, Graul and Miller 1984, Savory 1988, Yoakum 1979).

Implicit is the caveat to keep all the pieces (Allen 1987, Chase 1988, Lyon and Marzluff 1984). What we may initially deem insignificant (e.g., mycorrhizal fungi) could be of ultimate importance in maintaining the stability of the ecosystem (Watt 1972, Wilson 1985). How important may lichens *(Rhizoplaca spp.)* be to the nutritional health of species such as the pronghorn *(Antilocapra americana)*? What does the insidious loss of forb diversity and abundance portend? Our imperfect knowledge of ecosystem dynamics and the ramifications of our actions make it imperative we retain the pieces and therefore, in many cases, options. Disruption and loss of indigenous ecosystems have occurred over extensive areas of the west through a combination of inadequate management, fire, and highly competitive exotics such as cheatgrass brome *(Bromus tectorum)* (Wright and Bailey 1982).

1. Floral Response.

a. Ecological basis. Much of the literature regarding fire effects, wildlife and wildlife habitat revolves around successional theory, thus, it is important to understand the concept. Floral succession - that somewhat orderly progression of occupancy by successively higher ecological order plant communities - is one of the primary descriptors of the natural environment. The progression, however, from a "pioneer" stage through various seral stages to that mostly esoteric end point called "climax," is more easily addressed in theory than fact. Although the successional trajectory is often portrayed as following a predictable path and timeframe, there are many who question that view (Bendell 1974, Dubos 1972, Leopold and Darling 1953, MacMahon 1980, Watt 1972). The observation that some species such as gambel oak *(Quercus gambelii)* are successional in one environment and climax in another further complicates the picture (Harper et al. 1985, Whittaker 1975). The "orderliness" of succession we like to envision is commonly disrupted and altered by stochastic events we can neither anticipate nor control (Rosentreter 1989). The apparent controversy does not invalidate successional theory, but it does point out the limitations of our knowledge and the need for being inquisitive and prudent.

b. Structural development. Habitat structure follows successional trends in most plant communities. Short fire intervals tend to maintain or promote early successional conditions typified primarily by herbaceous species and comparatively limited structural diversity. Long fire intervals favor community development along the successional trajectory. This normally results in increased woody species development and greater horizontal and/or vertical structural diversity.

c. Postburn plant community.

(1) For the most part, preburn plant composition and the individual plant species response to fire determine membership of the initial postburn floral community. Understanding plant survival mechanisms (see VI.B., this Guide) is essential for assessing wildfire effects on habitat and in providing for desired prescribed fire effects.

(2) Plants stressed through drought, disease, insect infestations, overgrazing, old age or a combination of these factors are likely to be negatively impacted by burning regardless of how they would respond if healthy (Bunting 1984, DeByle 1988b, Wright and Britton 1982). However, trees that die as a result of fire can provide an important habitat component for certain species.

(3) Herbaceous production increases occur most often on range sites in high Fair or better ecological condition (Bunting 1984, West and Hassan 1985).

(4) Various animal species disseminate seed and can influence subsequent floristic makeup of an area. Pinyon jays *(Gymnorhinus cynanocephalus)* have been credited with the ability to

transport upwards of 30,000 pinyon *(Pinus spp.)* seeds per day up to 6 miles (9.7 kilometers). Both birds and small mammals are considered instrumental in the expansion of pinyon-juniper woodland (Evans 1988). Small mammals, such as chipmunks *(Tamias spp.)*, disperse spores of mycorrhizal fungi into burned areas, an essential component for the establishment and survival of many plant species. (See V.B.c.(4).)

(5) Increases in plant nutrient density, palatability and earlier "greenup" are not unusual occurrences (Bendell 1974, Daubenmire 1968a, Leege and Hickey 1971, Wagle and Kitchen 1972). Earlier greenup of burned areas is largely a function of heat absorption by the dark ash and resultant increases in soil temperature. Reduced soil shading is also a factor. Plants surviving a fire take advantage of higher nitrogen levels provided by the ash and remain greener, more nutritious and more palatable for a longer period of time (Brown 1989). However, this phenomenon is normally shortlived. Commonly, available nitrogen returns to normal levels within two to three growing seasons. Personal observation (Anderson 1983) indicated elevated crude protein levels for 7 years following burning on a bighorn sheep winter/spring range. The extended period of increased protein levels was believed due to subsequent utilization made by bighorns rather than from burning, however.

2. Faunal Response.

a. Ecological basis. Faunal succession follows floral succession, i.e., a given set of vegetational conditions provides habitat for a more or less distinct collection of wildlife species (Bendell 1974, Burger 1979, Dasmann 1978, Evans 1988, Huff et al. 1984, Smith et al. 1984, Wolf and Chapman 1987). Maser et al. (1984), however, pointed out that ecological distinctions between plant communities do not uniformly correlate with differences in animal communities. That apparent inconsistency is explained by the observation that wildlife species most often select habitat on the basis of structure rather than plant species composition. Thus, most of the literature discusses faunal response in terms of general vegetational structure and successional stage.

Fires which set succession back to a grass/forb stage primarily benefit herbivores (vertebrates and invertebrates) and those species for which herbaceous vegetation is desirable for cover. Various vertebrate and invertebrate predators of herbivores also may benefit (Beck and Vogl 1972, Bendell 1974, Hansen 1986, Huff et al. 1984, Lyon and Marzluff 1984, Lyon et al. 1978, McGee 1982). Red foxes *(Vulpes fulva)*, gray foxes *(Urocyon cinereoargenteus)*, and weasels *(Mustela spp.)* are associated with early to mid-successional stages and the ecotones between these stages and climax vegetation communities (Allen 1987). Whitetail deer *(Odocoileus virginianus)*, bobwhite quail *(Colinus virginianus)* and cottontails *(Sylvilagus spp.)* are common to early-mid stages (Dasmann 1978). Fagen (1988) notes a dependency on old growth timber, especially during years of heavy snowfall, by Sitka black-tailed deer *(Odocoileus hemionus sitkensis)*. Muskrats *(Ondatra zibethicus)* are favored by early successional conditions (Allen 1987). The spotted owl *(Strix occidentalis)* is primarily an inhabitant of old growth forests (USDI-Fish and Wildlife Service 1989). As broad as the preceding descriptions may be, they do provide a relative perspective of floral/faunal relationships. Habitat manipulation almost invariably involves efforts to either set back, retard or accelerate plant succession (Burger 1979, Huff et al. 1984, Wolf and Chapman 1987). Most animal species are more cosmopolitan in their use of various successional stages and structural conditions for feeding than for cover, escape or reproduction (Dealy et al. 1981, Maser et al. 1984).

b. Structure.

(1) The consistency with which the value of structure is referred to in the literature (e.g., Allen 1987, Geier and Best 1980, Harris and Marion

1981, McAdoo et al. 1989, McGee 1982, Smith et al. 1984) gives credence to the assumption that structure is possibly the single most important habitat clue to which wildlife responds. Structure may indicate a feeding site for one animal. For another, that same structure may be sought as nesting cover. The perceptual environment differs from species to species and a structural "clue" for one may mean something entirely different to another species (Dubos 1972). Although specific plant species are frequently insignificant (Johnsgard and Rickard 1957, McAdoo et al. 1989), vegetational complexity associated with developing structural diversity is significant, particularly when faunal species richness is the objective (Campbell 1979, Germano and Lawhead 1986, Johnsgard and Rickard 1957, Short 1983, Willson 1974). A variety of "clues" accommodates interspecies variance and automatically denotes a variety of potential habitat niches. Addressing Great Basin habitats in southwestern Utah, Germano and Lawhead (1986) found postbreeding bird diversity significantly correlated with vertical habitat layering and that diversity of both rodents and lizards correlated with horizontal habitat heterogeneity. Birds make more efficient use of habitat volume or vertical space than other classes of wildlife. Although structure is not height limited, Hall (1985) equated herbaceous stubble of 1 to 2 inches (2.5 to 5.1 centimeters) to bare ground for many species that nest and feed on the ground.

(2) The addition of a single structural element to a plant community can greatly enhance faunal species diversity (Germano and Lawhead 1986, Willson 1974). Maser and Gashwiler (1978) reported that the presence of pinyon and juniper trees provided habitat for four additional Life Forms that would otherwise not be present. Conversely, the loss or lack of a single structural component can eliminate some species and be just as lethal as a direct mortality factor (Knopf et al. 1988, Lyon and Marzluff 1984, Wecker 1964).

(3) Habitat and community structure can include dead as well as living components. Large diameter logs provide habitat in the form of travel routes, as well as feeding, nesting, and reproduction (Bartels et al. 1985). Snags are critical nesting and feeding habitat for many species of birds, as well as mammals and amphibians (Neitro et al. 1985). Insectivorous birds that inhabit snags not only harvest insects in recently burned areas, but help regulate populations of insects in adjacent unburned areas, and in the newly developing forest (Wiens 1975 in Neitro et al. 1985).

(4) The interplay between only one or two structural components and subsequent wildlife use is illustrated in the following examples. In a sage *(Artemisia spp.)* shrubsteppe, horned larks *(Eremophila alpestris)* and meadow larks *(Sturnella neglecta)* are favored primarily by a grass stage. Sage *(Amphispiza belli)* and Brewer's sparrows *(Spizella breweri)* are favored by a shrub stage. All four species can exist in nearly a 1:1 ratio in a mixed grass/shrub type. Lark sparrows *(Chondestes grammacus)* appear to benefit more from the mixture than either the grass or shrub stage alone (McAdoo et al. 1989, Rotenberry and Wiens 1978). Rotenberry and Wiens (1978) noted that horned larks replaced sage sparrows as dominant following a burn that eliminated sagebrush. Pronghorn have a somewhat similar potential response to the lark sparrow's. They, too, are benefitted most by a combination of herbaceous and shrub components. Their habitat quality drops rapidly if either component is depauperate or, in the case of sagebrush, too tall or dense (Yoakum 1980).

(5) The Life Form (Thomas 1979, Maser et al. 1984) and Habitat Guild (Short 1982) concepts are organized around the relationship between plant succession and resultant structural conditions, and wildlife. The basic premise is that there are groups of animals with similar ecological requirements that are met by similar successional stages of the plant community. Not unlike successional theory, there are some who question aspects of that approach (Block et al. 1987, Szaro 1986). The Life Form approach to assessing and predicting effects (see Maser et al. 1984) can be of considerable value, however, when large scale fires are addressed. It is important to recognize that within a given Life Form, a full range of species adaptability and species-specific niche requirements may be encountered. The habitat needs of individual species in a given Life Form may have to be scrutinized to assure adequate consideration is given sensitive animals.

c. Species adaptability. Animals that are broadly adaptable behaviorally and in their habitat preferences can accommodate change more efficiently -- they have more options (Dubos 1972, Knopf et al. 1988, Vogl 1978, Watt 1972, Wecker 1964). These species are frequently termed generalists, species of high versatility (Maser et al. 1984) or eurytopic (Knopf 1988). Deer mice *(Peromyscus maniculatus)* and the ubiquitous coyote *(Canis latrans)* are two common generalists. At the opposite end are species that have a narrow range of environmental conditions under which they can survive and flourish. At the extreme are the species that appear on "sensitive" or threatened and endangered species lists. Species of specialized adaptability are commonly termed specialists, of low versatility (Maser et al. 1984), obligates (Kindschy 1986) of a particular habitat component, or stenotopic (Knopf et al. 1988). Hammond's flycatcher *(Empidonax hammondii)* has very narrow food and cover requirements (Maser et al. 1984). The sage grouse *(Centrocercus europhasianus)* and pinyon jay are considered obligates of sagebrush and pinyon-juniper woodland respectively (Kindschy 1986, Hardy 1945). Allen (1987) notes that many furbearers of forested and wetland cover types have specific habitat requirements and are less resilient in adapting to habitat modifications. Specialists commonly can be eliminated by loss of a single habitat component. Species of intermediate adaptability such as robins *(Turdus migratorius)* and red-wing blackbirds *(Agelaius phoeniceus)* are referred to as mesotopic (Knopf et al. 1988). Knopf et al. (1988) separated riparian avifauna into eurytopic, mesotopic, and stenotopic guilds to accommodate variations in habitat sensitivity.

d. Food and cover. The two most visually obvious determinants of habitat suitability are food and cover.

(1) Dense timber travel lanes are frequently preferred by elk *(Cervus elaphus)*. Sagebrush can reach heights and densities that inhibit or prevent pronghorn movement. Voles *(Microtus montanus; Clethrionomys gapperi)* require certain litter layer or woody debris habitat components. Birds require various structural conditions for nest sites, hunting and song perches. Some species (e.g., the white-tail deer) select for denser woody vegetation. Species such as bighorn sheep or pronghorn normally select against it (Lyon et al. 1978, McGee 1982, Yoakum 1980).

(2) One species may consume primarily grass and forbs (e. g., grasshoppers or elk), another mostly forbs and shrubs (e.g., sage grouse or pronghorn) and yet another, such as turkey *(Meleagris gallopavo)*, may make heavy use of mast. Hobbs (1989) provides a strong case that the habitat mule deer *(Odocoileus hemionus)* need for thermal cover correlates well with the nutritional plane of the animal. Hobbs and others have noted the inherent value of forage diversity and availability. Hobbs and Spowart (1984) found substantially improved winter diet quality for both deer and bighorn sheep as a result of burning although there were only relatively small changes in the quality of individual forages. They attributed the diet quality increase to improved availability of forage items and enhanced forage selection opportunities.

It is commonly assumed that increased herbaceous production automatically occurs after burning.

There is sufficient documentation, however, indicating production increases are not a foregone conclusion (Peek et al. 1979, Yeo 1981 and others). Certainly, some highly valued food items can be eliminated if burning is too frequent. For example, McCulloch et al. (1965), determined that Gambel oak produce very few acorns before reaching 2 inches dbh (diameter at breast height) (5.1 centimeters) and that maximum production of mast probably does not occur until healthy stems are 12 to 14 inches (30 to 36 cm) in diameter. Management of Gambel oak for both mast production and deer winter browse may not be possible on a given site.

(3) The differential effect fire has on wildlife is typically noted in a realignment of species as some species become favored over others as a result of changes in abundance of food and cover. For some classes of wildlife (e.g., birds and small mammals), stability of total numbers has beennoted, even though species composition changed (Lyon and Marzluff 1984, Lyon et al. 1978, McGee 1982). Bendell (1974) explained changes in the kind and frequency of parasitic infections following burning as likely a result of alteration of habitat structure and cover favored by intermediate hosts. He noted a trend towards more species of parasites in greater frequency of infection with longer time after burning. He stressed, however, that there is a broad response range and that any blanket statement regarding fire effects on parasites must be qualified.

(4) Various efforts have been made to "codify" vegetational requirements for a few species. Yoakum (1980) outlines some fairly specific vegetational conditions that constitute quality pronghorn range in shrub-grasslands. Autenrieth et al. (1982) and others have developed similar recommendations for sage grouse. Allen (1987) and Parker et al. (1983) note vegetational criteria for pine marten *(Martes americana)* and lynx *(Lynx canadensis)* respectively. A growing number of plant community/wildlife association listings are available. Most are built around the Life Form (Thomas 1979) or Habitat Guild (Short 1983) concepts that relate wildlife use to structural conditions. Maser et al. (1978) address wildlife species of the western juniper type; Maser et al. (1984), fauna of the Northern Great Basin; Allen (1987), furbearers; Harper et al. (1985), wildlife of the oak brush type, and Thomas (1979) and Brown (1985) address coniferous forest wildlife. Brown (1985) is somewhat unique in the level with which fish and amphibians are addressed.

e. Influence of time.

(1) Each wildlife species has a unique reaction to fire and the subsequent ecological changes caused by fire. Some species exhibit an almost immediate reaction. For others, behavioral time lags and site tenacity may extend the response time nearly indefinitely (Wiens et al. 1986). Eurytopic or generalist species can likely accommodate the change more efficiently than stenotopic or obligate species (Lyon et al. 1978).

(2) Immediate postburn effect assessments can be a poor reflection of long-term animal use. For example, McGee (1982) noted that mountain voles *(Microtus montanus)* could not sustain populations on severe fall burns. Repopulation was contingent on the subsequent development of an adequate herbaceous mulch layer. Red-back voles *(Clethrionomys gapperi)* require woody cover and moisture conditions (Getz 1968) that may be eliminated by fire. Four years following a fire, however, red-back voles may be the most common mammal present (Moore 1989). A decadent stand of bitterbrush under a pinyon/juniper overstory may be providing valuable deer forage. Burning that habitat may be the only hope for assuring <u>any</u> bitterbrush is available for some future generation of deer (Bunting et al. 1984). Huff et al. (1985) found that the highest diversity of birds occurred in a 19-year-old forest in the Olympic Mountains of Washington. Moore (1989) stated that burned forest 5 to 10-years-old becomes prime habitat for chipmunks, possibly even better than the original forest. He associated improved chipmunk habitat with development of complex shrub layers.

Bunting et al. (1984) observed managers must not be so concerned with the short-term effects that they lose sight of the future needs of species. A fire-damaged tree, for example, may not die for several years, but then provides important habitat for cavity nesting birds for a long time. The species that use the snag change over time as the snag decomposes. In many more years the snag falls, becoming a forest floor log that is habitat for many other species.

f. Extrinsic/intrinsic influences. Both extrinsic and intrinsic factors direct individual species response to fire effects. Extrinsic factors include such items as food, cover, water, predators, and other elements of the external environment. A noticeable population increase, decrease, or shift in use patterns would indicate burning had modified some extrinsic factor(s), creating or removing a limiting element, or creating a more desirable (not necessarily required) condition. Physiological, behavioral andgenetic characteristics are intrinsic factors that play a large role in determining how a species responds to a burned area. When no particular response occurs it may be assumed the impact of fire was inconsequential or that there were overriding intrinsic factors that inhibited or prevented a response (Bendell 1974, Moen 1979, Wolf and Chapman 1987). The presence or absence of one species can influence the presence, absence or habitat utilization pattern of another (Bendell 1974, Peek et al. 1984). Extrinsic and intrinsic factors are not mutually exclusive and the seemingly endless combinations of the two can easily make an absolute determination of fire effects virtually impossible in many cases. Peek et al. (1984) found positive results from fire on seven different bighorn sheep ranges. They indicated however, that in each case, definite proof was lacking because they were unable to isolate the effects of fire from other potential factors. Wiens et al. (1986) also noted confounding elements associated with trying to evaluate the effects of small-scale burning on shrubsteppe avifauna.

3. Burn Characteristics: Influence on Potential Faunal Response.

a. Size of burned area. A number of small burns produces more edge than a single large burn. More edge is commonly assumed to provide more benefit to more species of wildlife (Odum 1966, Thomas et al. 1979, and others). When particular species are considered, however, the picture is not that clear. A number of 5 to 10-acre (2 to 4-hectare) burns in a pinyon-juniper/ sagebrush-bunchgrass type might be relished by mule deer but be of little or no value to pronghorn. Some species, such as the western flycatcher *(Empidonax difficilis)* and brown creeper *(Certhia americana)*, are seldom found associated with edge and actually may be harmed if edge is increased at the expense of adequate forest interior (Rosenberg and Raphael 1986). The creation of fragmented habitat for some species is a potential concern, especially with large scale fires. Burned areas are a considerable attraction for many herbivores. Larger species such as elk, moose *(Alces alces)*, bighorn sheep, domestic cattle *(Bos taurus)*, and wild horses *(Equus caballus)* are capable of overutilizing burns of insufficient size to accommodate their demand.

b. Burned area configuration.

(1) Patchy or irregular burns can enhance habitat diversity, particularly in an area with only one or a few communities all in the same structural condition. Increased diversity and resultant increases in edge effect makes more niches available for partitioning. Edge length, width, configuration, contrast, and stand size largely determine the degree of benefits. It must be recognized that diversity and edge cannot be increased indefinitely. Beyond some threshold, the pieces become sufficiently small and mixed that they assume a sameness or homogeneity (Thomas et al. 1979). Also, increased fragmentation of habitat components caused by maximizing edge can eliminate those species requiring larger tracts or that inhabit stand interiors (Reese and Ratti 1988). The amount and type of diversity sought with prescribed fire is determined by the management goals and objectives. Maximum diversity provides for species richness but is incompatible with an objective to maximize a particular species. The reader is referred to "Edges" by Thomas et al. (1979) for one of the more concise and understandable treatments of diversity and edge. "Edge Effect: a Concept Under Scrutiny" (Reese and Ratti 1988) is a good companion treatise offering qualifying considerations of the edge concept.

(2) Areal extent, composition, and orientation of habitat components (food, cover, water, space) determines habitat suitability for individual species and/or groups of species. Juxtaposition, interspersion, complexity, diversity, and mosaic are terms commonly used to reflect the physical mix and patterning of edges, structural components, plant communities, and seral stages. These somewhat generic terms give a relative idea of the variety and positioning of resources within a given area. Nearly mystical qualities have been attached to these terms and they are frequently (and improperly) used without qualification. The ideal mosaic for a soil-surface invertebrate obviously sayslittle about the optimum mosaic for sage grouse. A sage grouse habitat mosaic has no relationship to a mosaic promoting coniferous forest avifauna species richness. A "good" mosaic is meaningful only within the context of a specific management goal or objective (Thomas et al. 1979).

(3) The scale of the postburn configuration or mosaic is a major consideration. Whether the components (structure, cover, openings) are measured in square inches, acres or miles can dictate which species benefit. Thomas et al. (1979) suggest that wildlife species richness should be approaching maximum in rangeland settings where "average" habitat size is approximately 200 acres (81 hectares). It is recognized that habitat requirements of individual species determine the relative potential benefit of a particular size burn where species richness is not the primary objective.

A seldom discussed aspect of habitat quality is that of habitat fragmentation (Reese and Ratti 1988, Rosentreter 1989). Small isolated islands of shrubs and trees following prescribed or wildfire are examples of fragmented habitat. Many wildlife species exhibit high extinction rates in fragmented habitat (Wilcox 1980). Fragmented habitat fails to provide areal extent and linkages between and among components that are implied with "quality" mosaic, juxtaposition, interspersion and diversity for a given species or collection of species. Adequate linkage of habitat components (e.g., a stringer of cover connecting larger areas of escape cover) is a determining factor for many species. As with other wildlife/habitat considerations, some species are favored over others by a particular mosaic or juxtaposition of elements and a few species may be eliminated entirely.

c. Burned area location. The location of a burn relative to animal use patterns can have a major influence on subsequent use. The proximity of propagules or potential inhabitants has an influence on what species may occur on a burn site. Species mobility also plays a part. A young bighorn ram may travel miles, a shrew *(Sorex spp.)*, hardly any. Whether potential inhabitants can see, smell or be expected to wander across a burned area may dictate the presence or absence of a particular species. An otherwise "excellent" burn (prescribed or wild) that is too distant from traditional use areas (e.g., bighorn) may not be utilized in any reasonable timeframe. Proximity of the burn to a critical habitat component such as water or cover also determines use. Sage grouse exhibit a reluctance to use water sources devoid of adequate surrounding cover. In contrast, pronghorn generally avoid water sources screened with tall dense vegetation. Loss of a critical habitat component and how soon - if ever - that element is replaced may be of paramount importance. Burn location can strongly influence ultimate vegetational establishment. A small stand of Douglas-fir *(Pseudotsuga menziesii)* on a steep south exposure may - from a practical standpoint - never regenerate whereas a similar stand on a more moist north exposure may be restocked in comparatively few years. Slope, aspect, and elevation affect snow deposition, snow crusting, thermal patterns, and wind conditions on burned areas. All of these factors have a bearing on habitat quality for a given species.

d. Completeness of burn. A fire of low burn severity and low fireline intensity, which consumes comparatively little of the existing plant community, may have no perceptible impact on wildlife. However, the high severity, high intensity fire can significantly alter habitat makeup, sometimes for an extended period of time. The first example may have influenced plant community succession very little. The second could result in a major adjustment of seral position potentially resulting in significant changes in structure, cover and the forage base. It is common to see both examples and many variations of the two on a single fire.

e. Timing of burn. Timing of a fire relative to plant phenology is a primary factor dictating the postburn plant community makeup. Perennial herbaceous species, for example, are most resistant if burned when completely dormant (Britton 1984, Bunting 1984). The size and phenological stage of coniferous tree buds influences their resistance to fire. Large buds such as on ponderosa pine *(Pinus ponderosa)* that have scaled out are more resistant than small and/or scaleless buds (Ryan 1988). (SeeVI.B.1.a., this Guide.) Negative impacts may occur if a vital habitat element for a species is burned at the "wrong" time. For example, a fall burn that consumes bighorn sheep winter range could be disastrous, at least in the short term. If that fire occurred early enough in the spring for regrowth, it may be beneficial (Peek et al. 1984). The effect of fire on birds nesting in residual herbaceous vegetation can vary markedly depending on whether the fire occurred before, during, or after nesting activities were completed.

4. Direct Mortality. Fire related mortality is popularly considered insignificant and generally ignored. That generalization can be very misleading - at least on a site specific basis. There is no reason to believe various wildlife species have some superior capability to predict fire behavior or to locate safety zones through dense superheated smoke. Animals do die, apparently, most often through suffocation (Lawrence 1966). At times, the number may be high. Quinn (1979) reported that an intense burn eliminated all small mammal species but the kangaroo rat *(Dipodomys heermanni)*. Nelson (1973, p. 139) relates a very graphic eyewitness account of large numbers of dead and dying buffalo *(Bison bison)* that were caught in a prairie fire in the early 1800's. There are indications that severe burning can cause potentially long-term reductions in some insects and other invertebrates of the soil surface layer (Lyon et al. 1978). A long-term loss of these invertebrates could be of particular significance in the altered and frequently truncated ecosystems affected by man. Highly mobile species and species that can escape underground or into rock crevices are least subject to direct mortality (Beck and Vogl 1972, Lawrence 1966, McGee 1982, Starkey 1985). Direct mortality is unlikely to have much effect on many species if the entire population or range of those species are considered. In a particular geographic setting, however, the potential significance of that loss should at least be considered.

5. Miscellaneous Considerations.

a. Noxious weeds and exotic plant species. Noxious weeds and exotic plant species are an increasing concern. Any wild or prescribed fire occurring or planned in areas subject to noxious plant invasion should be evaluated from that standpoint.

b. Human effects on subsequent use. Fences, roads, human activity and other similar factors, either on or off-site, can significantly influence subsequent wildlife use.

c. Livestock use. Burns are an attraction to many animals and livestock are no exception. Fires occurring on slopes less than about 35 to 40 percent may be subjected to heavy use by all classes of livestock. Horses may make excessive use of burns regardless of location. Livestock can easily influence potential wildlife values of a burned area by altering plan responses, reducing herbaceous habitat structure, removing forage, and merely by their presence. Control and management of livestock is essential.

d. Snow crusting. Wind crusting of snow is a common problem on some deer winter ranges. Carpenter (1976) documented that a sagebrush canopy disrupts wind crusting in addition to providing frequently melted out areas around larger plants. Sagebrush stands approaching 20 inches (51 centimeters) in height have been found to collect up to 1 inch (2.5 centimeters) more water in the form of snow than open grassland (Hutchison 1965). Haupt (1979) noted that small burns in climax coniferous forests accumulate more snow than unburned forest. He found a similar result on larger burns if there was residual tree cover. However, if burn size exceeded four times the height of surrounding tree cover, snow accumulation could be reduced through wind scour.

e. Indirect effects. The possibility of remote or very indirect wildlife-fire effect relationships should be considered. For example, there is little question the current concern and management efforts directed toward aspen *(Populus tremuloides)* stand rejuvenation are valid. Robb (1987) however, found that 85 to 90 percent of all gastropods infected with bighorn sheep lungworm *(Protostrongylus spp.)* larvae in her study area, resided in aspen stands and aspen edges. Could fire in an aspen type in one area have potential implications for bighorn monitored in another? As difficult as it may be, effort must be extended to look at the whole system; to think and act within an ecological perspective (Savory 1988).

f. Stochastic events. Climatic and weather related events such as drought, abnormally high precipitation, shifts in precipitation regimen, and unusually hot or cold temperatures can have a marked effect on the interpretation of fire effects. Large populations of insects such as the Mormon cricket *(Anabrus simplex)* can consume remarkable amounts of vegetation in a relatively short time thus clouding fire effects evaluations.

C. Resource Management Considerations

Understanding existing management goals and objectives for an area is essential. It is recognized that site specific objectives may not be in place to adequately address every wildfire situation. Some objectives may even be mutually exclusive. It is recommended that a familiarity with fire terminology be developed to facilitate communication on wildfires and in planning for prescribed fire. The Fire Effects Information System (F.E.I.S.) contains a wealth of information and is an invaluable aid in predicting and assessing fire effects. (See XII.D.4., this Guide.)

Every fire is a "wildlife" burn. Only through careful consideration, accommodation and management of factors influencing the results of fire can wildlife goals and objectives hope to be met.

1. Define Terrestrial Wildlife Habitat Goals and Objectives.

2. Standard Considerations for Fire Suppression.

a. Protection of habitat improvement projects. Protect habitat improvement projects such as guzzlers, nest structures, browse plantations, fences, and recent prescribed burns.

b. Water. Water quality and flow considerations are of vital importance to many species of wildlife. Efforts to protect water include, but are not limited to, the following:

(1) Prohibiting the washing or rinsing of any container or equipment containing potentially harmful substances in or near any spring, stream, pond or lake. Containers would include such items as helicopter buckets, retardant tanks, engine tanks, backpack pumps, and other such items. Potentially harmful substances include, but are not limited to, wet water, foaming agents, and petrochemicals in any form.

(2) Avoid the dropping or spraying of retardant, wet water, or foaming agents directly on, or immediate to, wetlands, springs, streams, ponds or lakes.

(3) Avoid alteration or damming of stream courses.

c. Control of vehicle and heavy equipment use.

(1) Restrict travel to existing roads to the extent possible.

(2) Avoid any travel in or across streams or wet meadows, or through unique or limited habitats.

(3) Physically close and rehabilitate all firelines that potentially offer ORV access.

d. Control of aircraft use.

(1) Establish low-level flight routes that avoid important habitat areas such as bighorn sheep summer range, raptor nest sites, and waterfowl nesting areas.

(2) Avoid harassment of big game or other species of wildlife.

e. Wildlife barrier management.

(1) Bulldozer-line windrows through timber or heavy brush should be broken up, lopped and scattered. At a minimum, they should be breached at all drainage crossings and at intervals between drainages to facilitate movement of big game.

(2) Where trees have been slashed around ponds or other bodies of water used to facilitate bucket drops, the slash should be reduced to a depth of no more than 18 inches (46 centimeters) and/or travel lanes cut through for wildlife access to the water.

3. Species Habitat Requirements.

a. Structure. Species structural requirements for feeding, hiding cover, reproductive cover, thermal cover and ease of movement should be a primary consideration. It is important to keep in mind that structure per se is not defined by height (i.e., a short-grass meadow is just as much a structural component as an impressive stand of grand fir). A number of species-specific habitat management guides are available which address structural requirements. Life Form, Habitat Guild, and other similar listings also can be of assistance.

(1) How do current structural conditions compare with the perceived optimum for a featured species or management for species richness? The areal extent, shape, height, age, density, and orientation of structural components, and the necessary linkages between and among those components should be addressed. If structural conditions of vegetation are at or near the perceived optimum and in a healthy condition, fire would not be of benefit. It is important to remember the value of nonliving structural components such as snags and downed logs. Fire may have a positive effect on these features, such as when fire creates snags, or a negative effect when a severe fire consumes most downed woody debris.

(2) How adequate are structural conditions adjacent to the proposed burn or wildfire? A number of species (e.g., elk, mule deer, and others) more readily use burned areas if their cover requirements are met in close proximity to the burned area. In some vegetation types (e.g., sagebrush-grass), the "best" habitat frequently burns because it has the highest fine fuel loading.

(3) What structural conditions and orientation are desired within the fire area? For example, some species may require certain cover characteristics along drainage courses, from drainage courses to ridge lines and/or along ridgelines to make efficient use of burned areas.

(4) What postburn timelags for structural development are tolerable? If no timelag is acceptable, that structural condition should be afforded protection from fire. Threatened or Endangered species, species classed as "sensitive," and species that are obligates of late seral conditions are species likely intolerant of any habitat loss duration if that loss is of sufficient size. The opportunity for accepting short-term structural losses for long-term gain should be explored, however.

Factors influencing plant response time include:

(a) Plant survival mechanism;

(b) Plant health;

(c) Phenological stage;

(d) Preburn and postburn management;

(e) Stochastic events such as drought, torrential rains, insect or disease outbreaks and other unpredictable and largely uncontrollable occurrences.

b. Behavior. Behavioral attributes may influence species response. Nominal home range size, territory, interspecific compatibility, sensitivity to human disturbance, site fidelity, preference for open vistas (e.g., antelope, bighorn sheep), preference for denser cover (e.g., white-tail deer, ruffed grouse), and other aspects of behavior can have a profound effect.

c. Food habits. Food habits must be considered. An animal that depends on a comparatively few select food items is more sensitive to fire effects than a species with more cosmopolitan dietary requirements. The following is oriented toward herbivorous species with the understanding that predators and scavengers are indirectly affected through their prey base.

(1) What shift in available food items may occur as a result of burning? Fire of sufficient fireline intensity and/or burn severity could benefit grazers at the expense of browsers. Browsers could benefit from a fire in a habitat type such as oak brush but species dependent on mast produced by that oakbrush may be negatively impacted for many years.

(2) Will food items be available when the species requires them? For example, a fall burn could eliminate critically needed forage for wintering herbivores. A fire on that same site early enough in the spring to promote substantial regrowth, could be beneficial.

(3) Is fire-sensitive vegetation involved in the food base? Long-term negative impacts can be incurred if an animal is dependent on a fire-sensitive species that burns. The opportunity, however, for accepting short-term forage loss for long-term enhancement of the food base should be considered.

d. Water availability. Free water availability dictates the presence or absence of many species following a fire.

(1) Is water present on or close to the burned area? Some species will travel miles for water. Others, however, need it immediately available.

(2) Is the water present yearlong or on a seasonal basis?

(3) What cover characteristics are present immediate to the water? Some species (e.g., sage grouse, white-tail deer) show a reluctance to use a water source deficient of adjacent cover. Other species, such as antelope, prefer good visibility.

(4) The potential adverse effects of fire on water quality -- both onsite and off-site -- should be addressed. Water quality can have a major influence on food chain relationships.

4. Miscellaneous Considerations.

a. Size of burned area. Is the burn of sufficient size to accommodate the forage demand of large herbivores? Deer, elk, bighorn sheep, livestock, and a number of other species are capable of making excessive use of burned areas that are of insufficient size. Many of these species exhibit a strong affinity for burned areas. Options to consider include:

(1) Burn additional similar size areas;

(2) Increase the size of the prescribed fire unit;

(3) Protect burned areas with fencing or by herding (as with domestic sheep);

(4) Reduce numbers of animals generating the demand;

(5) Unless otherwise accommodated by management, conduct prescribed burns for wildlife on cattle allotments on areas not readily accessible to livestock, such as steep slopes.

b. Noxious weeds and exotic plants. Potential for noxious weed and/or exotic plant invasion should be addressed and management adjusted as necessary.

c. Potential changes in human access. Is the burned area in or near zones of human activity? Human activity associated with roads, campgrounds, ORV use, and other sources can strongly influence the presence, absence or habitat use efficiency of many species. Bulldozed firelines may create undesirable access. Burned areas may attract recreational use by snowmachiners, skiers and others. Closures to protect wildlife may be necessary.

d. Snow. Will the habitat quality of some species be altered by changes in snow deposition and crusting factors as a result of burning?

e. Snags.

(1) Adequate snag protection should be incorporated into prescribed fire plans. The opportunity to protect snags under certain wildfire situations may also be present. Damage to snags from fire can be limited by such measures as hand pulling or machine piling fuel away from their base, and applying fire retardant foams around the bottom and along the bottom part of important snags. Leave some living trees with broken tops that will eventually become snags, and provide raptor nesting habitat in the interim.

(2) When conducting postfire salvage logging, leave some dead or dying trees to become future wildlife trees.

f. Downed logs. When prescribed burning, ensure that a certain number of logs of a minimum specified diameter are left onsite. If broadcast burning, prescribe moisture contents high enough in the larger diameter material that it does not burn. If pile burning, leave logs out of the piles.

D. Methods to Monitor Fire Effects

This section views some considerations for monitoring fire effects rather than specific techniques.

1. Animal Population Changes. It is suggested that monitoring of animal population changes be avoided unless there are overriding reasons to do so. As noted previously, intrinsic factors and synergistic relationships between and among plants and animals can easily confound cause/effect assessment of fire effects on populations. The sophistication of study design and execution, time and cost of such studies is normally beyond field office capabilities. Contract studies or support of cooperating agencies, organizations or institutions should be investigated when such information is required.

2. Objectives.

a. Well-defined and measurable. Well-defined, measurable objectives that describe essential plant community characteristics greatly facilitate monitoring. For example, an objective to increase herbaceous production for elk forage would require much more monitoring effort if it were written to increase specific plant species rather than the total production of herbaceous plants elk utilize. Yoakum (1980) indicates a variety of forbs is required on quality pronghorn range. Specific forbs are apparently of little importance. An objective to increase the variety of forbs is much simpler to monitor than one that requires the absolute determination of plant species. The same consideration holds for objectives relating to structure.

b. Expressed in absolute terms. Objectives expressed as percent composition are essentially meaningless unless accompanied by an element addressing ground cover, pounds production, or some other absolute. For example, 50 percent grass and 50 percent sagebrush on a rangeland could mean anything from one grass plant and one sagebrush plant to 500 pounds (227 kilograms) production of grass and 500 pounds of sagebrush.

3. Monitoring Level. The level of monitoring required needs to be defined. There is a temptation to set up monitoring procedures of detail or sophistication that are unwarranted for management purposes. Monitoring procedures should match the issue sensitivity. For example, if only an index of bighorn sheep preference for a burned area is needed, a simple grazed-ungrazed plant transect inside and outside of the burn may suffice. An effort that addressed factors such as comparative production by plant species, chemical analysis of forage, and pounds of various plant species utilized by bighorns inside and outside the burn would be "overkill." The latter approach might be appropriate, however, if bighorn sheep use of a burn was a controversial issue or some detailed information regarding their ecology was needed.

4. Consistency of Technique. Maintaining consistency of technique is essential. The plot that is estimated one year, measured the next, and perhaps photographed the next, does not allow for any meaningful comparisons among years.

5. Observations. The value of observational information should not be overlooked. Frequently, this subjective information provides the critical links with more formal data to clarify what actually transpired due to fire or whether the apparent effects were due to another reason. Anyone who has had an occasion to be on the site, either during or following the fire, is a potential source of information. Observations should be properly documented and filed. Subjective information alone seldom provides the confidence needed to make politically sensitive management decisions.

6. Overall Effect. Addressing overall effect of fire on wildlife for a given area that has burned is most easily approached by going from the general to the specific.

a. What was the original successional stage and structural condition?

b. What Life Forms or Guilds were associated with those preburn conditions?

c. What will the new successional stage and structural makeup be?

d. What Life Forms or Guilds will be favored by the new conditions?

e. How were species of management or public interest affected?

f. How may any obligate or otherwise sensitive species have been affected?

7. Sources. An increasing number of sources for assistance in developing monitoring programs are available. Only a few of the more readily available are listed here. Inventory and Monitoring of Wildlife Habitat (Cooperrider et al. 1986) contains a wealth of information. Species specific habitat guidelines have been developed for pronghorn (Yoakum 1980), sage grouse (Autenrieth et al. 1982) and a number of other animals. Thomas et al. (1979) outlines simple procedures for measuring and evaluating edge diversity. Estimating Wildlife Habitat Variables (Hays et al. 1981) is an excellent field-oriented guide that not only addresses procedures but offers estimates of time and cost involved with various techniques. Chapters VI and X of this Guide should be referred to for further considerations and direction on monitoring and evaluation.

E. Summary

Fire is a shock - frequently, nearly instantaneous - to the ecological setting involved (Huff et al. 1984, Lyon et al. 1978). Some wildlife species are able to adapt to the rapid change in environment and some cannot (Lyon and Marzluff 1984, Parker et al. 1983, Rotenberry and Wiens 1978, Wecker 1964). The habitat for some species is greatly improved, while for others it may be degraded if not eliminated, and there will be endless variation in between (Beck and Vogl 1972, Bendell 1974, Evans 1988, McGee 1982, Wolf and Chapman 1987). No fire - either wild or prescribed - is uniformly "good" or "bad." Effects are differentially imposed.

A righteous attempt at providing for desired fire effects through prescribed burning or evaluating wildfire effects on wildlife and its habitat requires an integrated effort of disciplines. An appreciation of the historical perspective can be invaluable. Contributions by plant or fire ecologists are essential - individuals may have the talent but not the title. Input from those with a thorough knowledge of fire is certainly important. Postburn management is <u>absolutely</u> <u>critical</u>. Obtaining good management necessarily requires close coordination with and commitment from specialists in range, forestry, recreation, and others.

Without adequate monitoring and evaluation, little knowledge can be gained and even less, shared.

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CHAPTER VIII - CULTURAL RESOURCES

By Dr. Richard C. Hanes

A. Introduction

Cultural resources include a range of different resource types. These resources include locations containing archaeological and architectural remains resulting from human activity in the prehistoric and historic periods; and locations of continued traditional use activities, primarily associated with areas of religious or traditional subsistence concern to Native Americans.

Prehistoric archaeological sites include artifact scatters at locations where tools were made, a series of depressions in the soil surface representing a pithouse village, pueblo ruins, and rock art panels where figures were carved or painted centuries ago. Historic period sites pertain to more recent activities. Examples include old cabins, early homesteads, trails, battlegrounds, early mining remains, and logging camps. The second category of cultural resources noted above includes traditional areas where soil or plants are collected, or ceremonies conducted for secular or religious purposes. In some cases, these areas coincide with locations where archaeological remains are found, but they are just as likely to be a spring, mountain, or other geographic feature not containing tangible reminders of past activities.

B. Principles and Processes of Fire Effects

Particularly important information concerning fire effects on cultural resource values has been developed by Peter Pilles (1982) of the U.S. Forest Service and by National Park Service programs (Kelly and Mayberry 1980). Much of the following information draws from those sources as well as others noted.

It is difficult to accurately assess the effects of prescribed fires or wildfires on cultural resources. One important factor is the widely varying responses of vegetation and soils to fire within the same burn area, or under the same prescription in different burn areas. At most cultural resource sites on public lands, artifacts are distributed on the soil surface, or buried within the soil, even when historic or prehistoric structures are also present. The amount of surface and subsurface heating depends upon the peak temperatures reached and the duration of all phases of combustion. The amount of subsurface heating is a function of a number of variables, including soil moisture content and coarseness, amount and distribution of woody fuels, occurrence of duff layer or other accumulations of organic litter, weather conditions, and fuel and duff moisture content. (See Chapter III.B.6., this Guide for a more detailed discussion of burn severity.) Fire behavior studies have shown that no clear relationship is currently known between surface temperatures attained in a fire and temperatures conducted below the soil surface. It appears that only a small percentage of surface heat penetrates the soil deposits, because soil temperature during a fire can decrease dramatically in just a few inches of depth. Smoldering and glowing combustion of both surface and subsurface fuels can be important contributors to heating of buried artifacts.

Few formal studies of the effects of fire on archaeological sites have been performed. Studies in the 1970's involved the Radio Fire in the Coconino National Forest near Flagstaff, Arizona (Pilles 1982),the La Mesa Fire near Bandelier National Monument, New Mexico (Armistead 1981; Traylor 1981), the Moccasin Mesa Fire at Mesa Verde National Park, Colorado (Switzer 1974), and the Dutton Point Fire at Grand Canyon National Park (Jones and Euler 1986). These were all wildfires, not prescribed fires. A handful of studies and recorded observations have been conducted in the 1980's, including experimentation with prescribed fires in the Cleveland National Forest in California (Pidanick 1982; Welch and Gonzalez 1982). The literature addressing the effects of fire and heat on archaeological materials is still very meager in quantity and largely unpublished. However, enough information exists to indicate that the effects are variable, depending upon the material that is heated and the level of heating attained.

Arguments are made that many of the prehistoric cultural resource sites have been exposed to fires, perhaps repeatedly, in the past. So why the concern over fires today except in the more obvious cases where perishable structures are involved? In addition to the rebuttal that we are likely dealing with cumulative information loss from repeated impacts, other important factors must be considered, including relative burn severity between historic and prehistoric fires, recent surface exposure of some ancient sites, and cumulative changes in erosion patterns.

A growing body of information, based on early historic accounts, ethnographic studies, and field fire history studies suggest fire was much more prevalent prior to implementation of fire suppression policies earlier this century. The higher frequency of fires earlier in time is at least partly attributed to aboriginal burning practices (Lewis 1973, 1985; Barrett and Arno 1982; Arno 1985; Gruell 1985). In areas where fires occurred frequently, the exclusion of fire has often led to levels of fuels higher than would have "naturally" occurred. Fires which occur now potentially have greater impacts on cultural resources because of the increased amounts of heat that can be released during burning.

Archaeological sites often contain a variety of cultural-related materials, including stone, bone, shell, ceramics, metal, glass, wood, leather, and other substances. The resultant combination depends on the technologies of the inhabitants, the specific activities being performed at that location, and the preservational characteristics of the setting.

1. Potential for Physical Damage to Materials. The effects of fire on these various materials, whether buried or on the surface, can vary significantly because of different inherent properties and locations where materials occur.

a. Stone. A commonly observed result of intense, high temperature fires is the occurrence of heat damaged stone artifacts. An example of such damage was presented by a very hot prescribed fire in California's chamise chaparral that resulted in temperatures over 700 F (371 C) at one location where archaeological materials had been placed to assess the effects (Pidanick 1982). Some chert chipped stone implements had shattered; other artifacts including obsidian items and grinding stones were heavily smudged. Artifacts placed at other locations where temperatures never reached 400 F (204 C) were unbroken and only lightly smudged. Laboratory experiments have demonstrated that the crystalline structure of many forms of silica-rich stone changes when heated above 700 F (371 C) (Purdy and Brooks 1971; Mandeville

1973). Beyond that temperature, stone will spall, crack, shatter, oxidize, or simply break from direct exposure to heat. Extensive heat spalling of lithic artifacts was observed in hot "spots" of the 1974 Day Burn in the Apache-Sitgreaves National Forest.

Many masonry pueblos in the Southwest are constructed with dressed sandstone blocks. Exposure to intense heat can lead to color changes through oxidation, severe cracking, spalling, and even crumbling as a result of burning away vegetation that has aided in stabilizing the remains through time (USDD-COE 1989; Traylor 1981).

Rock art sites, including those where designs are painted on stone (pictographs) and others that are pecked into stone (petroglyphs), are especially susceptible to damage by fire. When exposed to intense heat, painted designs can be soot blackened, scorched or completely burned away while petroglyphs on friable stone, such as sandstone or limestone, can exfoliate (Pilles 1982; Noxon and Marcus 1983).

b. Ceramics. Pottery is another class of artifact that may be seriously affected by fire (Burgh 1960). Pottery is made by subjecting fabricated clay vessels to intense heat where sintering and other physical and chemical changes give it clastic properties. However, fire of long duration or high intensity may refire prehistoric or historic ceramics, which can recombine the constituents, oxidize certain elements, burn out carbon paints and cause increased brittleness. Smudging or the deposition of surface carbon residue also may make it difficult to identify and date ceramics.

In the Apache-Sitgreaves fire, ceramics became highly vitrified and appeared as hard black sponges. Similarly, after the 1972 Mesa Verde National Park wildfire, spalling and discoloration of ceramics was noted. The primary impact observed at the Dutton Point Fire was smudging of shards that may disappear naturally after years of weathering. In summary, fire can burn pot shards, affect their chemical composition, change their colors, and alter their decorative paints and glazes, making identification of styles and manufacturing techniques difficult in some cases. (See also Pilles 1982, p. 6.) The more substantial changes begin to occur with temperatures of 925 F (496 C), a threshold higher than for stone.

c. Organics. Objects made or manipulated by a site's occupants are not the only materials used to reconstruct a scenario of activities that were performed at a given location. Many sites contain shell and bone, giving evidence about the nature of prehistoric diets. When exposed to a high level of heat, shell will become calcined and very friable. The effects of fire on bone have yet to be thoroughly investigated, but at Custer's Battlefield, old bovine bone fared very poorly compared to stone and metal objects (Scott 1987). Pollen grains, used for paleoenvironmental as well as dietary studies, are destroyed at temperatures above 600 F (316 C) (Traylor 1981). Artifacts made of organic materials such as woven baskets, wooden digging sticks, rawhide cordage, and fur clothing are usually very fragile in the archaeological record and are highly susceptible to charring and consumption at very low temperatures (Seabloom, Sayler, and Ahler 1991).

d. Metal and glass. Not much is known about the effect of wildland fire on inorganic materials largely associated with historic period sites in the West, such as metal implements and glass bottles or beads. A recent North Dakota prairie fire study found that small lead and glass items became fused or melted when subjected to ground surface heating (Seabloom, Sayler, and Ahler 1991).

2. Effects on Dating Techniques. The archaeologist today has several techniques available for deriving absolute dates of site occupation (Michels 1973). In addition to the impacts to artifacts and material types noted above, materials used for several dating techniques also may be affected by fires (Traylor 1981, Pilles 1982).

a. Tree rings. Tree ring records preserved in wooden beams or other construction materials used in dendrochronologic studies are highly susceptible to fire.

b. Radiocarbon. Charcoal samples used for radiocarbon dating can become contaminated from ash and charcoal produced by a fire, and could yield a date more recent than the true date of the sample.

c. Thermoluminescence. Pottery fragments, when subjected to thermoluminescent dating techniques, could provide significantly younger dates than expected after being exposed to high heating episodes.

d. Obsidian hydration. Obsidian hydration is a dating technique that measures the amount of moisture present in the external surface of an obsidian artifact. Moisture is absorbed from the atmosphere by freshly flaked obsidian surfaces at a constant rate. Heat from a fire (apparently at only high levels of heating) can alter the moisture content, thus yielding an inaccurate date or erase the record altogether.

e. Archaeo-Magnetic. Archaeo-magnetic dating measures the orientation of electrons in stones from prehistoric hearths and compares this data to changes in the earth's magnetic field over the past several thousand years. If these features are subjected to temperatures above 975 F (524 C), they can give erroneous information by releasing electrons to realign with the current magnetic fields of the earth.

f. Cation-Ratio. Cation-ratio dating is a new technique for dating rock art through chemical analysis of surface varnish. It is possible that smoke from a fire could alter the ion structure of these features, thereby preventing accurate dating.

3. Impacts of Burn Area Preparation and Mechanical Suppression. The most dramatic and predictable effects of fire activities on cultural resources result from the use of equipment in burn area preparation, fire suppression, or burn area rehabilitation work. Impacts from these activities are also the most preventable. Pilles (1982, p. 6) has noted that:

Studies of the La Mesa Fire at Bandelier National Monument and the 1977 fire on the Coconino National Forest found that heavy equipment used during suppression activities and mop-up operations had a greater effect to archaeological sites than did the actual fire itself. Artifacts are broken and displaced and small sites can be completely destroyed by one pass of a bulldozer blade. Depending on the depth of a blade cut and the proximity of site features to the surface of the ground, buried features, such as caches, burials, and firepits, can also be destroyed by bulldozer work. During both the Radio and La Mesa forest fires, about 15 percent of the archaeological sites in the area were damaged by heavy equipment. In the La Mesa Fire, however, most sites were damaged during mop-up and restoration activities after being initially avoided by fire suppression activities. Both instances point out the importance of timely planning and continued coordination with archaeologists for projects involving the use of heavy equipment.

Obviously, burn area preparation could cause damage to cultural resources if mechanical equipment is used. Additionally, construction of heliports, vehicular traffic, and hand construction of firelines can impact cultural values. Postfire erosion control measures such as mechanical seedings, contour trenching and furrowing, and construction of sediment traps are restoration activities that pose significant threats to archaeological sites.

4. Erosion and Looting. Loss of ground cover normally leads to greatly enhanced visibility. In many regions of the West, wildfires have long been noted for their propensity to expose sites previously difficult to find; consequently large numbers of people can be found cleaning the surface of diagnostic tools and excavating sites where archaeologically rich deposits are discovered following fires. Similar behavior has been noted of fire crews who had not previously been advised of the significance of such activities (Traylor 1981). This is of increasing concern, as the illegal collection and excavation of archaeological materials has escalated during the past 30 years. The water holding capabilities of litter, duff and surface soils are also reduced by fire, which sometimes generates erosion hazards.

C. Resource Management Considerations

The preceding sections briefly describe a diverse array of impacts that fire and associated fire management activities pose for cultural resource values. However, many of the heating effects only occur at significantly high temperatures and many associated on-the-ground activities can be planned ahead of time. Consequently, the fire process can be managed to minimize harmful effects and serve as a useful tool in managing cultural resources.

1. Fire Planning. The most effective means of addressing fire effects is through development of a management plan that takes the above concerns into account. (See Anderson 1985.) Various facets of the land management planning process may be used. The cultural information may be provided in a prescribed fire plan, a wilderness management plan, a general resource management plan, or, for areas that are of particularly high cultural resource values, a cultural resource management plan. All agency cultural resource management plans should include a section on the effects of fire suppression. Regardless of the type of plan employed, it should provide information about the number, type and distribution of cultural resources, known or predicted to occur, in a proposed project area (Pilles 1982, p. 8.) and how susceptible these resources are to impacts from fire. Are there abundant cultural resources in the area? Are there historic settler's cabins or sawmills present that could be destroyed by fire? Has the area ever been examined by a professionally qualified archaeologist? Are there any Native American concerns that might be affected by a prescribed burn? Are there any areas considered highly religious? Would burning at a particular time of the year disrupt traditional religious pilgrimages, plant collecting, or hunting practices in the area?

Prior to prescribed fires or the next wildfire season, baseline cultural resource information should be gained minimally through an updated synthesis of existing information, contacts with the appropriate Indian tribes, coordination with the State Historic Preservation Office, and inclusion of information from other knowledgeable sources. Further information may be gathered through field reconnaissances, sample field surveys, or detailed individual site assessments. From the information gathered, areas of unusual sensitivity or highly significant sites may be identified on maps and their vulnerability to fire effects assessed. Management direction regarding fire activities may then be established and the resulting information provided to those in charge of planning and directing field activities. The management direction for wildfire suppression and prescribed fire projects should be coordinated with the State Historic Preservation Office, again, to streamline any required Section 106 consultation needs that may arise when fire activities are imminent. The plans should include procedures for training fire crews about the illegality of artifact collecting and the associated stiff penalties, and for educating crews to identify sites so that damage to these resources can be avoided during fire suppression activity.

2. Maintaining Historic Plant Communities. Constructive use of fire also can be identified in terms of reestablishing the historic environmental context of important cultural resources and maintaining certain Native American traditional practices (Larson and Larson 1988). Examples of the former case includes restoration of grassland from recent pinyon-juniper invasion at a historic fort site and removal of brush thickets from historic trails, thus opening them to recreational use (Pilles 1982).

In the case of Native American needs, burning can be used to promote the growth of certain plants used for food, medicine, or craft manufacture. An outstanding case is presented in California where prescribed burning by the U.S. Forest Service allows growth of new plant shoots. This new growth has the proper strength and resiliency for the Yurok tribe to use in weaving baskets and hats for tribal ceremonies and as traditional apparel (Pilles 1982). Such activity is helping revitalize certain areas of traditional Indian culture.

3. Use of Prescribed Fire to Minimize Potential Damage from Wildfire. As noted above, the heating effects of low temperature prescribed fires appear to be substantially less than the effects of much hotter wildfires. Archaeologists can learn much from fire history studies and effects on soil properties. Knowledge of the frequency of prehistoric fires would likely indicate the possible cumulative effects on cultural resources, with high fire frequencies likely associated with "cool" fires and minimal impacts on resources.

a. Subsurface resources. During prescribed fires, effects of heating are usually not severe. Most artifacts are insulated from the heat of a fire by an earth cover and ideal temperatures for most prescribed fires are less than those that critically affect artifacts. If a condition is present of low fuel loads or a fire occurs with higher duff or soil moisture content, there is less potential for heating. However, if fire burns with high heat per unit area, then damage to surface artifacts is likely. Also, if a fire is of long duration, damage to buried artifacts is more likely.

In some areas fire suppression policies of the past century have led to "artificially" high fuel loads, thus increasing the potential for damaging cultural resources through severe heating when fires occur. Additionally, encroachment of shrubs and/or trees allows deep litter layers to accumulate and increases the potential for longer duration fires. In some situations, an agency may need to use prescribed burning or some other fuel load reduction strategy to attain its mandated mission for protecting cultural resources.

b. Aboveground structures. Historic period and prehistoric architectural sites pose special concerns. The historic period sites were created either during, or just before, the period of enforcing strict fire suppression policies without the augmentation of prescribed burns. Consequently, the sites have not been subjected to any form of fire, and preservation in many cases may be very good. Old buildings and ruins constructed of wood are obviously susceptible to destruction by fire. Burn prescriptions will need to be designed to avoid impacts on these types of cultural resources yet reduce the fuel load buildup around them. Possible prescriptions might be to put a fire line around sites with wooden structures so they are not burned; modify project boundaries to avoid rock outcrops where rock art is located; remove combustible materials from the surface of a site so the fire will either burn around it or burn with "cool" temperatures above it; shift the location of a control line, staging area, or utilized water source to avoid a site; change the dates of the burn so it does not impact Native American use of the area; or simply have the archaeologist monitor the burn while it is in progress.

It can be concluded that by burning under favorable conditions where burn severity may be controlled and monitored, management of vegetative communities through fire can be pursued while enhancing the agency's ability to protect cultural resources.

D. Methods to Monitor Fire Effects

The above discussion briefly addresses a number of complex issues for which few objective data are available. Most information thus far has been collected in association with major wildfires occurring in areas of heavy fuel buildup. Very little quantification of the effects of fire on archaeological sites has been documented, and little has been reported of prefire site conditions. There is a great need for experimentation, particularly utilizing prescribed fire conditions. In addition to recording artifactual and site preburn and postburn information, detailed documentation of fuel load, fuel and soil moisture, weather, fire rate of spread, temperature at and below the soil surface during combustion, duration of heating, and other fire factors needs to be accomplished. In anticipation of a fire, half of a site could be excavated prior to the fire, and the remainder excavated afterwards. By employing such procedures, the effects of the burn on various aspects of the archaeological record could be evaluated, and correlated with the behavior and severity of the fire.

More study is needed to determine the full range of effects to cultural resources by wildfire and, especially, controlled burning projects. In order to accomplish this, a variety of experiments in different environmental settings with different kinds of cultural resources needs to be done. Only by observing fire effects in a variety of conditions can we know for certain that there is no significant or irreplaceable loss to cultural resources as a result of prescribed fire programs or the degree of damage posed by wildfire. To accomplish this goal, land management plans can commit specific cultural resource sites to "management use" for fire effects experimentation.

In addition, furnace tests on archaeological materials are needed to establish controls for comparison of field test results. Furnace tests can assess the effects of different temperature levels and heating periods on specimens of ceramics, metal, glass, bone, shell, and stone. Such information can aid in documenting actual fire effects on artifact friability, weight, and visible characteristics (e.g., color, form, decorative patterns, and trademarks).

Experiments, such as those described above, would provide an ideal opportunity for interagency cooperation and could assess the relative success of the fire management procedures and philosophies of different agencies. The value of such cooperation has already been demonstrated by studies of the Radio Fire and the La Mesa Fire, conducted by the Coconino National Forest and the Southwestern Regional Office of the National Park Service. These two studies are the main data sources available for assessing the impacts of fire and fire management activities on cultural resources. An interagency clearinghouse for assembling data and reports pertaining to fire effects on cultural resources could greatly assist the otherwise disjointed approach taken by the various organizations.

As acknowledged above, visibility of cultural resources is greatly enhanced by burning away the ground cover. A 1977 tundra fire in Alaska removed shrub growth, revealing prehistoric stone-lined pits where none were previously visible (Racine and Racine 1979). Wildfires in the early 1980's in pinyon-juniper woodland areas near Las Vegas and Carson City similarly resulted in the discovery of prehistoric rock ring features where none were previously known. The Las Vegas rings were likely associated with past pinyon nut caches; the Carson City features are possibly remains of habitation structures. In Mesa Verde National Park, prehistoric farming terraces were revealed when fires burned away the dense underbrush. Before this, archaeologists were not aware of the abundant existence of such features in the Mesa Verde area. Consequently, the coordination of prescribed fires with postfire archaeological surveys would be beneficial to the agencies in achieving goals for two programs. The benefits to the cultural resources program are obvious in the greater efficiency achieved in conducting inventory efforts. Also, patterns identified in heat damage to artifacts by the above furnace and field studies have promise to aid in the interpretation of heat damage sustained by artifacts in early historic and prehistoric times (Seabloom, Sayler, and Ahler 1991).

E. Summary

Damage to cultural resources posed by wildfires and prescribed fires can be severe, ranging from chemical alteration of cultural materials to exfoliation of building materials and rock art panels. However, almost all impacts can be avoided through advanced planning. Protective measures can include removal of high fuel loads by hand or prescribed fire, careful use of fire breaks for avoiding fire effects on wooden structures and other highly susceptible resource values, and use of archaeological monitors on wildfires in sensitive areas to avoid fire suppression damage.

The experiments and observations thus far conducted indicate that cultural materials below the surface, unless directly exposed to a burning duff layer or burning underground roots, normally do not sustain significant damage, if any at all (Traylor 1981). Though the Cleveland National Forest found that many surface artifacts were damaged by a prescribed fire in chamise chaparral, no subsurface artifacts were affected (Pidanick 1982). Measurements taken at the prescribed fire documented temperatures in excess of 800 F (427 C) at the ground surface, but only 100 F (38 C) at 5 centimeters (2 inches) below the surface. Obviously, the magnitude of fire effects on the soil and its contents is proportional to heat penetration. In conifer forests, temperatures of 200 F (93 C) have been recorded one/half inch (1.3 centimeters) deep in the soil, with duff layers considerably above that figure. Obviously, such heating depends on the thickness of the duff layer, duff moisture content, amount and moisture content of large diameter dead woody fuels, and soil type and its moisture content. Given current knowledge of fire effects on cultural resources, it is apparent that fires involving larger fuel loads, longer duration burns, and large total heat release pose significantly greater hazards to cultural resources, than fires with short duration "cool" combustion temperatures.

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CHAPTER IX - PREFIRE AND POSTFIRE GRAZING MANAGEMENT

By Ken Stinson

A. Introduction

The impacts of grazing management before and after a fire have a dramatic effect on the response of vegetation to the fire, and what one can expect in the long term. The history of management on burned areas has included such things as:

1. Seeding introduced species for increased livestock or wildlife forage that resulted in additional management conflicts.

2. Short-term rest from livestock grazing to allow seedling establishment.

3. Some followup long-term grazing management but with heavy to severe utilization rates.

4. Conducting prescribed fires to temporarily increase production on an allotment to avoid adjustment in grazing use.

5. Temporarily increasing utilization by improving palatability of rank plant species such as Tobosa grass *(Hilaria mutica)*, sacaton *(Sporobolus spp.)*, and sprouting browse species.

6. Grazing use in an area during a prefire rest period required to accumulate adequate fuel to carry a fire, resulting in a subsequent prescribed fire that could not meet objectives.

The need for increased intensity of grazing management on burned areas can be understood by realizing the potential change in the plant community and associated animal response that can result from a burn. If one is not willing to commit to long-term grazing management, prescribed fire should not be considered or approved.

B. Principles of Prefire and Postfire Grazing Management

1. General Need for Improved Management. "Prescribed fire should not be a substitute for good range management. A problem rooted in inappropriate range management practices may not be corrected by vegetation treatment. In these instances management should be altered prior to application of prescribed fire. If livestock have premature access to the burn, the full benefits of the prescribed fire may not be realized and negative impacts may occur unless management of the livestock is included in the plan" (Bunting et al. 1987). "Followup management is the most important aspect of a controlled burn and must be provided for in the overall management plan" (Smith 1981). "Grazing management following burning may significantly affect the degree of change in forage species productivity and possibly the composition of the postburn vegetation" (Smith et al. 1985). The need for management of livestock use on a burned area is most critical the first growing season after fire, particularly in plant communities of arid and semiarid regions (Trlica 1977). Livestock use must be managed on the sites of both prescribed fires and wildfires.

Fire results in changes of animal behavior including grazing pattern, preferences, utilization rates, forage consumption, and frequency of grazing use. Wild and domestic animals are attracted to recently burned areas resulting in greater utilization of the burned area than surrounding vegetation (Pase and Granfelt 1971; Bunting et al. 1987). Cattle, horses, and sheep usually have the greatest impact. Grazing animals frequently concentrate on a burn because the herbage or browse is more accessible, palatable, and nutritious (Wright and Bailey 1982). Plant growing points may also be exposed, increasing the likelihood of damage from a foraging animal. Carbohydrate reserves of sprouting plants are usually depleted because of energy required to regenerate after a fire. Repeated use of these plants can cause considerably reduced vigor, and sometimes death of key forage or browse species. (See VI.B.4.b. (1) and (2), this Guide, for a more detailed discussion of carbohydrate reserves.)

Grazing in forested areas can help forest regeneration if competing plant species are grazed, or hinder regeneration if tree seedlings or sprouts are eaten or trampled. Extensive damage to young conifers from trampling has occurred in clearcut areas that were seeded to grasses (McLean and Clark in Urness 1985), and has been observed in burned clearcuts where postfire growth of grasses and sedges attracted livestock (Zimmerman 1990). The presence of larger diameter logging slash can discourage livestock and big game use.

2. Rest and Deferment.

a. Prefire. Prefire rest from grazing is required on many range sites to allow the accumulation of enough fine fuel to carry the fire. This is important in shrub/grass and pinyon-juniper types as well as in forested areas, particularly aspen ecosystems where grass and shrub litter may be the main carrier fuels (Jones and DeByle 1985). Allowing grazing, sometimes even for a short period of time during the year before the fire, can remove enough fuel to limit fire spread. A patchy fire may occur, or the fire may not be able to carry at all, and in both cases fire treatment objectives are not met. Prefire rest may also be required to restore levels of plant carbohydrate reserves on heavily grazed sites, shrub dominated sites, or where shrubs are very old and in poor condition. More than one year of prefire rest from grazing may be required to obtain adequate fuel to carry the fire, or to achieve the desired postfire response, especially in areas with severely depleted understories.

b. Postfire. The amount of nonuse necessary after a fire varies considerably with the vegetal composition, site conditions, and objectives of the burn (Bunting et al. 1987). The initial concern following burning is the restoration of plant vigor and seed production. Generally, at least two growing seasons rest are recommended (Pase et al. 1977; Wright et al. 1979; Blaisdell et al. 1982), both to allow reestablishment of preferred species and to deter reinvasion of shrubs. Bunting (1984) usually recommends rest for one year and deferment of grazing until after seeds have ripened the second year, if the range is otherwise in fairly good condition. Some species of sprouting shrubs take much longer than two years to recover, such as bitterbrush, and rest for a longer time period is necessary if reestablishment of browse species is an objective.

"Anticipated results from the best prescriptions for a burn may be seriously modified if destructive grazing practices are allowed afterwards. Only a small amount of forage is produced the first year, and grazing may cause serious damage to soil and desirable perennials. Despite the apparent abundance of green herbage, most plants are low in vigor and will be further weakened or destroyed by grazing. Furthermore, grazing will disturb the inadequately protected soil and allow increased water and wind erosion. Protection through the second growing season will allow restoration of vigor and the typical heavy seed production of perennial grasses and forbs. However, after seed dissemination, light grazing may serve a useful purpose in helping to plant the seed" (Blaisdell et al. 1982). Early establishment of a good grass cover, and subsequent conservative management, virtually assures soil stability and low sediment yields on moderate slopes (Pase et al. 1977).

Grazing in the early growing seasons immediately following burning may accelerate sagebrush reestablishment. This is particularly true when areas with dense sagebrush and low production of grasses are burned (Laycock 1979; Smith et al. 1985). This may be desirable if sagebrush is an important habitat component for wildlife species. Grazing systems that provide for periodic rest during the growing season will extend the useful lifetime of the project (Britton and Ralphs 1979; Smith et al. 1985).

Evans (1988) gives a general rule that newly seeded areas should not be grazed for at least two years following seeding. Low potential sites and those seeded with slow developing and slow growing species may require as many as four seasons of nonuse to develop into productive stands. Below average amounts of postfire rainfall also can retard the recovery of a site.

3. Proper Followup Grazing Management. "Improvement of an overgrazed range--that is, improvement in range condition--starts with a decision to stock the pasture at a rate to permit improvement" (Dyksterhuis 1958). Burning an area that is in poor condition because of overgrazing can temporarily increase production of desired species. However, the improvement will be short-lived if grazing practices remain unchanged. "Various combinations of rotation land deferment. . . have all proven to be successful where such factors as range condition, kind of livestock, stocking rate, season, and intensity were given proper consideration. Rate of stocking--balancing numbers and time of grazing animals with forage resources--is the most important part of good grazing management ... Seemingly there has been over-optimism in judging grazing capacity and allowable use, which has been an important factor in range deterioration . . . It has become increasingly apparent that former utilization standards are often several times more than can be tolerated continuously, and that reduction in livestock numbers is often necessary to correct unsatisfactory conditions" (Blaisdell et al. 1982).

Holechek (1988) researched and published utilization guides by precipitation zone for different range types in the USA. The recommended average degree of use of the key species varies from 20 to 50 percent with the upper levels only on good condition ranges or for dormant season grazing. Heavy grazing invariably leads to a gradual loss in forage productivity and vigor, high death losses, and higher costs for supplemental feed in drought years. Pechanec, Stewart, and Blaisdell (1954) found in the sagebrush-grass type that proper followup management, i.e., protection from livestock use the first year, and light grazing the second year with proper stocking thereafter, resulted in increased grazing capacity 9 years later. Capacity on this area was increased by 83 to 106 percent, but there was only a 4 percent increase without this management. The area with proper management had five sagebrush plants per 100 square feet (9.3 square meters), compared to 55 plants per 100 square feet on the area without the above management.

On desert grasslands postfire rest must occur, and careful, conservative management followed until the weakened grass cover has completely recovered (Pase et al. 1977). Postfire recovery of browse species in these arid areas may take much longer than on more mesic sites (ibid.) A common goal for all grazing systems should be reduction of damage from grazing while promoting beneficial effects, and many systems appear equally effective (Blaisdell et al. 1982). Many combinations have proven to be successful where such factors as range condition, kind of livestock, stocking rate, season, and intensity were given proper consideration.

4. Economic Considerations. One of the primary reasons for the interest in using prescribed fire and limited control of wildfire is the perception that fire is a cheap brush control treatment. Smith (1981) states that prescribed burning provides an inexpensive brush control method, but labor will greatly increase the cost of prescribed burning, so the planning process should emphasize practices (such as fuel breaks) that will reduce labor needs to a minimum. He also lists one disadvantage as the risk of fire escaping and consuming valuable forage, ensuing property damage and danger to lives, resulting in expensive suppression costs and civil suits.

Bunting et al. (1987) suggest that selection of area to be burned will dictate many of the economic variables such as fire prescription and characteristics and whether it achieves its objectives . . . the higher potential sites produce the highest benefit. He also states that burning during the spring with snow lines and increased fuel moisture on varying aspects adjacent to the proposed treatment area may aid in fire control and reduce overall cost. The limited burn size, however, may increase the amount of time and personnel required for ignition resulting in higher average costs per acre or not achieving the planned objectives. Bunting says that economics is also a factor in determining the size of fires. The costliest portion of conducting prescribed fires is establishing and burning out the fire lines. The smaller the size, the greater the perimeter per unit area. Without natural fuel breaks, an extensive system of fire lines may have to be established to restrict the fires to the desired size. This often makes the prescribed burns economically unfeasible.

From an economic standpoint, spring burning is cheaper as it can be accomplished with fewer individuals and without firebreaks in some situations (Blaisdell et al. 1982). West and Hassan (1985) state that the highest potential for prescribed burns is on sites in good condition. Haslem (1983) provides several guidelines to maximize returns from burning including realistic prescriptions, treating manageable units, using livestock use for controlling escapes, use of natural control barriers, and use of test burns. Smith (1981) states that followup management is essential in extending the fire's useful lifetime.

Young and Evans (1978) state a general rule that one must be able to step from one bunch grass plant to another to have a reasonable chance of enhancing the site by recovery of existing plants. Bunting (1984) also notes the bluebunch wheatgrass response is from existing plants for the first 3 or 4 years after a fire. Dramatic increases in numbers of plants of exotic annual species can occur after fire, particularly if the existing bunchgrass community was in poor condition or many of the plants were killed by the fire. This potential for site invasion must be considered along with the above guidelines when deciding if a site can recover without artificial seeding.

5. Examples of Different Intensities of Grazing Management. Many different grazing management strategies have been implemented after burns, however, few have been intensively monitored to determine their impacts on fire effects. Two prescribed burns in northwest Wyoming, which escaped into adjoining grazing allotments, were monitored during 1987 to provide data on effects of postfire grazing management on vegetative response.

a. Blue Creek Coordinated Resource Management Plan (CRMP). In 1984, the operators agreed to a CRMP with the Wyoming Game and Fish Department and the Bureau of Land Management. The area is located south of Meeteetse, Wyoming, in the 15 to 19 inch (38 to 48 centimeters) precipitation zone at the 7,800 foot (2,377 meters) elevation. The vegetation type is composed of limber pine *(Pinus flexilis)* and mountain big sagebrush *(Artemisia tridentata vaseyana)* on a shallow loamy range site. Key graminoid species include Idaho fescue *(Festuca* *idahoensis)*, green needlegrass *(Stipa viridula)*, and rhizomatous wheatgrasses. The growing season in this area is from mid-May until about September 1. The adjoining allotment received heavy livestock use from the first of July until snowfall each year. The unburned site had a mountain big sagebrush canopy cover of 60 percent and produced an estimated 700 pounds per acre (785 kilograms per hectare) of annual sagebrush growth.

The planned actions started in 1981, including nonuse of livestock grazing, conducting prescribed burns, and fencing for grazing strategy implementation. In the year of the prescribed fire, snow left the area earlier than normal. The prescribed burn was conducted on April 3, 1985. The fire escaped across the allotment boundary fence into the adjoining allotment, burning about 20 acres (9 hectares) of mountain sage type. About 15 acres (7 hectares) of the escaped fire area were deferred from grazing for two growing seasons by a temporary electric fence, that is, no grazing occurred during the growing season. The original intent was to exclude animal use entirely but elk tore down the electric fence in September both years.

Herbaceous production data was collected during July 1987 in four adjoining sites. Weight estimates were made on ten plots, and two plots were clipped to obtain a correction factor. The study included one transect in the Blue Creek allotment that received no grazing for 4 years before the prescribed fire, and no use in the 27 months after the prescribed fire when the production data was collected. There were three areas studied in the adjoining allotment, all of which were grazed in the years before the prescribed fire. One area was unburned; the second received season long grazing in 1985 and 1986; and the third site was the fenced area where grazing was deferred throughout two growing seasons. Table IX-1 details the site and species production data that was collected.

Table IX-1: Herbaceous Production (pounds/acre dry weight) - Blue Creek.

Species	Unburned	No Rest	Deferred	Nonuse
Idaho Fescue	280	74	568	1,056
Rhizomatous wheatgrasses	19	447	210	762
Green needlegrass	24	51	182	103
Other Grasses and Forbs	590	920	828	713
Forbs	NA	579	1140	859
TOTAL	913	2071	2828	3493

All burned areas had higher grass production than the unburned area. However, the areas with deferred grazing and nonuse had much higher production of Idaho fescue, the preferred species. The nonuse area had twice the fescue production of the area that was grazed the year before the fire and deferred for two seasons afterwards. The unrested burned area had one-quarter of the Idaho fescue as the unburned area. Higher postfire palatability of this preferred species likely resulted in higher utilization rates by livestock, causing it to all but disappear from the site.

b. Orchard-Woods allotments. The second study area is located south of Ten Sleep, Wyoming, in the upper 10 to 14 inch (25 to 36 centimeters) precipitation zone at the 6,800 foot (2,073 meters) elevation. Vegetation was dominated by mountain big sagebrush, bluebunch wheatgrass *(Pseudoroegneria spicata)*, green needlegrass, and Idaho fescue. The permittee for the Orchard Ranch was conducting a prescribed burn on September 14, 1983, and the fire escaped into the adjoining Woods Allotment. The Orchard pasture had received light to moderate grazing with a deferred rotation strategy for two years prior to the fire and was rested for two growing seasons afterwards. The adjacent Woods pasture received heavy season-long grazing prior to the fire, and was deferred until seed ripe the first two seasons after the burn. Herbaceous production data was collected on July 30, 1987, from both sides of the division fence by clipping ten plots (Table IX-2).

Table IX-2: Herbaceous Production (pounds/acre dry weight) - Orchard-Woods.

Key Species	Woods (Deferred)	Orchard (Rested)
Green needlegrass, Bluebunch wheatgrass, Idaho fescue	131	447
Other Grasses and Forbs	1,648	864
TOTAL	1,779	1,311

Total production on the deferred Woods allotment was higher than on the Orchard area. However, much of this production was weedy grasses and forbs. Production of the three preferred grass species on the Woods allotment was only one third of that on the Orchard allotment which had been rested after the prescribed fire.

c. Management implications. The primary implication of the preceding examples is that in order to increase production of late successional species, there must be a commitment to rest after the burn and proper grazing management in the long-term. The two growing-season rest after a burn greatly speeds the recovery and improvement of the key species. If a burned area is not rested, the extra moisture available after the sagebrush or other shrub and tree species are eliminated is used by rhizomatous grasses, or early successional grasses and forbs, depending upon what species are present before burning. If there are sprouting shrubs present that are not used by livestock such as rabbitbrush, they can become dominant in the community. Lupine, which is a legume, will take up the extra moisture in the mountain sage type for the first two seasons if the late successional grass species are damaged or lacking in the understory. The lack of sagebrush invasion into the burn sites on Blue Creek shows the importance of maintaining the maximum vegetative ground cover and vigor to help slow the recovery of sagebrush seedlings on the site. It can be expected that forage would continue to increase on these sites because peak production on a sage site generally does not occur until the third to fifth year after a burn. An additional implication for prescribed fire is that increased production of herbaceous species does not necessarily mean that the site is enhanced, if, as on the Woods pasture, much of the production is composed of annual weeds and rhizomatous grasses.

C. Resource Management Considerations

1. Fire Effects. The effects of fire on plants and their response characteristics are described in detail in Chapter VI, this Guide. The following impacts and changes must be considered in planning proper site management to obtain the desired fire effects.

a. Damage to key forage and browse species by repeated heavy utilization by animals or burning is very similar; therefore, many areas are impacted by fire and then again by grazing and/or browsing animals.

b. Increased palatability and accessibility of grasses, forbs, and shrubs, influenced by the green period, nutrient content, growth form, and removal of dead material, occurs during the first few growing seasons after a burn.

c. Carbohydrate reserves of burned plants are lowered the first few seasons after a burn.

d. Fire effects result in changes of animal behavior including distribution, utilization rates, forage consumption, and frequency of use.

e. Prefire and postfire grazing management largely determines the benefit/cost ratio because of its considerable influence on the life of the beneficial aspects of the burn.

2. Plant Maintenance Factors. Little research has been conducted to determine the best long-term management practices for burned areas. However, the findings from plant ecology studies conducted over the past 60 years can be applied. The following factors should be considered to determine grazing management requirements:

a. Plant community health (carbohydrate reserves for plant vigor and recovery potential).

b. Composition of plant species that occupy the site and their successional position.

c. Recovery period required by species after burning for vegetative, root, and reproductive growth.

d. Utilization limits for species and season of grazing. A key factor in range deterioration has been overly optimistic estimates about the amount of allowable use that an area can sustain. The research literature gives a range of 20 to 40 percent utilization of annual growth during the critical growth period and 40 to 60 percent during the plant dormant period depending upon species and management objective. Use at higher levels can cause deterioration of the plant community. **e.** Site potential, which considers the range site or habitat type description.

f. Plant species morphology and reproductive mechanism.

g. The type of grazing and browsing animals that use the site, both wild and domestic, and rodent or insect use.

3. Length of Postfire Rest Period. The length of the period of postfire rest from livestock use depends on these factors:

a. Ecosystem type and ecological condition before burning.

b. Vigor of vegetation prior to fire.

c. Season of fire.

d. Growing season conditions, including temperature and precipitation, prior to and following burning.

e. Whether establishment of new plants is by seed or sprouting. Seedlings require a longer rest period to become resistent to grazing damage. Browse species often require second year growth before producing seeds.

f. The management objectives for the area. For example, the length of postfire rest may vary significantly for the same area depending upon whether the area is being managed primarily for range, wildlife, forestry, or recreation, or a mix of these activities.

4. Summary Recommendation for Rest. Postfire rest from grazing is required on both prescribed fires and wildfires, on seeded areas and unseeded areas. The length of the period of rest from grazing is dependent upon accomplishment of measured objectives for the area. General recommendations are:

a. Prefire.

(1) At least one growing season of rest before a prescribed burn is needed on many sites to increase both root reserves and fine fuels needed to carry fire.

(2) Severely depleted sites may require several years of rest before burning in order for key plants to regain vigor and reproductive capabilities.

b. Postfire.

(1) Burned areas should be rested until a good ground cover and a litter layer are present to provide soil and watershed protection.

(2) One growing season of rest may be adequate on some highly productive or high condition sites after a low severity fire. A minimum of two growing seasons rest is recommended for most burned areas.

(3) Areas under intensive grazing management can be grazed immediately after a fall-winter burn to harvest some unburned vegetation within the unit. This short use period prior to regrowth could assist in nutrient cycling, roughing of the soil surface, and breaking of any sealed ash layers. Because this use can also result in physical damage to the plant roots and crowns, it should be closely monitored and occur during a short time period. Time control grazing may be used to help accomplish postfire objectives if proper consideration is given to animal impacts and plant community recovery periods.

(4) Sites that are burned to increase utilization of unpalatable species should be grazed immediately after the fire while the undesirable plants are young and actively growing.

(5) Heavy utilization grazing of an area by wildlife or wild horses will require further rest from permitted livestock if their numbers cannot be controlled. Wild horses are difficult to move from their normal territories but any water close to the burn area can be regulated, especially during the first few growing seasons. Wildlife numbers can be controlled by harvesting excess animals, although this is only a feasible management strategy in the event of an extremely large wildfire.

(6) Below normal precipitation may delay vegetative recovery. A burned area should be inspected to determine if it is ready for grazing. It cannot be assumed that vegetation is ready for grazing just because the prescribed period of rest has occurred. The degree of accomplishment of measured objectives should be the most important criteria in determining the length of the postfire rest period.

(7) If recovery of wildlife browse is a key objective, the length of the postfire rest period may need to be much longer, and permitted levels of utilization by livestock may be much lower, than if target species for an area are grasses and forbs to be used by livestock.

c. After postfire rehabilitation. If postfire rehabilitation has been conducted, livestock should not be allowed back onto the burned area without evaluating whether rehabilitation objectives have been met. Postfire rehabilitation considerations are discussed in Chapter VI.C.6.a., this Guide.

5. Effects of Management Strategy. The commitment to long-term grazing management on burned areas is vital for real long-term improvement in plant productivity and composition. Determination of desired plant community objectives is necessary before the grazing strategy can be decided. Many sites proposed for burning are in poor condition or have mature stands of trees with little understory vegetation. These sites require long periods of time to progress through successional changes to meet objectives. If introduced plant species are seeded on the burned area, a grazing strategy must be designed to meet their survival needs. Seeded sites may have to be fenced separately from native rangeland due to phenology and palatability differences.

Effects of fire and grazing management cannot be easily separated; therefore, fire effects on vegetation should be monitored and evaluated both in the short term (prior to start of grazing or at the end of second growing season) and the long term (after two cycles of the grazing strategy or 8 to 10 years). The short-term evaluation assesses achievement of fire treatment objectives, while the long-term evaluation considers the attainment of desired plant community objectives.

Specific grazing strategies must be designed to meet key species requirements and land use planning objectives. These general principles should be kept in mind in post fire management of burned areas.

a. Type of grass. Bunchgrasses require lighter utilization rates and longer rest periods than do rhizomatous or annual plant species.

b. Palatable shrubs.

(1) Highly palatable sprouting shrubs often require long rest periods after fire to allow restoration of carbohydrate reserves, to produce seeds, and permit seedling establishment.

(2) Upland shrubs such as sagebrush require bare soil surface and minimal herbaceous competition to enhance reestablishment. Therefore on many of the drier and lower snowfall sites heavy spring grazing could promote sagebrush establishment, if that is the management objective.

c. Riparian communities. Riparian communities with shrubs and trees require long-term rest to recover and light utilization of key shrubs to maintain a healthy community. It may be necessary to fence riparian areas for several years to allow initial recovery.

d. Forb-dominated communities. In order to maintain a forb-dominated plant community, the grasses must receive heavy grazing pressure or be burned more frequently.

6. Economic Factors. The following general guidelines can be used when analyzing fire effects that have an influence on economics.

a. Site selection criteria. The first step is to consider the site potential and set objectives accordingly. These selection criteria can be used for analysis of fire effects on either prescribed burns or wildfire rehabilitation.

(1) Ecological condition. Sites in high-fair or better condition should be selected for prescribed burning if increased herbaceous production of desirable species is a short-term objective and reseeding is not planned.

(2) Presence of desirable species. If improvement of native species is the objective, it must be determined if there are enough desirable remnant plants to make the burn worth conducting. Seeding is very costly and the native species are more adapted to the sites.

(3) Invader species. Are undesirable species or noxious weeds present on the site that might be favored by fire? Species such as cheatgrass, rabbitbrush, and horsebrush are common problems on arid and semiarid rangelands. Any reseeding must be completed before the first growing season to avoid dominance by introduced annual species such as cheatgrass. Noxious weeds may require treatment if there is a significant number present.

(4) Burn size. Is the burn acreage within the management unit large enough to avoid livestock and wildlife concentrations, thus negating the positive fire effects? Refer to Chapter VII.B.3.a. and b., this Guide, for a discussion of the impacts of burn size and configuration on wildlife habitat.

b. Factors to consider in analyzing benefit/cost. The following points are given to consider in planning prescribed burns to improve native herbaceous vegetation. If the objective is to prepare a site for seeding or removing undesirable species, then other principles could be involved.

(1) Conduct burns during the spring period or other seasons that require minimum fire control efforts. The three major costs that can be reduced are equipment, personnel, and fuel break construction. Offsetting factors include ignition problems and objective accomplishment because fuel or soil moisture or weather limit the likelihood of a successful fire.

(2) Burn the largest acreage possible within the constraints of the objectives for the burn. The impact on animal species and other resources will determine the maximum size.

(3) Sites with higher ecological condition and plant vigor respond more quickly and favorably to burning. Greater production increases, a quicker recovery, and better chance for seeding establishment occur when a given ecological site in the mid to upper precipitation zone is burned and less damage to desirable species can be expected.

(4) Extending the life of the fire effects through long-term grazing management can improve the long-term cost effectiveness of a burn project.

E. Summary

Proper site management based on specific objectives and plant species is essential in the management of fire effects. Improper grazing management can easily nullify efforts put into prescription burning or wildfire rehabilitation, as well as impede natural vegetative recovery after wildfire. Impacts of long-term grazing management before and after a fire can be easily overlooked; therefore, proper grazing management including the appropriate kind of livestock, the stocking rate, the season and the intensity of utilization, and the length and frequency of use are most important.

The period of nonuse by livestock necessary after a fire varies considerably with the vegetative composition, site conditions, resource conflicts, and objectives of the burn. Grazing closures apply to prescribed fires and wildfires, whether they are artificially reseeded or recovery is by natural means. In some situations, the only way to ensure nonuse of critical areas after a fire is to construct fences.

Proper grazing management before and after a fire has a major impact on fire effects, vegetation changes, economics, and rehabilitation success. In analyzing fire effects, several site selection criteria should be considered including the site potential, the ecological condition, the presence of desirable and invader plant species, the acreage of burn within the management unit, and the livestock management. The consideration and implementation of these factors determines the benefit/cost ratio and the success of a burn project or postfire rehabilitation effort.

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CHAPTER X - EVALUATION

By Ken Stinson and Melanie Miller

A. Introduction

Documentation is the collection, organization, and storage of all information pertinent to a specific project, in order to maintain a long-term record, and to facilitate project evaluation. Evaluation is both an objective and intuitive process of examining and assessing data and observations to determine if planned objectives were met. Fire effects documentation and evaluation are important in order to replicate positive fire effects and avoid duplication of negative fire effects in future prescribed fires; to assess postfire management of land uses on both wild and prescribed fires; to assess the effectiveness of postfire rehabilitation in preserving site quality; to provide rationale for fire use to the public; to meet legal requirements and liability; to permit evaluation of objectives; and to provide a basis for improvements in project planning, implementation, and management. Fire effects monitoring and evaluation should be included in area monitoring and evaluation plans.

1. Documentation. All documents shall be labeled by project name and number, date, and legal description. A detailed list of types of data that can document a wildland fire are listed in C.3.a., this chapter, pages 3 to 5. The following are general classes of information used to evaluate fire effects, and should be included in project records.

a. A complete description or reference shall be documented for the method and technique used to collect the monitoring data. Monitoring studies appropriate to evaluate land-use objectives should be used.

b. Photographs properly labeled, dated, and filed, including compass direction of photo.

c. Observations before, during, and after a prescribed fire or wildfire.

d. Schedules and responsibility of evaluation and documentation.

2. Organization and Maintenance of Documents. Because fire effects information requires long evaluation periods and the data accumulates over time, it is important that fire monitoring and evaluation records are placed in a permanent file. Records may be needed many years after the original incident to study the site, or for comparison with more recent events on similar sites. The data should be stored in the appropriate permanent project file, such as allotment/ management unit files, or timber sale file. Records should be kept indefinitely or archived and labelled as "Permanent File - Do Not Destroy."

B. Considerations for Fire Effects Evaluation

Evaluation is both an objective and intuitive process of examining and assessing data and observations to determine whether planned objectives were met. There are two aspects of postfire evaluation, evaluation of fire effects, and an evaluation of the effectiveness of postfire management actions. A "cookbook" approach to evaluating fire effects should not and can not be applied in all situations (Pellant 1989).

1. Prescribed Fire versus Wildfire Evaluations.

a. Fire effects. This evaluation provides the decision maker with the necessary information to make sound decisions regarding postfire rehabilitation and management actions.

(1) Prescribed fire. For a prescribed fire, fire effects are evaluated and compared to the fire treatment objectives. The fire treatment is assessed to determine whether it indeed enhanced resource management objectives for the site. Because the event occurs under pre-planned conditions, site-specific prefire and postfire monitoring can occur. If desired effects are not attained, then an evaluation must determine why they were not. If postfire rehabilitation is planned, an assessment must be made whether the fire treatment suitably prepared the site for seeding or planting. A decision is made whether postfire site management objectives that were developed before the prescribed fire are still suitable.

(2) Wildfire. The area in which a wildfire occurs may have established resource objectives, and fire treatment objectives may have been prepared in the Escaped Fire Analysis. However, because wildfires are random events, any prefire monitoring with the specific intent of documenting the effects of that fire can rarely be done. Some monitoring data may be available that was performed for other resource management programs or projects, and some of these data may be suitable for postfire comparisons.

After a wildfire, an assessment is made of the degree to which fire may have affected the ability of the land to meet any resource management objectives that may have been established for the area. In particular, the expected response of the vegetation on the site, and the potential for site erosion, must be evaluated. A plan to take actions paid by Emergency Fire Rehabilitation funds may be written if action is required to mitigate the effects of the wildfire.[(1)] The decision to use EFR funds is very important; therefore, appropriate personnel should be utilized and the evaluation process should be initiated as soon as possible. Fire effects evaluation can begin long before the fire is declared out. Recommendations are made whether a continuation of present site management practices is acceptable, or if changes are needed, such as in grazing management or use by all terrain vehicles.

b. Effectiveness of postfire actions. An evaluation should be conducted to determine whether postfire site rehabilitation treatments limited negative effects of fire, to assess the impacts of postfire activities such as salvage logging, and to establish the effectiveness of postfire site management actions, such as grazing restrictions, in preserving or improving site quality.

c. Effect of fire and postfire actions on attainment of site objectives. Depending on whether a wildfire or prescribed fire is being evaluated, and the level of land use goals and objectives that existed for the site before the fire occurred, an assessment must be made of whether resource management, fire treatment, and/or rehabilitation objectives were met.

2. Degree of Evaluation. The degree of evaluation that is required depends on:

a. The complexity of the project; several resources may have been monitored and documented.

b. Whether the resource and/or treatment objectives were tied to ecosystem effects or individual species responses.

c. The potential for controversy, involving such factors as designated Wilderness, critical wildlife or watershed values, Areas of Critical Environmental Concern, or political considerations.

d. Experience with or understanding of site specific fire effects, or rehabilitating or managing similar areas.

e. Time and funding availabilities.

f. Availability of existing data bases. Other disciplines may have extensive records, such

as streamflow, weather data, or soils inventories, that can be incorporated into the evaluation.

3. Steps in the Evaluation Process. Existing agency policies and procedures may require certain steps to occur in specific order. The following steps are suggested for postfire evaluation.

a. Identify parties responsible for conducting the evaluation process.

b. Review objectives.

c. Assemble data.

d. Interpret data/observations.

e. Determine if and to what degree objectives were met.

f. Prepare evaluation report, which includes recommendations, for decision maker.

g. Disseminate new findings to colleagues.

h. Observe long-term changes.

C. Evaluation Procedure

1. Identify Parties Responsible for Conducting the Evaluation. An interdisciplinary team approach is used to evaluate both prescribed fires and wildfires. Managers designate both a team and a team leader. All resource specialists involved in planning a prescribed fire or who had the lead in conducting it (Burn Boss or Project Coordinator), should be involved with a prescribed fire evaluation. Any resource specialist involved in the suppression of a wildfire, particularly the Resource Advisor or Environmental Specialist, should be included on the team that assesses the effects of a wildfire. It is very important that individuals with fire effects experience, particularly in similar vegetation types, be involved in the evaluation.

2. Review Objectives. Review any general or specific objectives developed for the site being evaluated. These can include land use decisions, fire planning objectives, resource management objectives, prescribed fire objectives, fire treatment objectives, and rehabilitation objectives. Rehabilitation objectives referred to here are those objectives established as goals for the rehabilitation treatment, such as reducing soil loss, or preventing invasion by exotic plant species. Other objectives that are standard operating procedures are also evaluated, such as keeping a prescribed fire within the prescribed fire target area, meeting safety concerns, protecting cultural resources, and staying within the cost target for the project.

3. Assemble Data.

a. Data sources.

(1) Project file.

(2) Project plan such as Allotment Management Plan, or Timber Sale Plan.

(3) Site specific plan such as Prescribed Burn Plan or Rehabilitation Plan.

(4) Records of any onsite evaluations conducted by Interdisciplinary teams, such as preparation for a prescribed fire, or observations made during or shortly after a wildfire.

(5) Any reports associated with the occurrence of a wildfire being evaluated, such as the Escaped Fire Situation Analysis, Burned Area Report, or reports by the Fire Behavior Analyst, Incident Commander, or Resource Advisor.

(6) Fire effects data or evaluations from similar projects, such as other prescribed fire or wildfire evaluations.

(7) Climatological data.

(a) National Oceanographic and Atmospheric Administration (NOAA), e.g., monthly summaries.

(b) Remote Automated Weather Stations (RAWS), including archived

data.

(c) Local manual weather stations.

(d) State climatologist.

(e) Soil Conservation Service (SCS), e.g., snow surveys and soil moisture indices.

(8) Weather, fuel, or soil moisture data collected at the time of the fire.

(9) Air quality permits, smoke observations, or data collected at the time of the fire.

(10) Any site specific monitoring information, such as preburn and postburn fuel, soil, vegetation, or fuels data. For a wildfire, some information on prefire vegetation composition may be obtained from any resource management or activity plan for the area, or from rangeland inventory, trend, and utilization studies.

(11) Aerial photographs.

(12) Resource maps, such as vegetation, soil, timber, or cultural site locations.

(13) Records that document uses of the area.

(a) Timber sales, regeneration surveys, and records of post-logging treatment, such as slashburning, pile burning, scarification, or no treatment.

(b) Actual use and utilization, and season of use by licensed livestock or wild horses; utilization levels on key species; unauthorized use records (trespass file).

(c) "On-the-ground" observations, such as extensive amounts of wildlife use during a particular period of time.

(d) Use by other public land users, such as sportsmen or conservation groups, fish and game agencies.

(e) Observations on extreme weather events, anomalous climatic trends, grasshopper or rodent infestations, wildlife use, and ORV impacts.

b. Data adequacy. Compare the amount of monitoring data collected with the amount planned for collection. If data collection was inadequate, determine the reasons why planned monitoring was not carried out.

c. Statistical analysis. Determine if the appropriate statistical analyses have been completed, or if inadequate data were collected to conduct the analysis. (See Data Analysis.) If data were collected in such a fashion that statistical analysis is not possible, or is inadequate, note should be taken so future data are properly gathered.

4. Interpretation of Data/Observations.

a. Uncertainty and reliability. Fiscal and time constraints do not allow for collecting enough data for managers to make risk-free decisions, and some uncertainty will remain, even if a statistical analysis is conducted. Decide what risk level to accept, and whether any data inadequacy is the result of uncontrollable factors such as acts of nature (floods, hailstorms), vandalism, or equipment failures. If poor study design prevents proper data analysis, document this fact. Be sure the decision maker is informed of the reliability of data; i.e., distinguish between use of complete and properly collected data (hard data) and assumptions made where data gaps exist (soft data).

b. Results of data analysis. Explain data analysis results in terms that non-statisticians can understand. Integrate any quantitative or qualitative, onsite data with observations and results from other studies. Be wary of interpreting data mechanically without considering the possibilities of undocumented causes for change. If necessary, use expertise from other offices to decipher complex interrelationships. Prepare a summary report of the findings. **5. Determine If and To What Degree Objectives Were Accomplished.**

a. Indicate whether each planned objective was met and to what degree it was accomplished.

b. If objectives were not met, identify reasons why. Some general areas to consider are:

(1) Objectives were unattainable or mutually exclusive.

(2) Timeframes for objective accomplishment were unrealistic.

(3) Operational procedures used to implement the project were not appropriate, such as an ignition method or ignition pattern that caused a more severe treatment than needed to produce the desired fire effects.

(4) The operational plan and/or procedures outlined in the project plan were not followed.

(5) Followup site management was inadequate; e.g., postfire grazing occurred too quickly or at too high an intensity, or off road vehicle use occurred on a burned area before vegetative recovery could sustain ORV use.

(6) Monitoring techniques or sampling intensity were not adequate to detect change.

6. Prepare Evaluation Report with Summary and

Recommendations.

a. Prescribed fire.

(1) Brief summary of actions taken and burning conditions.

(a) Date, time of day fire occurred, acreage burned, and ignition pattern and method.

(b) Weather, fuel, and soil moisture conditions during the fire.

(c) Observed fire behavior and characteristics.

(d) Observed smoke production and characteristics.

(e) Any differences between fire prescription and actual burning conditions.

(f) Map of the proposed fire area and the area that actually burned.

(2) List of objectives of prescribed fire.

(3) Summary of monitoring results and observations.

(4) Describe which and to what degree objectives were met (expected results versus actual results).

(5) If objectives were not met, briefly describe those factors that were responsible for the lack of achievement.

(6) Make recommendations to management.

(a) Immediate changes needed in management strategies.

(b) Changes in how future prescribed fires could be planned or implemented, including changes in prescription or ignition that would lead to better results.

(c) Changes in postfire site management.

(d) Changes in monitoring procedures.

(7) After management review and decision(s), evaluation report should be filed in the appropriate file(s) and required followup actions assigned and initiated. This report should be signed and dated by the preparer and the reviewing official.

(8) Followup on management decision(s) - repeat evaluation process.

b. Wildfire - fire effects evaluation.

(1) Brief summary of wildfire extent and effects.

(a) Date and acreage of public lands burned.

(b) Prefire vegetation.

(c) Map of the burned area including soil mapping units.

(d) Multiple-use objectives identified in land use plan, e.g., watershed value, wildlife habitat, livestock forage.

(e) Interdisciplinary team findings.

i. Estimated percent survival of key plant species.

ii. Potential for postfire erosion and sedimentation.

iii. Potential for invasion of site by exotic species.

(2) Recommendation for postfire actions.

(a) Postfire rehabilitation actions, e.g.,

i. Seeding, including recommended species and seeding rates.

ii. Contour falling of trees.

iii. Construction of instream structures to control channel erosion.

(b) Feasibility of conducting salvage logging.

i. Locations within burned area.

ii. Methods that will not cause negative impacts on soil.

(c) Changes in postfire land uses, e.g.,

i. Temporary restriction or exclusion of grazing to allow recovery of perennial plants or establishment of seeded or planted vegetation.

ii. Road or all terrain vehicle closures.

(d) Need for fence construction to exclude livestock and/or wildlife.

(3) Specific objectives for postfire management: specify desired condition, e.g.,

(a) Limit erosion to a specified amount.

(b) Restore site to a productive state by salvaging merchantable timber and replanting trees, and leaving a specified number of wildlife trees that do not pose a safety hazard.

(c) Obtain a specific level of plant cover or productivity.

c. Wildfire - rehabilitation and/or management evaluation.

(1) Prepare a brief summary of actions taken.

(a) List management objectives and planned actions.

(b) Describe actions taken and date.

(c) Include a map of treated or rested areas.

(2) Summary of monitoring results and observations.

(3) Describe which and to what degree objectives were met.

(4) If objectives were not met, briefly summarize what factors were the cause for lack of achievement.

(5) Make recommendations to management.

(a) Immediate changes required in management strategies.

(b) Recommendations for improvement in rehabilitation and postfire site management practices.

d. Post evaluation actions.

(1) After management review and decision(s), the evaluation report should be filed in the appropriate file(s) and the required followup actions assigned and initiated. This report should be signed and dated by the preparer and the reviewing official.

(2) Followup on management decision(s) - repeat evaluation process if appropriate.

(3) If no management action was taken (e.g., the burned area was grazed, or no erosion control efforts were made), document any adverse impacts that occurred and what future management actions need to be taken under similar conditions.

7. Disseminate New Findings To Other Interested Parties.

a. Fire Effects Information System.

b. Internal or interagency newsletters.

c. Professional meetings.

d. Local, state or regional workshops.

e. Agency technical publications.

f. "Expert Systems" or relational databases.

g. Office-to-office memos.

h. Professional journals.

i. Videos and other visual media.

j. Electronic bulletin boards.

8. Observe Long-Term Changes.

a. Within what timeframe did the original target species reestablish?

b. Are noxious weeds or unwanted species increasing in the postfire environment?

c. Have cyclic climatic events, such as drought or a series of wet years, affected postfire vegetative recovery? What effect have they had on plant community composition and characteristics?

d. Are postfire management actions still having the desired effects?

e. Is the level or type of postfire management adversely affecting desirable native vegetation or seeded species?

f. How have the different species that were originally seeded persisted?

g. Is cumulative acreage burned by wildfires large enough to affect whether future prescribed burns should be carried out?

D. Summary

Evaluation of both monitoring data and the impacts of postfire activities must be conducted in order to ensure that lands receive the best possible fire treatment, rehabilitation, and postfire management. Once we have monitored and evaluated enough projects and management actions on similar sites, and adjusted our actions based on these results, we can become more confident that the proper treatment is being implemented. The same degree of monitoring and evaluation need not be carried out on all subsequently treated areas if vegetation type, soil type, and treatment prescription are similar to that of other successful treatments. However, it is professionally unacceptable to conduct no prefire or postfire monitoring or site observation, to assume that an area is ready for grazing because the designated length of time has passed since the occurrence of a prescribed or wildfire, or to conduct no evaluation of the implementation or effectiveness of postfire site management in preserving or enhancing site quality. Without some check on the results of our activities, accumulated assumptions can lead to land treatments that do not meet resource management goals and objectives, and lead to deterioration, instead of enhancement, of site quality.

1. See BLM Manual Handbook H-1742-1 (USDI-BLM 1985b) for a discussion of EFR planning procedures on BLM land.

CHAPTER XI - DATA MANAGEMENT

By Dr. Robert Clark

A. Introduction

Managers can gain a number of benefits from the analysis of their data. The results of the analysis helps managers make informed decisions leading to favorable outcomes. Analyzing and manipulating the data gives the manager a special familiarity with the data and a feel for how precise and repeatable the sampling is likely to be. Describing and defending the credibility of the data is easier when it has been appropriately analyzed and described in standard statistical terms. Practice and familiarity with the most common statistical analyses provides a basis for better understanding, interpretation, and proper application of results presented in professional fire literature.

This chapter will acquaint fire managers with some elementary statistical methods that can be used to help make decisions. The methods are demonstrated by example. Managers interested in more detailed discussion or more sophisticated methods are referred to Freese (1962, 1967), Little and Hills (1978), and Eshelman et al. (1986). Managers are also encouraged to contact a statistician ***before*** data are collected to ensure the data will be usable and appropriate, and that assumptions of analysis are not violated.

Statistical terms used in this chapter are included in the Glossary of this Guide. Additional discussion is available in statistics texts.

B. Principles and Procedures of Data Analysis

1. Determination of Sample Size (Number of Observations).

Samples (observations) cost money, but so do decisions based on inadequate data. Therefore, it is helpful to collect an appropriate number of observations. One useful method to determine the appropriate sample size for continuous data is (Freese 1967, p. 12):

$$n = (t^2 s^2) / E^2$$

where:

n = the required number of observations.
t = tabulated Student's t available in most statistics texts.
s = sample standard deviation (and s^2 = sample variance). Note the unfortunate situation here, that some estimate of variance is required to calculate sample size, i.e., there is one equation with several unknown values. This is usually handled by collecting a preliminary sample from which to estimate a variance and a mean, which in turn are used to calculate the sample size.
E = desired precision, expressed as a proportion of the sample mean.

Example: Suppose the manager needed to know the foliar moisture content of basin big sagebrush on a proposed prescribed fire area. Because fire behavior in sagebrush is closely related to foliage moisture content, the manager decided to sample such that the moisture content was estimated within 10 points of the true moisture content level at the 90 percent confidence level. This indicated that the manager wanted to know the moisture content on the site within plus or minus 10 points of the true average value (precision level), and wanted the estimate to be correct 9 out of 10 times (confidence level). 10 points of error was selected because fire behavior in sagebrush can begin to change quite significantly over a range of 20 percent foliar moisture content (plus or minus 10 percent).

The manager collected foliage from 12 different plants, dried and weighed the samples, calculated the moisture content, and recorded the following results. What sample size is required to meet the manager's needs? Was the preliminary sample adequate?

plant	x = moisture content (%)	x^2

1	70	4900
2	90	8100
3	80	6400
4	70	4900
5	60	3600
6	80	6400
7	90	8100
8	100	10000
9	90	8100
10	80	6400
11	70	4900
12	60	3600
n = 12	Êx = 940	Ê x^2 = 75,400

Note that the Ê symbol means summation.

a. Step 1. Look up t value from t table in most statistical texts (e.g., Freese 1967, p. 77). Degrees of freedom (df) are n - 1 (12 - 1 = 11). The appropriate t value at the 90 percent confidence level (or 10 percent error rate) is 1.796.

b. Step 2. Calculate the variance (s^2) of the preliminary, 12-observation sample according to [note that this calculation results in the sample variance (s^2) rather than the sample standard deviation (s)]:

s^2 = Êx^2 - ((Êx)^2/n) / n-1 = 75,400 - ((940)^2 / n) / 11 = 160.6

c. Step 3. Decide how much error can be tolerated. In this example, the manager wanted the estimate to be within 10 percentage points of the true mean moisture content. Therefore, E = 10.

d. Step 4. Calculate the appropriate sample size according to the formula described above:

n = (t^2 s^2) / E^2 = $(1.796)^2$ (160.6) / $(10)^2$ = 5.18 = 6 observations

e. Conclusion. The preliminary sample of 12 observations was adequate. If the preliminary sample of 12 observations had been inadequate, the manager had several options. First, more observations could have been taken. Second, the acceptable precision level could have been lowered. Third, a lower confidence level could have been accepted and some added risk incurred. Or fourth, the fire prescription might have been altered (e.g., faster windspeed or lower fuel moisture content) to compensate for the added uncertainty. The important point is that the manager had some quantitative information on which to base a decision.

f. Note. The size of the preliminary sample depends on the variable being measured, on the objective for measuring the variable, and on funds, time, and work force constraints. Inherent variability associated with most natural resource sampling indicates that 10 or more observations may usually be required to obtain a reasonable estimate of variance. One observation is **never** adequate because degrees of freedom would be zero and no analysis is possible.

2. t-Test for Paired Observations (Plots). This test is especially useful for comparing effects between two fire treatments, such as burned vs. unburned, or backing fires vs. heading fires. An assumption of the test is that pairs of plots are established prior to the treatment, each group has the same population variance, the population of observations follows the normal distribution, and that treatments are randomly assigned to each individual plot. It is possible, however, to establish plot pairs adjacent to firelines (one plot on either side) such that these assumptions are not violated. A similar test for unpaired plots (Freese 1967, p. 24) should be used where plots are not paired; that test, however, is slightly more time consuming and a slightly greater loss of sensitivity may occur. Only the t-test for paired plots is illustrated here.

Example: A fire manager suspected that fire residence time might be an important factor affecting postburn Idaho fescue *(Festuca idahoensis)* production. The manager designed an experiment to evaluate the suspicion. An area was selected where part of the burning would be done by a backing fire and part by a heading fire. A series of 10 plots were clipped, oven-dried, and weighed preburn to establish that the two areas were from the same population and had similar variances, and to determine preburn fuel loading. The heading and backing fires were then conducted on the same day under similar environmental conditions.

Sufficient rate of spread and flame depth measurements were made during the fires to establish that residence time was significantly ($p = 0.05$) different between the two treatments. One year later the fire manager clipped, oven-dried, and weighed Idaho fescue standing crop to compare "production" on the two treatments. Based on the following results, and assuming that residence time (and its implications) was the only difference between the heading and backing fires, does residence time influence postburn production?

Quadrat	Heading	Backing	d = Heading - Backing	d^2
1	22	24	-2	4
2	15	13	+2	4
3	19	19	0	0
4	19	12	+7	49
5	21	17	+4	16
6	20	17	+3	9
7	18	19	19	1
8	20	12	12	64
9	21	15	15	36
10	14	14	14	0
n = 10	$\hat{E}_h$=189	$\hat{E}_b$=162	$\hat{E}_d$=27	$\hat{E}_d^2$=183

$x_h = 18.9$ $\qquad$ $x_b = 16.2$

a. Step 1. Note that by inspection the two means appear to be different (18.9 appears greater than 16.2). The null hypothesis, H_o, is that the two means are equal, i.e., x_h is equal to x_b. The alternate hypothesis, H_a, is set up to allow rejection, i.e., x_h is not equal to x_b. This hypothesis will result in a "two-tailed" test; one-tailed tests also can be established by setting the means in the hypotheses "greater than" or "less than" rather than "not equal to." Note that one-tailed tests require different t values than two-tailed tests.

b. Step 2. Establish the confidence level for testing. By convention, testing in this example was set at the alpha = 0.05 level (this is the same as the 95 percent confidence level, or that the acceptable risk is to be wrong one chance in 20 due to chance alone). From the t table, with n-1 degrees of freedom and an error rate of 0.05, t = 2.262. This is the tabulated value below which the difference between sample means is likely to be due to chance alone. A calculated value larger than 2.262 would suggest that a real difference exists between the two means, and on the average, this conclusion would be correct 19 of 20 times.

c. Step 3. Calculate the variance of the difference between sample means according to:

s_d^2 = Êd^2 - (Ê$(d)^2$ / n) / n - 1 = 183 - $((27)^2/10)$ / 9 = 12.2333

d. Step 4. Calculate t according to:

t = $(x_h - x_b)$ / §[s_d^2 / n] = (18.9 - 16.2) / §[12.2333 / 10] = 2.4412

§ = square root

e. Conclusion. The calculated value of 2.4412 is larger than the tabulated t value of 2.262. This suggests that a real difference exists between the two means. The fire manager may report to the supervisor that postburn Idaho fescue production was different (p = 0.05) between the heading fire and backing fire sites, and that more production occurred on the sites burned with a headfire. If it is certain that the **only** difference between treatments is residence time, it is also reasonable to assume that long residence time was more detrimental than short residence time. This conclusion is likely to be correct 19 out of 20 times. Note that this statistical significance does not necessarily imply biological significance. Also note that this "study" was not replicated, nor was a "control" treatment used; both are strongly encouraged. Also note that the method and level of testing, and the hypothesis, were established before any sampling or burning occurred.

3. Chi-Square Analysis of Counts. Chi-square is a nonparametric method that is useful for analyzing binary enumeration data that fall into two categories, such as scorched or not scorched, alive or dead, scarred or unscarred, or sprouted or not sprouted. These types of data are common in fire management. Although several procedures are available, the following example illustrates a useful method for many fire management data.

Example: One half of a plant community containing bitterbrush *(Purshia tridentata)* plants was burned in the fall and the other half was burned in the spring. The fire manager tagged 20 randomly located bitterbrush plants in each area, before burning, to estimate mortality. One year after burning, the fire manager found 11 tagged plants alive on the spring burned site and eight tagged plants alive on the fall burned site. At the 90 percent confidence level, was there a differential response between fall and spring burned plants?

a. Step 1. Set up a 2 x 2 contingency table (2 rows and 2 columns of observations):

	Spring	Fall	Total
Alive	11	8	19
Dead	9	12	21
Total	20	20	40

b. Step 2. Calculate chi-square. The general procedure, based on a 2 x 2 contingency table, is:

	I	II	Total
1	a	b	a+b
2	c	d	c+d
Total	a+c	b+d	a+b+c+d

chi-square = $([(|ad - bc|) - 1/2\ n]^2\ n) / (a + c)(b + d)(a + b)(c + d)$

chi-square = $[(|(11 \times 12) - (9 \times 8)| - 1/2\ (40)]^2\ (40)) / (20)(20)(19)(21)$ = 0.4010

Note that the vertical bar (|) indicates absolute value, so it is irrelevant which cross-multiplication product is subtracted from the other (i.e., 11 x 12 - 9 x 8, or 9 x 8 - 11 x 12). Use of the Yates Correction, [- 1/2 (n)] and [1/2(40)] found in the numerators of the above equations, decreases the chi-square value, and thus reduces the chance of declaring a significant difference when one does not exist.

c. Step 3. Look up the tabular value for chi-square in a chi-square table, such as Freese (1967, p. 82). Values in a chi-square table are different than those in a t-table, and a t-table should never be used to obtain values for a chi-square test. Degrees of freedom (df) for the 2 x 2 contingency table are calculated according to: [(rows - 1) x (columns - 1)]. In this example, df = (2-1)(2-1) = 1. Since the fire manager decided to test at the 90 percent confidence level (10 percent error level), the appropriate chi-square value to test this example is 2.71.

d. Conclusion. Since all values below 2.71 are below the 10 percent error threshold, the calculated value of 0.4010 is not significant. The manager concluded that, in this example, the sprouting of bitterbrush was not different between spring and fall fire treatments. Note that no cause and effect was implied. If a statistically significant difference had been observed, the difference could have been due to environmental factors or other unknown causes. This point is especially important for data gathered during different time periods.

4. Other Tests. Two additional statistical testing methods that are beyond the scope of this Guide but may be useful in routine fire management work include analysis of variance (often abbreviated AOV or ANOVA), and linear correlation and regression.

a. **Analysis of variance.** ANOVA is a method used to separate sources of variation *within* treatments and *among* treatments and may be used with several different experimental designs such as the completely randomized design or the randomized complete block design. ANOVA produces an "F" test of significance, and is often used with several mean separation tests, such as Duncan's New Multiple Range Test, or Orthogonal Comparisons. Although ANOVA has many potential applications in fire management, a statistician should be consulted before its use. Several texts, for example, Little and Hills (1978), provide excellent discussions of ANOVA.

b. Linear correlation and regression. Linear correlation and regression analysis are mathematical methods used to describe how independent variables relate to each other, and how dependent variables relate to independent variables. These approaches to describing and understanding relationships common in fire management work **do not** imply any cause and effect; they merely describe associations and relationships. Because the potential for misuse is so great, many statisticians discourage their use by apprentices. They are especially useful, however, when used with the aid of a reputable statistician.

5. Crunching Numbers. Development of microchip technology has made the manipulation of large data sets rapid and easy for fire managers. The three simple tests (sample size, t-test, and chi-square) that are described above can all be easily calculated on simple, 4-function, hand held calculators. Many small, hand held calculators are commercially available that have internal, preprogrammed statistical functions that quickly calculate the sample mean, variance, and sometimes, even complete linear regression. One caution is that the user should be aware whether the calculator in question uses n or n - 1 for degrees of freedom in the variance and standard deviation, and should understand the implications.

The Hewlett-Packard HP-71B hand held calculator, which is routinely used for fire behavior calculations, has several Custom Read Only Memory (CROM) modules available. One CROM (American Micro Products, Inc. 1984) contains many parametric and nonparametric tests and is powerful enough to handle relatively large data sets. Such devices make statistical analyses of fire management data routine.

Many statistical packages have been developed and are available for micro and mini-computers (e.g., StatSoft, Inc. 1987). Some packages include graphics capability, and many are available for several hundred dollars or less. Fire managers who anticipate the collection of large amounts of data for statistical manipulation are encouraged to investigate statistical packages that are available.

Mainframe statistical packages (Dixon 1985, SAS Institute Inc. 1985) are designed to handle very large data sets and complete an enormous number of statistical tests. These large machines are usually restricted to research institutions; however, fire managers should be aware that most colleges, universities, and experiment stations have access to mainframe computers. The use of such machines might be cost effective if used only when infrequent but large data sets must be manipulated.

C. Resource Management Considerations

1. It is essential to know what the questions are before sampling and data analysis can be designed to obtain answers. Sampling and data analysis must be objective driven. Further, objectives should be developed with sampling and data analysis in mind so that the objectives are reasonable, measurable, and lend themselves to analysis.

2. Sampling design and intensity (number of observations) should be determined **before** initiating any data collection to ensure sampling and analysis procedures are appropriate. The number of required observations depends on desired precision and confidence levels; for most management purposes it is usually adequate to sample within 20 percent of the mean at the 80 percent confidence level.

3. Experienced statisticians should be consulted **before** data are collected and **after** data are analyzed.

4. It is not necessary, nor feasible in most cases, to sample and analyze data from every community on every wild or prescribed fire. Usually it is better to do an adequate job on one community than an inadequate job on two or more communities. Further, it is often possible to design a series of prescribed fires with a sampling and data analysis scheme such that one fire or one stratum is emphasized and the remainder are spot checked. Biologically oriented statisticians can provide advice on how best to sample and analyze data when time and funding are constrained.

5. Inadequate sampling is often more expensive than excessive sampling; optimum sampling is usually the most cost effective.

6. Replicates are necessary to determine "sample error," and untreated "controls" are necessary to isolate fire effects from other effects.

7. After data are collected and analyzed, consider both statistical significance and biological significance; it is possible to establish statistical significance that has no biological significance.

8. Place measured trust in results of data analysis. An unexpected or undesired result is not a valid reason for discarding results and "massaging" the data. Data analysis should be used to enhance understanding as well as provide support.

9. It is tempting to draw inappropriate conclusions from analyzed data. Caution is advised.

D. Methods to Monitor Fire Effects

1. The appropriate sample size for each data set should be determined using the method described in B.1, or another statistically acceptable method. A more detailed description of this method to determine proper sample size is given in Norum and Miller (1984).

2. The t-test described in B.2 is especially useful for comparing the means of several treatments, such as burned vs. unburned, or fall burned vs. spring burned. Means that are helpful to compare include "production," cover, and density. The test assumes that identical treatments were applied; therefore, the comparison of spring burned vs. fall burned treatments should be approached with caution.

3. Chi-square is a nonparametric test that is especially useful for analyzing counts of binary data; such data fall into two convenient categories, such as alive or dead, or sprouted or not sprouted.

4. A statistician should be consulted for additional data analysis methods. Biometricians and statisticians who can provide assistance may be located at agency offices, national and regional service centers such as the BLM Service Center at Denver, experiment stations and universities.

E. Summary

Statistical analysis of data and interpretation of results are helpful for understanding fire effects and provide an essential tool for the decision making process. Calculation of the appropriate sample size is essential, and is based on desired precision and confidence levels. The t-test for paired plots, and chi-square analysis of counts, are particularly useful for understanding fire effects. Other, more sophisticated techniques may require assistance of a statistician.

CHAPTER XII - COMPUTER SOFTWARE

By Melanie Miller

A. Introduction

Many packaged computer programs have been developed that can provide information useful for managing or interpreting fire effects. Some programs are accessible on mainframe computers, while others are available on floppy disks that can be loaded onto personal computers. Additional information about programs described here can be obtained by reading the referenced publications, or contacting specific offices, where indicated.

B. Weather Analysis

Weather records can be used to manage fire effects by helping to establish seasons and parameters for appropriate fire prescriptions, and to document prefire weather trends and weather conditions at the time of the fire. Records of postfire weather can be used to make inferences about causes for observed fire effects. Long-term records are required to establish a statistical basis for occurrence of seasonal trends in elements such as temperature or rainfall, to detect deviations from seasonal trends, or to document the likelihood of occurrence of specific sets of weather conditions, such as combinations of temperature and relative humidity.

1. Computerized Databases. Sources for data on which computerized searches and analyses can be performed are the network of Remote Automated Weather Stations, and the National Interagency Fire Management Integrated Database.

a. Remote Automated Weather Stations (RAWS). The Remote Automated Weather Station (RAWS) System is a program that provides current remotely sensed weather data from stations established on Federally administered lands (German 1988). The RAWS network contains about 350 weather stations on BLM lands in the lower 48 States, as well as about 250 weather stations established on other agency lands. There are approximately 40 RAWS stations in Alaska. Over the next 5 years, approximately 60 to 100 additional RAWS stations will be established.

The RAWS system uses self-contained meteorological collection platforms and a mini-computer controlled satellite receiving station located at the National Interagency Fire Center (NIFC). The weather stations collect weather data, summarize it on an hourly basis, and then transmit data on a one hour or three hour basis to NIFC through the GOES satellite system. A computer at NIFC immediately distributes the weather data to all users on the Initial Attack Management System (IAMS), to the Weather Information Management System (WIMS) (see item 1.b. below), and to the National Weather Service. BLM users access this "real-time" weather data through the IAMS computer located in most BLM District Fire Management offices. Other agency users acquire data through WIMS.

NIFC archives all weather data for permanent storage. A copy of this archived information is distributed quarterly and annually to the Desert Research Institute at the University of Nevada at Reno. Weather information can be obtained in a quarterly summary report for each station, and through specially requested reports and studies. BLM users request these reports and studies from BLM staff at NIFC, while other agency users obtain these data from the Desert Research Institute.

b. National Interagency Fire Management Integrated Database (NIFMID). The National Interagency Fire Management Integrated Database contains the historic fire weather information previously contained in the obsolete National Fire Weather Data Library. It is located at the National Computer Center - Kansas City (NCC-KC), managed by the U.S. Department of Agriculture. The library contains daily observations from all fire weather stations, collected at the time of peak burning conditions (1300 hours, local standard time) throughout the fire season. The Weather Information Management System database (WIMS), a replacement for AFFIRMS, stores the1300-hour observations from automated and manual fire weather stations, as well as 60 days of 24-hour observations from RAWS stations. The long-term historical weather database of NIFMID stores 1300-hour observations.

Access to NIFMID is available through computer software packages that reside in the computer library at the Kansas City Computer Center. In order to use these programs, users must establish an account. Once an account is opened, all access information is provided to the user. Telecom- munications packages on most computers can be used to communicate with this USDA computer. See item 2. Computer Programs, for descriptions of computer programs available through NCC-KC.

2. Computer Programs.

a. KCFAST. KCFAST is a program that facilitates use of data stored in NIFMID. KCFAST is a utility that assists the casual user of this database by removing the need to remember all of the commands required to extract and format weather data stored for a particular weather station. KCFAST itself does not perform analyses on data sets from the fire weather data library, but it makes it easy to transmit these data to a personal computer, where weather analysis can be performed. It facilitates operation of the other programs described in this section, as well as other software such as that used for fire planning. KCFAST is available in versions that operate on the U. S. Forest Service Data General system, as well as IBM compatible personal computers. To use the PC version, communication software is required that can transmit an ASCII file to a remote computer and log screen activity to a PC file (Bradshaw and Andrews 1990).

b. FIREFAMILY. Three different computer programs were consolidated into FIREFAMILY: FIRDAT, SEASON, and FIRINF. These programs use historic weather data to predict future fire management needs, and integrate fire management with other land management activities. The weather data used is that stored in the NIFMID (1.b., above), and FIREFAMILY software is on the USDA Kansas City computer. These programs are fully described in the publication by Main et al. (1988).

(1) FIRDAT. FIRDAT can combine up to 100 years of historical weather records with National Fire Danger Rating System (NFDRS) equations to produce frequency distributions and graphs of NFDRS indices and components. This can include the NFDRS calculated fuel moisture. FIRDAT can also produce lists of daily weather observations.

(2) SEASON. SEASON uses FIRDAT data to summarize variations in NFDRS inputs and fire danger severity during a fire season, and seasonal patterns of fire danger over many years. Fuel moisture values, fire danger indices, or fire behavior components can be presented in tabular and graphic form.

(3) FIRINF. FIRINF allows the analysis of the co-occurrence of any two NFDRS indices over 10-year periods.

c. PCFIRDAT. PCFIRDAT is a version of FIRDAT that has been rewritten for use on IBM compatible computers. Using the PC version of KCFAST, data are transferred from the NIFMID to a personal computer, and PCFIRDAT is then used to perform data analyses. PCFIRDAT, as well as PC versions of SEASON and FIRINF are in the final stages of development. Work is being performed by the California Department of Forestry.

d. PRESCRB. This program provides a climatological summary of specific fire weather occurrences. The program counts the occurrences of days on which all variables specified in a prescription are met, and gives the probable number of days in any year on which the prescribed conditions specified are likely to occur. The average number of days of occurrence of prescribed conditions during successive 10-day periods of the year is given. The average number of days in each burning period also can be obtained, giving the prescribed fire planner an idea of whether days meeting prescribed conditions are likely to occur singly, or in a series. Furman (1979) documents use of the PRESCRB program. An advantage of this program is that it can use weather data from other sources than NIFMID.

e. RXWTHR. RXWTHR (Prescribed Fire Weather) provides climatological summaries and co-occurrence frequencies of user selected fire weather and fire danger rating parameters (Bradshaw and Fischer 1981a, 1981b). The data base is the computerized fire weather records within NIFMD. Simultaneous occurrence of two or three of fifteen prescription parameters can be summarized, and shown in tables. For example, tables can include summaries of temperatures, two way co-occurrence tables for wind direction and wind speed, and three way co-occurrence tables for temperature, relative humidity, and wind speed. Once a user has obtained a feeling for the general pattern of occurrence of desired prescribed conditions, screening out those that have a low probability of occurring simultaneously, a more detailed analysis can be conducted using RXBURN. Please note that a sixteenth prescription parameter available for analysis in the program is duff moisture. This value is based upon a duff moisture model that was offered on an experimental basis in RXWTHR and RXBURN. This model does not provide accurate results, and the duff moisture output produced by these two programs should not be used.

f. RXBURN. RXBURN (Prescribed Fire Conditions) provides an analysis of the frequency of occurrence of a set of prescription conditions, also using the NIFMID as a database. Up to 15 parameters can be used in a single prescription. Users define a preferable range of prescribed conditions, and a broader range of conditions that is still acceptable. Output tables include a summary table of the percentage of weather conditions that are preferable, acceptable, and unacceptable; a table that shows frequency of occurrence of preferable, acceptable, and unacceptable conditions in each successive 10-day period, and by month; the number of successive days that prescribed conditions have occurred in each 10-day period and each month; and the probability of meeting the prescription 1, 2, and 3 days in the future for each month. As discussed under 2.d., RXWTHR, the duff moisture model that provides a basis for parameter 16 is inaccurate; this parameter should not be used.

There are two limitations to the use of RXWTHR and RXBURN. Weather observations are rarely recorded for more than the five months of the year that are the normal "fire season," and many prescribed fires are staged before and after this period. The single observation per day that must represent the entire day's weather is taken during average worst case conditions, 1300 hours (Standard Time), southwest aspect, midslope, and open canopy. This may poorly represent the weather at the time of the day when a prescribed fire would be implemented.

g. Other. An additional set of climatological software is described by Bradshaw and Fischer (1984). Eight computer programs for extensive climatic summaries of weather variables, temperature, relative humidity, wind, and precipitation, records are available. Five basic climatology programs analyze NIFMID records by 10-day periods and by month. Three averaging programs adjust results from the climatology programs to smooth variances caused either by short periods of record or incomplete station data.

C. Fire Behavior

1. BEHAVE. BEHAVE is a set of interactive, user friendly computer programs that are used for estimating behavior of wild and prescribed fires. BEHAVE is an integral part of the Fire Behavior Prediction System that is used by Fire Behavior Analysts to estimate fire potential under various fuels, weather, and topographic situations (Burgan and Rothermel 1984). BEHAVE predicts the behavior of a steady state fire advancing in surface fuels along a front. It cannot be used to estimate long-term fuel burnout, non-flaming combustion, or extreme fire behavior. The BEHAVE system consists of two subsystems: BURN and FUEL.

a. FUEL subsystem. The FUEL subsystem has two programs: NEWMDL and TSTMDL (Burgan and Rothermel 1984). NEWMDL ("NEW MODEL") is used to construct fuel models for specific application when the existing 13 stylized fuel models in the Fire Behavior Prediction System do not adequately describe fuels at a particular location. TSTMDL ("TEST MODEL") is used to test the fuel models developed in NEWMDL. The FUEL subsystem is rarely used except by trained Fire Behavior Analysts or for research purposes. Please note that these fuel models are not the same as those used in the National Fire Danger Rating System.

b. BURN subsystem. The BURN subsystem is frequently used on wild and prescribed fires. This subsystem has three components, FIRE 1, FIRE 2, and RXWINDOW (see section C.2.). FIRE 1 contains the wildland fire behavior prediction model developed by Rothermel (1972). It includes modules that allow prediction of fire behavior (DIRECT and SITE), fire growth (SIZE), containment requirements (CONTAIN), and spotting distance (SPOT). Fire effects on the tree overstory are predicted in SCORCH and MORTALITY, described in sections D.1.a., and D.1.b., this chapter. FIRE 2 allows calculation of fine dead fuel moisture (MOISTURE), the probability of ignition (IGNITE), and relative humidity (RH). Contents and operation of the BURN Subsystem are described in Andrews (1986) and Andrews and Chase (1989). Users require training for optimum application of this software to wildland fire situations.

2. RXWINDOW. RXWINDOW, a component of the BURN subsystem in BEHAVE, essentially runs DIRECT backwards, enabling prescribed fire planners to obtain detailed windows of required fuel moisture and wind conditions based upon desired fire behavior (Andrews and Bradshaw 1990). Desired fire behavior such as flame length or rate of fire spread are entered, and the program calculates which combinations of fuel and weather parameters would result in the fire behavior specified. The program uses one of the 13 standard fire behavior fuel models, or a custom model developed in the FUEL Subsystem. Other input values include slope of the burn area and exposure of fuels to the wind. The user can optionally set limits on 1-hour, 10-hour, 100-hour, live woody, and live herbaceous moisture content, and effective windspeed.

a. FIRE. The FIRE module generates tables that display combinations of effective windspeed and weighted fuel moisture that result in the desired fire behavior. It indicates those pairs of wind and dead fuel moisture that yield a fire within the fire behavior prescription. For those fuel models where live fuel moisture is a component, the program prints ranges of live fuel moisture for each appropriate wind and dead fuel moisture pair.
b. WIND. The WIND module requires input of site description parameters, and whether the prescribed fire will be a headfire, backfire, or flanking fire. Based on a constant slope, the program prints a table of 20-foot windspeeds coming from different directions with respect to slope that will produce a range of effective windspeeds. WIND can use either effective windspeeds identified in the FIRE output table or windspeeds selected by the user.

c. MOISTURE. The MOISTURE module of RXWINDOW displays ranges of moisture contents for the 10-hour fuel size class that results in a specific weighted fuel moisture, for a given 1-hour fuel moisture. MOISTURE also produce a table that shows for a given herbaceous moisture content, the range of woody fuel moisture that results in a specific weighted live fuel moisture.

3. Availability. Contact agency fire management staff for information on obtaining access to FIRE 1, NEWMDL, and TSTMDL. A copy of the complete BEHAVE program (including FIRE 2 and RXWINDOW) that runs on an IBM compatible computer can be purchased through a government contract at a low cost. The program disks will become available through the Publications Management System at the National Interagency Fire Center.

D. Fire Effects

1. Models within BEHAVE. There are few fire effects that relate directly to fire behavior, the activity of the flaming front of the fire. However, two programs within the BEHAVE system predict two effects, crown scorch height, and tree mortality, that can be directly or indirectly related to fire behavior. Access to both of these programs is through the BEHAVE system, described in section C. of this chapter.

a. SCORCH. A module of the FIRE 1 program of BEHAVE, SCORCH predicts lethal crown scorch heights from flame length, ambient air temperature, and midflame windspeed. This model estimates the maximum height in the convection column at which the lethal temperature for live crown foliage is reached, assumed to be 140 F (60 C). Scorch heights can be estimated during a prescribed fire operation, based on observations of the flame length, and ignition can be adjusted accordingly if desired scorch heights are not being achieved. SCORCH can be run by linking it to outputs from DIRECT. A more detailed description of this model can be found in Andrews and Chase (1989).

The SCORCH model has several limitations. It was developed for flat ground, so should be used on slopes with care. The model may not be valid outside the range of conditions for which it was developed, with fireline intensities ranging from 19 to 363 Btu per foot per second, equivalent to flame lengths of 1.8 to 6.8 feet (0.5 to 2.1 meters). Air temperatures are 73 to 88 F (23 to 31 C), and midflame windspeeds are 1.5 to 3 miles per hour (3.4 to 4.8 kilometers per hour). Under these conditions, scorch heights ranged from 6.5 to 56 feet (2 to 17 meters). Also, this model considers the heat released by the flaming front of the fire, not from the long-term burnout of large fuels. If significant amounts of residual burnout of large diameter fuels is anticipated, expected or observed flame length resulting from this burnout can be entered directly, instead of using the flame length calculated by BEHAVE in DIRECT.

b. MORTALITY. Tree mortality is predicted by the MORTALITY module of BEHAVE, also located in the FIRE 1 program. The model was developed by Reinhardt and Ryan (1988b), and its use within the BEHAVE program is described in Andrews and Chase (1989). MORTALITY predicts the percentage of tree mortality from estimates of crown and bole damage for a specific species, as different species have varying abilities to survive a set amount of damage. Data on mortality was collected from the following species, which can be specified when running the model: western larch, Douglas-fir, western hemlock, Engelmann spruce, western red cedar, lodgepole pine, and subalpine fir.

One can select one of these species to represent a species not listed if the bark thickness is similar. Inputs required to run this program are scorch height, tree height, crown ratio, and bark thickness, which is calculated from the species of tree and its diameter at breast height. Percentage of crown volume scorched is calculated from scorch height, tree height, and crown ratio. Bole damage is assumed to be proportional to bark thickness. The output is given in percent mortality. A 30 percent mortality means that 30 of 100 trees would be expected to die if subjected to the same fire, or that there is a 30 percent probability that any individual tree would die.

A linked run can be made from DIRECT to SCORCH to MORTALITY. Flame length can be calculated for a range of windspeeds in DIRECT. From these values, SCORCH will calculate a range of scorch height values, which provide one of the inputs to MORTALITY.

The model is limited by the assumption that the fire is of an average duration. Mortality may be under predicted if a fire of long duration occurs, caused by consumption of extremely dry duff or large diameter fuel. If fuel is very light or patchy, mortality may be over estimated.

2. First Order Fire Effects Model (FOFEM). The FOFEM program computes duff and woody fuel consumption, mineral soil exposure, fire-caused tree mortality, and smoke production for forest stands (Keane et al. 1990). A current version of the program is being tested by field users. Future versions will allow prediction of soil heating and successional changes. Default input values are derived from fuel models provided for natural and activity fuels by many forest cover types. For further information, contact the Fire Effects Research Work Unit at the Intermountain Fire Sciences Laboratory, Missoula, Montana.

3. CONSUME. CONSUME is a PC based software program that predicts the amount of fuel consumption on logged units based on weather data, the amount and moisture content of fuels, and other factors that describe a burn unit (Ottmar et al. 1993). The program allows a resource manager to determine when and where to conduct a prescribed fire to achieve desired objectives, while reducing impacts on other resources. CONSUME can be used for most broadcast and understory burns on forest lands where the dead woody fuels are relatively homogeneous. The program applies to western forests dominated by Douglas-fir, western hemlock, red alder, lodgepole pine, or mixed conifer species. Program disks and the users guide are available through the Publications Management System at NIFC.

4. Fire Effects Information System. The Fire Effects Information System (FEIS) is a computerized information storage and retrieval system that contains detailed information about the effects of fire on specific plants, plant communities, and wildlife species. Plant species information, for example, is organized into sub-categories of distribution and occurrence; value and use; botanical and ecological characteristics; fire ecology; and fire effects. Descriptions of the results of fire effects case studies are included if available. The FEIS is not a typical bibliographic data base that lists citations with key words and abstracts. The Fire Effects System provides information in a text format, providing reviews of the key facts in the literature, summarized into appropriate sub-categories. Numerical codes in the text refer to citations listed in a references section included in each species write-up. Where conflicting information about plant response has been found, interpretations are made if differences in season, burning conditions, or ecotype, for example, can explain why observed variations occurred. Information about wildlife species and ecosystems is handled in a similar fashion.

The system was developed by the U.S. Forest Service Intermountain Fire Sciences Laboratory in Missoula, Montana, and money for development of the prototype data base was provided by the BLM. Many plant species of ecosystems managed by the Bureau are presently in the system. Interagency funding is being used to expand the database to include species of the eastern U.S. and Alaska, as well as additional species of the western U.S.

The Citation Retrieval System (CRS) is an associated program that contains all of the references used in compiling species and plant community information for the FEIS. The CRS can be searched for a specific citation, author, or key word. It can prepare a bibliography from a list of citation index numbers selected by the user from a species write-up.

The FEIS and CRS are available on the Forest Service Intermountain Region computer in Ogden, Utah. Access is through the Forest Service Data General System, or by phone modem from any personal computer with software that allows communication with a Data General computer. There is no cost for PC users other than telephone line charges. Data can be saved to a temporary file on the main frame computer, and sent over phone lines to the user's computer. Department of Interior employees can contact their national fire management offices to obtain access information. U.S. Forest Service employees can obtain assistance from their regional FEIS coordinator. States provide information through their State FEIS coordinator.

E. Smoke Modeling.

1. SASEM. The Simple Approach Smoke Estimation Model (SASEM) is a tool for the analysis of smoke dispersion from prescribed fires (Sestak and Riebau 1988). It is a screening model, in that it uses simplified assumptions and tends to over predict impacts, yielding conservative results. If violations of air quality standards are not predicted by SASEM, it is unlikely that they will occur. Inputs to the model include basic descriptions of the fuels, such as type and loading, expected fireline intensity, and expected burn duration. Windspeed and direction, dispersion conditions, and average mixing height are considered, as well as distance and direction of the fire from sensitive receptors. The model calculates fuel consumption and particulate emission factors from fuel loading and expected fireline intensity. Model outputs include maximum particulate concentration and the distance from the fire at which it will occur, ranges of distances from the fire at which any primary or secondary particulate standards would be violated, and the reduction in visual range at selected receptors. Outputs are given in tabular fashion for a range of dispersion and windspeed conditions.

SASEM is extremely simple to use, and requires no data inputs that are not normally acquired as part of the prescribed fire planning process. The program is available on floppy disks for operation on IBM compatible machines. For further information, contact agency air resource or fire management specialists, or the air quality staff at the Environmental Science and Technology Center, National Biological Survey, Fort Collins, Colorado.

2. TSARS. The following discussion is taken from Sestak (et al. 1991). The Tiered Smoke/Air Resources System, TSARS, allows fire management field officers to test fire prescriptions for smoke management problems. Models with a high degree of rigor can be used to solve more complex problems, however, higher user proficiency is required.

a. Existing components. Models currently in the system include SASEM, explained in item E.1. above, EPM, and VALBOX.

(1) The Emission Production Model, EPM, is a more elaborate model of heat and particulate production than SASEM. Originally designed for forest fuels, particularly logging slash, data are presently being collected to broaden the model to other fuel types, particularly rangeland fuels.

(2) VALBOX models an airshed in complex terrain. Air in a mountain valley is divided into a series of connecting boxes, with dimensions calculated from topographic map data. This model best describes conditions when an inversion exists and air is stagnant.

b. Proposed components. BEHAVE, described in Section C. of this chapter, when added to the TSARS system, will expedite the running of EPM and SASEM because many of their input values are contained in the standard fire behavior fuel models used in BEHAVE. Use of these models together will allow consideration of smoke impacts when developing prescribed fire prescriptions.

TAPAS, the Topographic Air Pollution Analysis System, is a set of meteorological and pollution dispersion models suitable for use with multiple emission sources in complex terrain. Two models contained in TAPAS, WINDS and CITPUFF, will be specialized in the TSARS program for use in prescribed fire planning. WINDS is a two dimensional wind field model. A three dimensional wind model in TAPAS can be used by if elevation grid databases are created, information that is generally available in in Geographic Information Systems.

Wind fields created by the two or three dimensional models can be used with a final proposed component of TSARS. CITPUFF approximates the dispersion of a pollutant as it follows a path across a simulation area. The emission information required by this model would be provided by SASEM or EPM emission calculations.

It is intended that TSARS be available on the second generation IAMS system, and also on IBM compatible personal computers. For more information, contact air quality staff, Environmental Science and Technology Center, National Biological Survey, Fort Collins, Colorado.

F. Library Services

The literature search services maintained by Federal Libraries are valuable sources of computerized information helpful for managing fire effects. The Bureau of Land Management Library at the Denver Federal Center, for example, has the capability of making on-line searches of several hundred data bases that include periodicals, books, reports, and other publications. Some data bases provide abstracts along with full citations. The user works with a Library staff person by defining the kind of information desired, as well as specifying key words to be used in the search. Bibliographies can be requested by subject area, author, report number, or other category. The library can also provide copies of articles, and loan books in their collection and through inter-library loan. Any BLM employee can contact the Library at the BLM Service Center in Denver for assistance. The phone number is 303/236-6646.

The Alaska Resources Library provides a full range of library services to all Federal employees, particularly in the natural resources field. The Library is located in the Federal Building in Anchorage, where it is administered by the BLM. The phone number is 907/271-5025; FAX is 907/271-5965.

The Department of the Interior Natural Resources Library provides a full range of library services to USDI employees in response to telephone and written requests. They are located in the Main Interior Building, Washington, D.C.; their phone number is 202/208-5815.

National Park Service employees can contact the NPS Service Center at Harper's Ferry, Virginia or Denver for library assistance. The phone number at Harper's Ferry is 304/535-6371; Denver is 303/969-2100.

U. S. Forest Service employees can obtain a full range of library services through FS INFO, available to them on the Data General System through the Information Center process. Both Forest Service Research and National Forest System Employees can seek assistance from the FS INFO center located in at the Forest and Range Experiment Station in their geographic region.

G. Summary

Computer technology and applications are developing so quickly that any list of software is incomplete as soon as it is published. Specialized computer programs, called expert systems, may be available in the next few years. Expert systems are being developed or planned that can assist in the development of fire prescriptions to meet specific resource objectives, and to achieve specific fire effects. Agency fire management and air quality specialists can be contacted for information about future computer software development.

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GLOSSARY OF TERMS

- A -

absolute value: the absolute value represents the distance that a positive or negative number is from zero, when numbers are arrayed on a line with negative numbers to the left of zero and positive numbers to the right of zero. The absolute value of a positive number is the number itself, whereas the absolute value of a negative number is the opposite (positive) number (Batchelet 1976).

accelerated erosion: erosion much more rapid than normal, natural, or geologic erosion, primarily as a result of the influence of the activities of humans, or, in some cases, of other animals or natural catastrophes that expose bare surfaces, for example, fires.

accuracy: the closeness of a measured or computed value to its true value. Accuracy cannot be determined from a sample, and usually remains unknown. (See precision.)

active crown fire: a fire in which a solid flame develops in the crowns of trees, but the surface and crown phases advance as a linked unit dependent on each other.

active layer: soil layer that overlies permafrost that thaws every summer (Viereck and Schandelmeier 1980).

activity fuels: fuels resulting from, or altered by, forestry practices such as timber harvest or thinning, as opposed to naturally created fuels. (See natural fuels.)

aeolian (eolian): movement of material, such as soil, through wind action.

aerial fuels: the layer of fuels that is above the surface fuels, including living tree and shrub crowns, mosses, lichens, vines, and dead branch material.

age class: classes that define the ages of individuals in discrete units, such as 0 to 5 years, or 6 to 20 years.

algal bloom: proliferation of living algae on the surface of lakes, streams, or ponds that is stimulated by nutrient, especially nitrogen or phosphate, enrichment.

alkaline soil: any soil with a pH value greater than 7.0. Often used interchangeably with "basic soil."

allelopathy: inhibition by a plant of the germination, establishment, and growth of seedlings.

ammonification: the biochemical process whereby ammoniacal nitrogen is released from nitrogen containing organic compounds.

analysis of covariance (ACOVA): test that combines regression with analysis of variance. In general, variation in *y* that is associated with *x* is removed from the error variance, which results in more precise estimates from more powerful tests. An example in fire management is the case where plants of different sizes are measured before burning; because plant size may affect postburn response, the effect of plant size is "controlled" so that differences due to burning will be apparent.

analysis of variance (ANOVA or **AOV):** test used when statistical inferences are to be made about more than two means. Test can "control" one or more sources of variation, depending on model and experimental design.

anion: negatively charged ion.

annual plant: a plant completing its life cycle in a year or less (Benson 1967).

Aridisol: soil with pedogenic horizons, low in organic matter, that are never moist as long as three consecutive months. They also have one or more of the following diagnostic horizons, including argillic, natric, cambic, calcic, petrocalcic, gypsic, or salic, and may have a duripan. Aridisols are approximately equal to Desert, Reddish Desert, Sierozem, Solonchak, some Brown and Reddish Brown, and associated Solonetz soils of the pre-1966 classification scheme.

arithmetic mean: the mean (or average) of sample data; it provides an unbiased estimate of the parametric mean; designated by x. (NOTE: There are other kinds of means, such as harmonic, geometric, and parametric.)

arrangement: see fuel arrangement.

ASCII: American Standard Code for Information Interchange. A code for transmitting information asynchronously on local and long distance communication lines; representing a standard set of letters, numbers, and control characters.

attributes: those variables that cannot be measured, but must be expressed qualitatively. (See variable.)

available fuel: the portion of the total fuel on the site that would actually burn under a given set of environmental conditions.

avoidance: a smoke emission control strategy that considers meteorological conditions when scheduling prescribed fires in order to avoid incursions into smoke sensitive areas (Mathews et al. 1985).

axil: the upper side of the point where a leaf meets a stem, or a branch meets another branch or the main stem of a plant.

axillary bud: a bud in a leaf axil.

- B -

backing fire: a fire, or that part of a fire, spreading or set to spread into the wind, or down a slope.

basal cover: the vertical projection of the root crown onto the ground. (See cover and foliar cover.)

BEHAVE: a system of two interactive computer programs for modelling fuel and fire behavior.

bias: systematic distortion that may be due to measurement error, method of selecting the sample,

etc. An example is the use of an uncalibrated balance that consistently overestimates mass by 10 grams, or neglecting to subtract tare weight (weight of the container) from packaged samples.

biennial plant: a plant that completes its life cycle in two years. Biennial plants usually produce only

basal leaves above ground the first year and both basal leaves and flowering stems the second (Benson 1967).

broadcast burning: allowing a prescribed fire to burn over a designated area within well-defined boundaries for reduction of fuel hazard, as a resource management treatment, or both.

Btu: British thermal unit. The amount of heat needed to raise the temperature of one pound of water (one pint) one degree Fahrenheit.

bud: a vegetative growing structure at the tip of a stem or branch with the enclosing scale leaves or

immature leaves; a young flower bud that has not yet opened. A vegetative bud may also be located along the surface of roots or rhizomes, or buried in woody stem or root tissue (Benson 1967).

bud primordia: a cluster of plant cells with the physiological potential to develop into a bud or actively growing shoot.

bulb: an underground bud covered by fleshy scales, the coating formed from the bases of leaves (Benson 1967).

bulk density: weight per unit volume. For fuels, this is usually expressed as pounds per cubic foot; for soils, grams per cubic centimeter.

burl: a mass of woody tissue from which roots and stems originate, and which is often covered with dormant buds (James 1984).

burn severity: a qualitative assessment of the heat pulse directed toward the ground during a fire. Burn severity relates to soil heating, large fuel and duff consumption, consumption of the litter and organic layer beneath trees and isolated shrubs, and mortality of buried plant parts.

- C -

cambium: a layer of dividing plant cells which add during each growing season a layer of woody material (largely xylem) on their inner side toward the center of the stem or root and a layer of bark (phloem and associated tissues) on the outer side (Benson 1967).

carbohydrates: starches and sugars manufactured by a plant and used to provide energy for metabolism, and structural compounds for growth (Trlica 1977).

carrier fuels: the fuels that support the flaming front of the moving fire.

cation: positively charged ion. Common soil cations include calcium, magnesium, sodium, potassium, and hydrogen.

caudex: a largely underground stem base which persists from year to year and each season produces leaves and flowering stems (Benson 1967).

chain: unit of measure in land survey, equal to 66 feet (20 meters) (80 chains equal one mile). Commonly used to report fire perimeters and other fireline distances, chains can be easily converted to acreage (e.g., 10 square chains equal one acre).

Class 1 Area: geographic areas designated by the Clean Air Act where only a very small amount or increment of air quality deterioration is permissible. Class 1 Areas include specified National Parks, Wilderness Areas, and certain Indian reservations.

clastic: being able to readily break into small fragments or pieces.

climax: the highest ecological development of a plant community capable of perpetuation under the prevailing climatic and edaphic conditions (Range Term Glossary Committee 1974).

clone: a group of individuals propagated vegetatively from a single individual of seedling origin (Barnes 1966 in Jones 1985). Individuals in a clone are genetically the same plant.

colonizer: species that establish on a burned (or otherwise denuded) site from seed (Stickney 1986).

combustion: consumption of fuels by oxidation, evolving heat, flame, and/or incandescence.

combustion efficiency: the relative amount of time a fire burns in the flaming phase of combustion, as compared to smoldering combustion. A ratio of the amount of fuel that is consumed in flaming combustion compared to the amount of fuel consumed during the smoldering phase, in which more of the fuel material is emitted as smoke particles because it is not turned into carbon dioxide and water.

confidence level: the percentage confidence that a statement is true. If data are normally distributed about an average value, the statement 'within 10 percent of the mean at the 80 percent confidence level' would indicate that a sample is likely to be within 10 percent of the true average value of the population 80 percent of the time.

continuity: see fuel continuity.

continuous variables: those variables which, at least theoretically, can assume an infinite number of values between any two fixed points. An example is fuel load, which theoretically can be anywhere between zero and infinity. (See discrete variables.)

control strategy: a way of implementing a prescribed fire that manages smoke output. (See avoidance, dilution, and emission reduction.)

convection column: the thermally induced ascending column of gases, smoke, water vapor, and particulate matter produced by a fire.

cookbook approach: to follow a particular procedure without deviation.

coordinated resource management: a process that directly involves everyone concerned with resource management in a given planning area.

corm: a bulb-like structure formed by enlargement of the stem base, sometimes coated with one or more membranous layers (Benson 1967).

cover: the area on the ground covered by the combined aerial parts of plants expressed as a percent of the total area. (See basal cover and foliar cover.)

criteria pollutants: those air pollutants designated by the Environmental Protection Agency as poten- tially harmful and for which ambient air standards have been set to protect the public health and welfare. The criteria pollutants are carbon monoxide, particulate matter, sulfur dioxide, nitrogen dioxide, lead, and ozone.

crown consumption: combustion of the twigs, and needles or leaves of a tree during a fire.

crown fire: a fire that advances by moving among crowns of trees or shrubs.

crown ratio: the ratio of live crown to tree height.

crown scorch: causing the death of tree foliage by heating it to lethal temperature during a fire, although the foliage is not consumed by the fire. Crown scorch may not be apparent for several weeks after the fire.

crown scorch height: the height above the surface of the ground to which a tree canopy is scorched.

crowning potential: a probability that a crown fire may start, calculated from inputs of foliage moisture content and height of the lowest part of the tree crowns above the surface.

- D -

data: the facts collected during observations; used as a plural noun, i.e., "data are," or "data were." (See datum.)

datum: a single fact collected during observation; the singular of "data." (See data.)

dead fuels: naturally occurring fuels without living tissue, in which the moisture content is governed almost entirely by absorption or evaporation of atmospheric moisture (relative humidity and precipitation).

decreaser species: plant species of the original vegetation that decrease in relative amount under overuse by grazing or browsing animals. Commonly termed decreasers.

degrees of freedom: the quantity n or n-1, where n is the number of observations (population or sample size) upon which a variance has been based; degrees of freedom is usually designated by the abbreviation "*df*."

density: the number of plants or parts of plants per unit area.

designated area: those areas identified as principal population centers or other areas requiring protection under state or federal air quality laws or regulations.

desired plant community: a plant community which produces the kind, proportion, and amount of vegetation necessary for meeting or exceeding the land use plan goals and activity plan objectives established for the site.

diffusion: the net movement of gas molecules from areas with higher concentration of that gas to areas with lower concentration.

dilution: a control strategy used in managing smoke from prescribed fires in which smoke concentration is reduced by diluting it through a greater volume of air, either by scheduling during good dispersion conditions or burning at a slower rate (Mathews et al. 1985).

disclimax: (disturbance climax) a stable plant community which is not the climatic or edaphic climax and which is perpetuated by man or his domestic animals (Odum 1966).

discrete variables: those variables that have only certain fixed numerical values, with no intermediate values possible in between; also known as discontinuous or meristic variables. An example is the number of offspring per litter, where only integers (whole numbers) are possible. (See continuous variables.)

diversity: the relative degree of abundance of wildlife species, plant species, communities, habitats, or habitat features per unit of area (Thomas 1979).

duff: the partially decomposed organic material of the forest floor that lies beneath the freshly fallen twigs, needles and leaves. The fermentation and humus layers of the forest floor (Deeming et al. 1977).

- E -

ecological condition (range): the existing state of the vegetation on a site compared to the natural potential (climax) plant community for that site. This term is used interchangeably with "range condition" and describes the deviation from the climax condition according to four arbitrary condition classes. Not synonymous with "forage condition" which does not relate to site potential.

ecological niche: the role or function a particular organism plays in the environment (Hanson 1962).

ecological site: a distinctive geographic unit that differs from other kinds of geographic units in its ability to produce a characteristic natural plant community. An ecological site is the product of all the environmental factors responsible for its development. It is capable of supporting a native plant community typified by an association of species that differs from that of other ecologic sites in the kind or portion of species or in total production.

ecology: the study of the interrelationships of organisms with one another and with the environment (Hanson 1962).

ecosystem: an interacting natural system including all the component organisms together with the abiotic environment (Hanson 1962).

ecotone: the area influenced by the transition between plant communities or between successional stages or vegetative conditions within a plant community (Thomas 1979).

edge: the place where plant communities meet or where successional stages or vegetative conditions within plant communities come together (Thomas 1979).

edge effect: the increased richness of flora and fauna resulting from the mixing of two communities where they join (Thomas 1979).

effective windspeed: a value that combines the speed of the wind with the additive effect of slope when a fire is burning up or across a slope.

emission: a release into the outdoor atmosphere of air contaminants such as smoke.

emission factor: the mass (weight) of particulate matter produced per unit mass of fuel consumed (expressed as grams per kilogram or pounds per ton).

emission inventory: a listing by source of the amounts of air pollutants discharged into the atmosphere.

emission reduction: a strategy for controlling smoke from prescribed fires that minimizes the amount of smoke output per unit area treated.

equilibrium moisture content: the moisture content that a fuel particle will attain if exposed for an indefinite period in an environment of specified constant temperature and humidity. When a fuel particle has reached its EMC, the net exchange of moisture between it and its environment is zero (Deeming et al. 1977).

eurytopic: having a wide range of suitable ecological conditions (Pennak 1964).

eutrophication: the process whereby water becomes excessively rich in nutrients and correspondingly deficient, at least seasonally, in oxygen. Often accompanied or followed by algal blooms.

exfoliation: the separation of concentric layers of rock from the original rock mass.

experimental design: the process of planning an experiment or evaluation so that appropriate data will be collected, which may be analyzed by statistical methods resulting in valid and objective conclusions. Examples include Completely Random Design, Randomized Block Design, Latin Square, Factorial Experiment, Split-Plot Design, and Nested Design.

experimental error: a measure of the variation which exists among observations on experimental units that are treated alike; "natural" variation.

- F -

feeder roots: small diameter roots that collect most water and nutrients for a plant, usually located near the soil surface.

fine fuels: small diameter fuels such as grass, leaves, draped pine needles, and twigs, which when dry, ignite readily and are rapidly consumed.

fire behavior: the manner in which a fire burns in response to the variables of fuel, weather, and topography.

fire intensity: see fireline intensity.

Fire Behavior Prediction System: a system that uses a set of mathematical equations to predict certain aspects of fire behavior in wildland fuels when provided with data on fuel and environmental conditions (Rothermel 1983).

fire regime: periodicity and pattern of naturally occurring fires in a particular area or vegetative type, described in terms of frequency, biological severity, and areal extent (Tande 1980).

fire severity: see burn severity.

fire spread model: a set of physics and empirical equations that form a mathematical representation of the behavior of fire in uniform wildland fuels (Rothermel 1972).

fire treatment: the use of prescribed fire to accomplish a specified objectives.

fire whirl: a spinning, vortex column of ascending hot air and gases rising from a fire and carrying aloft smoke, debris, and flame. Fire whirls range from a foot or two in diameter to small tornadoes in size and intensity. They may involve the entire fire area or only a hot spot within the area.

fireline intensity: the heat released per unit of time for each unit length of the leading fire edge. The primary unit is Btu per lineal foot of fire front per second (Byram 1959 in Albini 1976).

firing pattern: see ignition pattern.

firing technique: see ignition pattern.

first order fire effects (FOFE): the direct and immediate effects of fire.

flame length: the average length of flames when the fire has reached its full, forward rate of spread, measured along the slant of the flame from the midpoint of its base to its tip.

flaming phase: the phase of combustion in which gases distilled from fuels rapidly combine with atmospheric oxygen, producing visible flames.

FLPMA: the Federal Land Policy and Management Act of 1976 (Public Law 94-579, 90 Stat. 2743, 43 USC 1701).

foliar cover: the projection of all plant parts vertically onto the ground. (See cover, and basal cover.)

forb: a plant with a soft, rather than permanent woody stem, that is not a grass or grasslike plant.

forward rate of spread: the speed with which a fire moves in a horizontal direction across the landscape, usually expressed in chains per hour or feet per minute.

frequency of occurrence: a quantitative expression of the presence or absence of individuals of a species in a population; the ratio between the number of sample units that contain a species and the total number of sample units.

fuel: combustible plant material, both living and dead that is capable of burning in a wildland situation.

fuel arrangement: the spatial distribution and orientation of fuel particles within a fuel bed.

fuel bed: an array of fuels usually constructed with specific loading, depth, and particle size, to meet experimental requirements; also commonly used to describe the fuel composition in natural settings.

fuel bed depth: average height of surface fuels contained in the combustion zone of a spreading fire front.

fuel continuity: the degree or extent of continuous or uninterrupted distribution of fuel particles in a fuel bed, a critical influence on a fire's ability to sustain combustion and spread. This applies both to aerial fuels and surface fuels.

fuel depth: see fuel bed depth.

fuel loading: the weight of fuels in a given area, usually expressed in tons per acre, pounds per acre, or kilograms per square meter.

fuel model: a characterization of fuel properties of a typical field situation. A fuel model contains a complete set of inputs for the fire spread model.

fuel moisture content: the amount of water in a particle of fuel, usually expressed as a percentage of the oven dry weight of the fuel particle.

fuel size class: a category used to describe the diameter of down dead woody fuels. Fuels within the same size class are assumed to have similar wetting and drying properties, and to preheat and ignite at similar rates during the combustion process.

- G -

glowing phase: phase of combustion in which a solid surface of fuel is in direct contact with oxygen, and oxidation occurs, usually accompanied by incandescence, and little smoke production.

graminoid: grasslike plant, including grasses, sedges, rushes, reeds, and cattails.

gravimetric: of, or pertaining to, measurement by weight.

grazing management (strategy): the manipulation of the grazing use on an area in a particular pattern, to achieve specific objectives.

grazing pattern: dispersion of livestock grazing within a management unit or area.

ground fire: fire that burns the organic material in the soil layer (e.g. a "peat fire") and often also the surface litter and low-growing vegetation.

ground fuels: all combustible materials below the surface litter layer, including duff, tree and shrub roots, punky wood, dead lower moss and lichen layers, and sawdust, that normally support glowing combustion without flame.

growth stage: the relative ages of individuals of a species, usually expressed in categories such as seedlings, juvenile, mature, and decadent.

guild: see habitat guild.

- H -

habitat: the sum total of environmental conditions of a specific place occupied by a wildlife species or a population of such species (Thomas 1979).

habitat guild: a group of species having similar ecological requirements and/or foraging strategies and therefore having similar roles in the community (Cooperrider et al. 1986).

heading fire: a fire front spreading, or ignited to spread with the wind, up a slope, or influenced by a combination of wind and slope.

heat content: the net amount of heat that would be given off if fuel burns when it is absolutely dry, noted as Btu per pound of fuel.

heat per unit area: total amount of heat released per unit area as the flaming front of the fire passes, expressed as Btu/square foot; a measure of the total amount of heat released in flames.

heavy fuels: dead fuels of large diameter (3.0 inches or larger) such as logs and large branchwood.

height: the vertical measurement of vegetation from the top of the crown to ground level.

herbivorous: feeding on plants; phytophagous (Cooperrider et al. 1986).

humidity: see relative humidity.

hydrophobicity: resistance to wetting exhibited by some soils, also called water repellency. The phenomena may occur naturally or may be fire-induced. It may be determined by water drop penetration time, equilibrium liquid-contact angles, solid-air surface tension indices, or the characterization of dynamic wetting angles during infiltration.

- I -

ignition method: the means by which a prescribed fire is ignited, such as hand-held drip torch, heli- torch, and backpack propane tanks.

ignition pattern: the configuration and sequence in which a prescribed fire is ignited. Patterns include, for example, spot fire, strip-head fire, and ring fire (same as ignition technique).

ignition technique: see ignition pattern.

illuviation: soil development process by which materials are translocated from an upper soil horizon and immobilized in a soil horizon at a lower level in the soil profile.

imbibe: to absorb liquid or moisture.

Inceptisol: soil that is usually moist and has pedogenic horizons of alteration of parent material but not of illuviation. Generally, the direction of development is not evident, or is too weak to classify in another soil order. Inceptisols are approximately equal to Ando, Sol Brun Acide, some Brown Forest, Low-Humic Gley, and Humic Gley soils of the pre-1966 classification scheme.

increaser species: plant species of the original vegetation that increase in relative amount, at least for a time, under overuse. Commonly termed increasers.

independent crown fire: a fire that advances in the tree crowns alone, not requiring any energy from the surface fire to sustain combustion or movement.

intensity: see fireline intensity.

interspersion: the intermixing of plant species and plant communities that provide for animals in a defined area (Thomas 1979).

introduced plant species: a species not a part of the original fauna or flora of an area.

invader species: plant species that were absent in undisturbed portions of the original vegetation and will invade under disturbance or continued overuse. Commonly termed invaders.

- J -

juxtaposition: the arrangement of stands of vegetation in space (Thomas 1979).

- K -

Kcal: a kilogram-calorie is the amount of heat needed to raise the temperature of one kilogram of water (1 liter) by 1 degree Celsius.

key forage species: forage species of particular importance in the plant community or which are important because of their value as indicators of change in the community.

- L -

ladder fuels: fuels that can carry a fire from the surface fuel layer into the aerial fuel layer, such as a standing dead tree with branches that extend along its entire length.

leach: removal of soluble constituents from ashes or soil by percolation of water.

life-form: a group of wildlife species whose requirements for habitat are satisfied by similar successional stages within given plant communities (Thomas 1979).

lignotuber: a mass of woody tissue from which roots and stems originate, which often covered with dormant buds (James 1984); same as root crown.

litter: the top layer of forest floor, typically composed of loose debris such as branches, twigs, and recently fallen leaves or needles; little altered in structure by decomposition. The L layer of the forest floor (Deeming et al. 1977). *Also* loose accumulations of debris fallen from shrubs, or dead parts of grass plants laying on the surface of the ground.

live fuel moisture content: ratio of the amount of water to the amount of dry plant material in living plants.

live fuels: living plants, such as trees, grasses, and shrubs, in which the seasonal moisture content cycle is controlled largely by internal physiological mechanisms, rather than by external weather influences.

live herbaceous moisture content: ratio of the amount of water to the amount of dry plant material in herbaceous plants, i.e., grasses and forbs.

live woody moisture content: ratio of the amount of water to the amount of dry plant material in shrubs.

- M -

mast: the fruit of trees suitable as food for livestock and wildlife (Ford-Robertson 1971).

mean: see arithmetic mean.

meristem: growing points of grasses, from which leaf blade elongation occurs during active growing periods.

mesotopic: having an intermediate range of suitable ecological conditions.

mho: meter/kilogram/second unit of electrical conductance, equal to the conductance of a conductor in which a potential difference of one volt maintains a current of one ampere.

midflame windspeed: the speed of the wind measured at the midpoint of the flames, considered to be most representative of the speed of the wind that is affecting fire behavior.

millimho: a unit of electrical conductance, equal to 0.001 mho.

moisture content: see fuel moisture content.

moisture of extinction: the moisture content of a specific fuel type above which a fire will not propagate itself, and a firebrand will not ignite a spreading fire.

Mollisol: soil with nearly black, organic-rich surface horizons and high supplies of bases; they may accumulate large amounts of organic matter in the presence of calcium. They have mollic epipedons and base saturation greater than 50% in any cambic or argillic horizon and are approximately equal to Chestnut, Chernozem, Brunizem, Rendzina, some Brown, Brown Forest, and associated Solonetz and Groundwater Podzols of the pre-1966 classification scheme.

mosaic: the intermingling of plant communities and their successional stages in such a manner as to give the impression of an interwoven design (Ford-Robertson 1971).

muck: a highly decomposed layer of organic material in an organic soil (Buckman and Brady 1966).

mycorrhiza (pl. mycorrhizae): a mutually beneficial (symbiotic) association between a plant root and a fungus, that enhances the ability of the root to absorb water and nutrients.

- N -

National Fire Danger Rating System: a multiple index scheme designed to provide fire and land management personnel with a systematic means of assessing various aspects of fire danger on a day-to-day basis.

native species: a species which is a part of the original fauna or flora of the area in question.

natural fuels: fuels resulting from natural processes and not directly generated or altered by land management practices. (See activity fuels.)

NFDRS: see National Fire Danger Rating System.

niche: (habitat niche) the peculiar arrangement of food, cover, and water that meets the requirements of a particular species.

NIFMID: National Interagency Fire Management Integrated Database.

nonparametric tests: statistical testing techniques that are not dependent on a given distribution. (See parametric tests.)

normal distribution: a continuous frequency distribution whose graphic representation is a bell-shaped curve that is symmetrical about the mean, which by definition has a mean of 0 and a variance of 1. Many other types of distributions exist that have different shaped curves, such as hypergeometric, Poisson, and binomial.

number: the total population of a species or classification category in a delineated unit, a measure of its abundance.

nutrient: elements or compounds that are essential as raw materials for organism growth and develop ment, such as carbon, oxygen, nitrogen, and phosphorus. There are at least 17 essential nutrients.

- O -

obligate: restricted to a particular condition of life, as an obligate seeder, a plant that can only reproduce by seed.

off-site colonizers: plants that germinate and establish after a disturbance from seed that was carried from off of the site (Stickney 1986).

onsite colonizers: plants that germinate and establish after a disturbance from seed that was present on the site at the time of the disturbance (Stickney 1986).

one-hour timelag fuels: dead fuels consisting of dead herbaceous plant material and roundwood less than 0.25 inches (0.64 cm) in diameter, expected to reach 63 percent of equilibrium moisture content in one hour or less.

one-hundred hour timelag fuels: dead fuels consisting of roundwood in the size range from 1.0 to 3.0 inches (2.5 to 7.6 cm) in diameter, estimated to reach 63 percent of equilibrium moisture content in one hundred hours.

one-thousand hour timelag fuels: dead fuels consisting of roundwood 3.0 to 8.0 inches (7.6 to 20.3 cm) in diameter, estimated to reach 63 percent of equilibrium moisture content in one thousand hours.

organic matter: that fraction of the soil that includes plant and animal residues at various stages of decomposition, cells and tissues of soil organisms, and substances synthesized by the soil population.

organic soil: a soil with a percentage content of organic matter greater than about 20 to 25 percent (Buckman and Brady 1966).

- P -

packing ratio: the percentage of a fuel bed that is composed of fuel particles, the remainder being air space among the individual particles (Burgan and Rothermel 1984); the fuel volume divided by fuel bed volume.

palatability: the relish that an animal shows for a particular species, plant or plant part; how agreeable the plant is to the taste.

parameter: a variable which can be measured quantitatively; sometimes, an arbitrary constant; associated with populations. One of the unknown values that determine a model. (See statistic.)

parametric tests: statistical tests that are based on normal distributions. (See nonparametric tests).

particle size: the size of a piece of fuel, often expressed in terms of size classes.

particulate matter: any liquid or solid particles present in the atmosphere. Particulate matter diameter is measured in microns.

passive crown fire: a fire in the crowns of trees in which trees or groups of trees torch, ignited by the passing front of the fire. The torching trees reinforce the spread rate, but these fires are not basically different from surface fires.

peat: a deposit of slightly or non-decayed organic matter (Buckman and Brady 1966).

percolation: passage of liquid through a porous body, as movement of water through soil.

perennial plant: a plant that continues to grown year after year (Benson 1967). (See annual plant and biennial plant.)

permafrost: a short term for "permanently frozen ground"; any part of the earth's crust, bedrock, or soil mantle that remains below 32 F (0 C) continuously for a number of years (Brown 1970 in Viereck and Schandelmeier 1980). (See active layer.)

petroglyph: a type of rock art in which a design is pecked into stone.

pH: the negative logarithm (base =10) of the hydronium ion concentration, in moles per liter. It is a numerical measure of acidity or alkalinity on a scale of 1 to 14, with the value of 7.0 being neutral.

phenology: the relationship of the seasonal sequence of climatic factors with the timing of growth and reproductive phases in vegetation, such as initiation of seasonal growth, time of blooming, time of seed set, and development of new terminal buds (Daubenmire 1968b).

phreatophyte: a plant that derives its water from subsurfaces, typically having roots that reach the water table, and is therefore somewhat independent of precipitation. Obligate phreatophytes require this situation, whereas facultative phreatophytes merely take advantage it.

pictograph: a type of rock art in which a design is painted onto stone.

pile burning: burning of logging slash that has been arranged into individual piles. (See broadcast burning.)

PM_{10}: particles with an aerodynamic diameter smaller than or equal to a nominal ten micrometers.

$PM_{2.5}$: particles with an aerodynamic diameter smaller than or equal to a nominal 2.5 micrometers.

population: all possible values of a variable; the entire group that is examined. (See sample.)

precision: the closeness of repeated measurements of the same quantity. (See accuracy.)

preignition phase: preliminary phase of combustion in which fuel elements ahead of the fire are heated, causing fuels to dry. Heat induces decomposition of some components of the wood, causing release of combustible organic gases and vapors.

prescribed burning: controlled application of fire to wildland fuels in either their natural or modified state, under specified environmental conditions that allows the fire to be confined to a predetermined area, and produce the fire behavior and fire characteristics required to attain planned fire treatment and resource management objectives.

prescribed fire: an intentionally or naturally ignited fire that burns under specified conditions that allow the fire to be confined to a predetermined area and produce the fire behavior and fire characteristics required to attain planned fire treatment and resource management objectives.

prescription: a written statement defining the objectives to be attained as well as the conditions of temperature, humidity, wind direction and speed, fuel moisture, and soil moisture, under which a fire will be allowed to burn. A prescription is generally expressed as acceptable ranges of the prescription elements, and the limit of the geographic area to be covered.

prevention of significant deterioration: a provision of the Clean Air Act with the basic intent to limit degradation of air quality, particularly in those areas of the country where the air quality is much better than standards specified in the Law.

probability: a measurement that denotes the likelihood that an event occurred simply by chance.

probability of ignition: the chance that a firebrand will cause an ignition when it lands on receptive fuels.

productivity: weight of dry matter produced in a given period by all the green plants growing in a given space (Daubenmire 1968b).

PSD: see prevention of significant deterioration.

pyrolysis: the thermal or chemical decomposition of fuel at an elevated temperature. This is the pre- ignition phase of combustion during which heat energy is absorbed by the fuel that, in turn, gives off flammable tars, pitches, and gases.

- R -

ramet: an individual member of a clone. For example, every individual stem in an aspen clone is a ramet.

random: the assignment of treatments to experimental units, or the selection of samples, such that all units or samples have an equal chance of receiving the treatment being estimated. It serves to assure unbiased estimates of treatment means and experimental error.

rate of spread: see forward rate of spread.

RAWS: see Remote Automated Weather Station.

reaction intensity: the rate of heat release, per unit area of the fire front, expressed as heat energy/area/time, such as Btu/square foot/minute, or Kcal/square meter/second.

regeneration: see vegetative regeneration.

regional haze: atmospheric haze over a large area with no attributable source.

relative frequency: see frequency of occurrence.

relative humidity: the ratio, in percent, of the amount of moisture in a volume of air to the total amount which that volume can hold at the given temperature and atmospheric pressure. Relative humidity is a function of the actual moisture content of the air, the temperature, and the atmospheric pressure (Schroeder and Buck 1970).

Remote Automatic Weather Station (RAWS): a self contained meteorological platform that automatically acquires, processes, and stores local weather data for subsequent transmission through a satellite to an earth receiving station.

reproduction: see vegetative reproduction.

residence time: the time required for the flaming zone of the moving front of a fire to pass a stationary point; the total length of time that the flaming front of the fire occupies one point.

residual colonizers: plants that germinate after a disturbance from seed that was present on the site (Stickney 1986).

respiration: oxidation of food in living cells, with the resulting release of energy; part of the energy is transferred to other compounds and some is used in the activation of certain cell processes (Meyer et al. 1973).

rhizome: a horizontal plant stem, growing beneath the surface, and usually covered with dormant buds.

root crown: a mass of woody tissue from which roots and stems originate, and which are often covered with dormant buds (James 1984); same as lignotuber.

running crown fire: a fire moving in the crowns of trees, dependent upon, or independent from the surface fire.

- S -

sample: part of a population; that portion of the population that is measured.

sample size: the number of items or observations in a sample; usually denoted by lower case letter n.

SASEM: Simple Approach Smoke Estimation Model (SASEM), a computer model for the analysis of smoke dispersion from prescribed fires. It is a screening model, in that it uses simplified assumptions and tends to over predict impacts, yielding conservative results.

second order fire effects (SOFE): the indirect effects of fire treatment that occur over the longer term.

seedbank: the supply of viable seeds present on a site. Seeds include those recently dispersed by plants, long-lived seeds buried in organic and soil layers, or those stored in cones in a tree canopy.

semi-serotinous: cones of coniferous trees that open and release their seeds over a period of years (Zasada 1986).

senescence: period of declining productivity after the period of most active growth, referred to both in terms of the seasonal life cycle of a plant, and the total life of a perennial plant.

seral: pertaining to a succession of plant communities in a given habitat leading to a particular climax association; a stage in a community succession (Cooperrider et al. 1986).

sere: the stages that follow one another in an ecological succession (Hanson 1962).

serotiny: storage of coniferous seeds in closed cones in the canopy of the tree. Serotinous cones of lodgepole pine do not open until subjected to temperatures of 45 to 50 C (113 to 122 F), causing the melting of the resin bond that seals the cone scales.

severity: see burn severity.

short-life species: a plant that grows several years before being replaced by a species more adapted to the changing site conditions.

simulation: a realistic visual portrayal which demonstrates the perceivable changes in landscape features caused by a proposed management activity. This is done through the use of photography, art work, computer graphics and other such techniques.

sinter: clustering of clay particles that occurs when pottery is fired.

SIP: see State Implementation Plan.

slash: concentrations of wildland fuels resulting from human activities such as logging, thinning, and road construction, and natural events such as wind. Slash is composed of branches, bark, tops, cull logs, uprooted stumps, and broken or uprooted trees.

smoldering phase: a phase of combustion that can occur after flames die down because the reaction rate of the fire is not high enough to maintain a persistent flame envelope. During the smoldering phase, gases condense because of the cooler temperatures, and much more smoke is produced than during flaming combustion.

soil structure: the combination or arrangement of primary soil particles, units, or peds. Examples include platy, prismatic, columnar, blocky, angular blocky, subangular granular, and crumb.

soil texture: the relative proportions of the various soil separates, primarily sand, clay, and silt. Subdivisions of the three basic separates, such as very fine sand, are often used.

spall: disintegration of a rock by breaking away of an outer layer.

species composition: a term relating the relative abundance of one plant species to another using a common measurement; the proportion (percentage) of various species in relation to the total on a given area.

species richness: a measurement or expression of the number of species of plants or animals present in an area; the more species present, the higher the degree of species richness (Thomas 1979).

spot fire: fire caused by flying sparks or embers outside the perimeter of the main fire.

spot forecast: a customized prediction of atmospheric conditions at a specific site that is issued by the National Weather Service, usually requested in connection with a wildfire incident or a prescribed fire.

spot weather forecast: see spot forecast.

spotting: production of burning embers in the moving fire front that are carried a short distance ahead of the fire, or in some cases are lofted by convective action or carried by fire whirls some distance ahead.

standard deviation: a measure of the variation, or spread, of individual measurements; a measurement which indicates how far away from the middle the statistics are; usually denoted by the lower case s for sample data; mathematically equal to the square root of variance.

State Implementation Plan: a plan that describes how a State intends to achieve Federal and State standards relative to the Clean Air Act, usually containing State regulations related to maintenance of air quality.

statistic: the number that results from manipulating raw data according to a specified procedure; associated with samples. (See parameter.)

statistics: the scientific study of numerical data based on natural phenomena.

stenotopic: having a narrow range of suitable ecological conditions (Pennak 1964).

stochastic: of, or pertaining to, randomness.

stolon: a branch of a plant which grows along the surface of the ground and produces plants and roots at intervals.

structure (vegetative): the form or appearance of a stand; the arrangement of the canopy; the volume of vegetation in tiers or layers (Thomas 1979).

succession: the process of vegetational development whereby an area becomes successively occupied by different plant communities of higher ecological order (Range Term Glossary Committee 1974).

successional change: see succession.

sum: the amount obtained by adding numbers or quantities; total; usually denoted by an upper case Greek sigma, .

surface area to volume ratio: the ratio between the surface area of an object, such as a fuel particle, to its volume. The smaller the particle, the more quickly it can become wet, dry out, or become heated to combustion temperature during a fire.

surface fire: fire that burns surface litter, dead woody fuels, other loose debris on the forest floor, and some small vegetation.

surface fuels: fuels that contact the surface of the ground, consisting of leaf and needle litter, dead branch material, downed logs, bark, tree cones, and low stature living plants.

survivors: plant species with established plants on the site that can vegetatively regenerate after the fire (Stickney 1986).

- T -

TAC: total available carbohydrates.

ten-hour timelag fuels: dead fuels consisting of roundwood 0.25 to 1.0 inches (0.6 to 2.5 cm) in diameter, estimated to reach 63 percent of equilibrium moisture content in ten hours.

thermoluminescence: a property of fired materials, such as ceramics, causing them to become luminous when gently heated again. Because this property decays at a known rate, the age of a ceramic artifact can be estimated by heating it and measuring the amount of phosphorescence.

tiller: new growth in a graminoid that originates from dormant axillary buds in the plant crown or on rhizomes (Dahl and Hyder 1977).

tillering: process of producing new grass growth from dormant axillary buds in the plant crown or on rhizomes (Dahl and Hyder 1977).

timelag: the time necessary for a fuel particle to lose or gain approximately 63 percent of the difference between its initial moisture content and its equilibrium moisture content.

torch: ignition and subsequent envelopment in flames, usually from bottom to top, of a tree or small group of trees.

total available carbohydrates (TAC): carbohydrates that are in a form that can be utilized as a readily available source of energy by a plant, including sugars, starch, dextrins, and fructosans (Smith et al. 1964 in Trlica 1977).

total fuel: all plant material, both living and dead, on a site.

trachea: in air breathing vertebrates, the tube that serves as the principal passage for conveying air to the lungs.

treatment: a procedure whose effect can be measured and compared with the effect of other procedures. Examples include a fall burned prescribed fire, an unburned "control", or an area burned with a specific ignition method or pattern.

- U -

underburning: prescribed burning in activity-created or natural fuels beneath a forest canopy, usually with the objective of preserving the dominant overstory trees.

utilization rates (limits): the proportion of the current year's forage production that is removed by grazing or browsing animals. It may refer to particular species or to the entire plant community and is usually expressed as a percentage.

- V -

vapor pressure: the contribution to total atmospheric pressure due to the presence of water molecules in the air (Schroeder and Buck 1970).

variable: any changing characteristic; in statistics, a measurable characteristic of an experimental unit.

variance: the sum of the squares of the deviates divided by one less than the total number of deviates; a measure which indicates how far away from the middle the statistics are; usually denoted by the lower case s^2. Variance is the standard deviation squared. In practice, it is easier to compute the variance, then take the square root to obtain the standard deviation. (See standard deviation.)

vegetative regeneration: development of new aboveground plants from surviving plant parts, such as by sprouting from a root crown or rhizomes. Even if plants form their own root system, they are still genetically the same as the parent plant (Zasada 1989).

vegetative reproduction: establishment of a new plant from a seed that is a genetically distinct individual (Zasada 1989).

vigor: a subjective assessment of the health of individual plants in similar site and growing conditions; or a more specific measure based upon a specific facet of growth, such as seed stalk or tiller production per plant or per unit area.

- W -

weight: as used in vegetation inventory and monitoring, the total biomass of living plants growing above the ground in a given area at a given time.

wildfire: a free burning and unwanted wildland fire requiring a suppression action.

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[1] The two publications by Frank Freese are no longer printed by the government, but have been reprinted under one cover by O.S.U. Bookstores, Inc., Oregon State Univ., Corvallis, 97331.

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IX. Prefire and Postfire Grazing Management - Ken Stinson

X. Documentation and Evaluation - Ken Stinson and Melanie Miller

XI. Data Analysis - Dr. Bob Clark

XII. Computer Software - Melanie Miller

EDITOR

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Wildland Fire in Ecosystems

Effects of Fire on Fauna

Abstract

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Fires affect animals mainly through effects on their habitat. Fires often cause short-term increases in wildlife foods that contribute to increases in populations of some animals. These increases are moderated by the animals' ability to thrive in the altered, often simplified, structure of the postfire environment. The extent of fire effects on animal communities generally depends on the extent of change in habitat structure and species composition caused by fire. Stand-replacement fires usually cause greater changes in the faunal communities of forests than in those of grasslands. Within forests, stand-replacement fires usually alter the animal community more dramatically than understory fires. Animal species are adapted to survive the pattern of fire frequency, season, size, severity, and uniformity that characterized their habitat in presettlement times. When fire frequency increases or decreases substantially or fire severity changes from presettlement patterns, habitat for many animal species declines.

Keywords: fire effects, fire management, fire regime, habitat, succession, wildlife

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Wildland Fire in Ecosystems

Effects of Fire on Fauna

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Cover photo—Male black-backed woodpecker on fire-killed lodgepole pine. Photo by Milo Burcham.

Preface

In 1978, a national workshop on fire effects in Denver, Colorado, provided the impetus for the "Effects of Wildland Fire on Ecosystems" series. Recognizing that knowledge of fire was needed for land management planning, state-of-the-knowledge reviews were produced that became known as the "Rainbow Series." The series consisted of six publications, each with a different colored cover, describing the effects of fire on soil, water, air, flora, fauna, and fuels.

The Rainbow Series proved popular in providing fire effects information for professionals, students, and others. Printed supplies eventually ran out, but knowledge of fire effects continued to grow. To meet the continuing demand for summaries of fire effects knowledge, the interagency National Wildfire Coordinating Group asked Forest Service research leaders to update and revise the series. To fulfill this request, a meeting for organizing the revision was held January 4-6, 1993, in Scottsdale, Arizona. The series name was then changed to "The Rainbow Series." The five-volume series covers air, soil and water, fauna, flora and fuels, and cultural resources.

The Rainbow Series emphasizes principles and processes rather than serving as a summary of all that is known. The five volumes, taken together, provide a wealth of information and examples to advance understanding of basic concepts regarding fire effects in the United States and Canada. As conceptual background, they provide technical support to fire and resource managers for carrying out interdisciplinary planning, which is essential to managing wildlands in an ecosystem context. Planners and managers will find the series helpful in many aspects of ecosystem-based management, but they will also need to seek out and synthesize more detailed information to resolve specific management questions.

— The Authors
January 2000

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Contents

Summary

Fire regimes—that is, patterns of fire occurrence, size, uniformity, and severity—have been a major force shaping landscape patterns and influencing productivity throughout North America for thousands of years. Faunal communities have evolved in the context of particular fire regimes and show patterns of response to fire itself and to the changes in vegetation composition and structure that follow fire.

Animals' immediate responses to fire are influenced by fire season, intensity, severity, rate of spread, uniformity, and size. Responses may include injury, mortality, immigration, or emigration. Animals with limited mobility, such as young, are more vulnerable to injury and mortality than mature animals.

The habitat changes caused by fire influence faunal populations and communities much more profoundly than fire itself. Fires often cause a short-term increase in productivity, availability, or nutrient content of forage and browse. These changes can contribute to substantial increases in herbivore populations, but potential increases are moderated by animals' ability to thrive in the altered, often simplified, structure of the postfire environment. Fires generally favor raptors by reducing hiding cover and exposing prey. Small carnivores respond to fire effects on small mammal populations (either positive or negative). Large carnivores and omnivores are opportunistic species with large home ranges. Their populations change little in response to fire, but they tend to thrive in areas where their preferred prey is most plentiful—often in recent burns. In forests and woodlands, understory fires generally alter habitat structure less than mixed-severity and stand-replacement fires, and their effects on animal populations are correspondingly less dramatic. Stand-replacing fires reduce habitat quality for species that require dense cover and improve it for species that prefer open sites. Population explosions of wood-boring insects, an important food source for insect predators and insect-eating birds, can be associated with fire-killed trees. Woodpecker populations generally increase after mixed-severity and stand-replacement fire if snags are available for nesting. Secondary cavity nesters, both birds and mammals, take advantage of the nest sites prepared by primary excavators.

Many animal-fire studies depict a reorganization of animal communities in response to fire, with increases in some species accompanied by decreases in others. Like fire effects on populations, fire effects on communities are related to the amount of structural change in vegetation. For example, understory fires and stand-replacement fires in grasslands often disrupt bird community composition and abundance patterns for only 1 to 2 years, but stand-replacement fires in shrublands and forests cause longer lasting effects, which are initially positive for insect- and seed-eating species and negative for species that require dense, closed canopy. Bird abundance and diversity are likely to be greatest early in succession. When shrub or tree canopy closure occurs, species that prefer open sites and habitat edges decline and species that prefer mature structures increase.

Major changes to fire regimes alter landscape patterns, processes, and functional linkages. These changes can affect animal habitat and often produce major changes in the composition of faunal communities. In many Western ecosystems, landscape changes due to fire exclusion have changed fuel quantities and arrangement, increasing the likelihood of large or severe fires, or both. Where fire exclusion has changed species composition and fuel arrays over large areas, subsequent fires without prior fuel modification are unlikely to restore presettlement vegetation and habitat. In many desert and semidesert habitats where fire historically burned infrequently because of sparse fuels, invasion of weedy species has changed the vegetation so that burns occur much more frequently. Many animals in these ecosystems are poorly adapted to avoid fire or use resources in postfire communities.

In the past 10,000 years, fire in North American ecosystems has not operated in isolation from other disturbances, nor has it occurred independent of human influence. In many areas, however, fire has been prevented or excluded for nearly 100 years, a level of success that is not likely to continue. Collaboration among managers, researchers, and the public is needed to address tradeoffs in fire management, and fire management must be better integrated with overall land management objectives to address the potential interactions of fire with other disturbances such as grazing, flood, windthrow, and insect and fungus infestations.

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Chapter 1: Introduction

Effects of wildland fire on fauna show almost infinite variety. Previous authors have limited discussion of this subject to only a few vertebrate groups (Bendell 1974), specific biotic provinces (Fox 1983; Stanton 1975), or general summaries (Lyon and others 1978). This report surveys the principles and processes governing relationships between fire and fauna. We recognize that this approach has limitations. We focus almost entirely on vertebrates, particularly terrestrial mammals and birds, because the information on those groups is most complete and the principles best documented. (Fire effects on aquatic vertebrates are summarized in "Effects of Fire on Soil and Water," another volume in this Rainbow Series.) We describe fire effects on specific faunal populations and communities by way of example, not as a survey of all that is known. Those seeking a detailed description of fire effects on fauna are referred to books that discuss the subject in general, such as Whelan (1995, chapter 6) and Wright and Bailey (1982, chapter 4); reports about fire effects in specific geographic regions (for example, McMahon and deCalesta 1990; Viereck and Dyrness 1979); and summaries of fire effects on specific faunal groups (for example, Crowner and Barrett 1979; Lehman and Allendorf 1989; Russell 1999). The Fire Effects Information System, on the Internet at www.fs.fed.us/database/feis, provides detailed descriptions of fire effects on more than 100 North American animal species and nearly 1,000 plant species.

Fires affect fauna mainly in the ways they affect habitat. Repeated fires have been a major force shaping landscapes and determining productivity throughout North America for thousands of years, with the possible exception of some portions of West Coast rain forests. Climate, vegetation, Native Americans, and fire interacted in a relatively consistent manner within each biotic region of North America before the advent of disease and settlers from Europe (Kay 1998). Prior to modern agriculture, fire suppression, and urbanization, vegetation patterns in each region were shaped by fire regimes with characteristic severity, size, and return interval (Frost 1998; Gill 1998; Heinselman 1981; Kilgore 1981).

The animal species native to areas with a centuries-long history of fire can obviously persist in habitat shaped by fire; many species actually thrive because of fire's influence. How? Animals' immediate response to fire may include mortality or movement. It is influenced by fire intensity, severity, rate of spread, uniformity, and size. Long-term faunal response to fire is determined by habitat change, which influences feeding, movement, reproduction, and availability of shelter (fig. 1). Alteration of fire regimes alters landscape patterns and the trajectory of change on the landscape; these changes affect habitat and often produce major changes in faunal communities.

Figure 1—Elk rest and graze in unburned meadow adjacent to area burned by crown fire, Yellowstone National Park. Fire and fauna have coexisted in this ecosystem for thousands of years. Photo by Rick McIntyre, courtesy of National Park Service.

This volume is organized on the premises that fire regimes strongly influence animal response to fire and that fire affects animals at every level of ecosystem organization. In this chapter, we describe the fundamental concepts of fire regimes and their effects on vegetation structure. Chapter 2 describes the role of fire in several North American vegetation communities prior to settlement by European Americans. Because the vegetation provides habitat for fauna, this chapter provides background for understanding examples used in later chapters. The next four chapters describe animal response to fire at four levels of organization: individual, population, community, and landscape. Chapter 7 surveys fire effects on wildlife foods. Finally, chapter 8 discusses management implications of fire-fauna relationships, particularly in light of past fire exclusion, and identifies information gaps and research needs. Scientific names of all animals described in this report are listed in appendix A. Appendix B lists scientific names of plants. Appendix C contains a glossary of technical terms.

Historic Perspective

Fire has influenced composition, structure, and landscape patterns of animal habitat for millennia, so it is reasonable to assume that animals have coexisted and adapted to periodic perturbations from fire. Records show that lightning starts more than 6,000 fires each year in the United States; surely this force was just as powerful and ubiquitous in past millennia as it is now (Pyne 1982). Prior to the 1500s, millions of Native Americans lived in North America. They used fire regularly for many purposes (Kay 1998). Only recently, since the advent of fire exclusion policy and other activities that strongly influence fire regimes, has fire's influence on fauna been intensely questioned and investigated (Kilgore 1976).

During the era of European settlement of North America, fire came to be viewed in some geographic areas as a hopelessly destructive event that could not be stopped. Early legislation promoting fire control responded primarily to the loss of lives and settlements and vast mortality of harvestable trees that occurred in large fires around the end of the 18th century (for example, the Peshtigo Fire of 1871). However, proclaimed reductions of elk, deer, bison, and other wildlife populations associated with these fire events were also important considerations in establishing fire control legislation (Brown and Davis 1973).

Resource protection and associated fire control began with the establishment of forest reserves throughout North America at the end of the 1900s. The reserves were established mainly to protect the land from abuse by timber and grazing interests, but early reports from the reserves singled out fire as the greatest threat to America's grasslands, forests, and wildlife (Komarek 1962). The primary fire-related mission of land management agencies was reinforced in the 1930s: to stop fires wherever possible, and to prevent large fires from developing (Moore 1974). From that time until the 1960s, most managers and many of the public viewed fire as an unnatural event and an environmental disaster. The land area under vigorous fire protection grew every year, and the resources assigned to fire suppression grew accordingly (Brown and Davis 1973).

Even in the early days of fire exclusion policy, there were dissenting voices. In 1912, ecologist H. H. Chapman recognized that longleaf pine in the South was adapted to grow and mature in the presence of repeated fires (Chapman 1912). Subsequent studies by other researchers found that controlled burning improved quality of ungulate forage (Green 1931) and improved, restored, and maintained habitat for certain game species, especially the northern bobwhite (Stoddard 1931, 1935, 1936). The scientific community was beginning to view fire as a natural process and a tool for wildlife habitat management, but many public and private land managers strongly resisted the concept (Schiff 1962).

During World War II, fire suppression capability declined. The disastrous 1943 drought-related fires in the Southeastern United States prompted major shifts in government policies (Schiff 1962). By the 1950s, controlled burning to reduce fuels and enhance habitat for specific wildlife species had become commonplace, but all other fires were vigorously controlled. Meanwhile, scientists began to report striking changes in plant community composition and structure associated with fire exclusion. Important functions of fire were described for ponderosa pine in the Pacific Northwest (Weaver 1943), California chaparral and ponderosa pine (Biswell 1963), Arizona ponderosa pine (Cooper 1960), Florida Everglades (Loveless 1959), and interior Alaska (Lutz 1956).

With the publication of the Leopold Report (Leopold and others 1963) on ecological conditions of National Parks in the United States, managers and the public began to see the benefits of fires in wildlands. The Leopold Report established the concept that wildlife habitat is not a stable entity that persists unchanged in perpetuity, but rather a dynamic entity; suitable habitat for many wildlife species and communities must be renewed by fire. Policy began to shift away from the assumption that all wildland fires are destructive (Pyne 1982). In 1968, the fire policy of the USDI National Park Service changed drastically as managers began to adopt the recommendations of the Leopold Report. Policy officially recognized fire as a natural process to be managed for maintaining ecosystems and improving wildlife habitat. Thus began the current era of fire management in which fire is recognized as an integral part of ecosystems, including those aspects relating to fauna (Habeck and Mutch 1973).

Fire Regimes

Knowledge of the ecological role of fire in past centuries and descriptions of significant changes in the role of fire over time are essential for communication among professionals and citizens interested in resource management. Nearly every North American ecosystem has been drastically changed from conditions of past millennia. Regardless of how fire might be managed in the future in various ecosystems, information about its past role is important. As Morgan and others (1994) said, "Study of past ecosystem behavior can provide the framework for understanding the structure and behavior of contemporary ecosystems, and is the basis for predicting future conditions."

Fire varies in its frequency, season, size, and prominent, immediate effects, but general patterns occur over long periods. These patterns describe *fire regimes*. The practice of organizing biotic information around fire regimes originated in North America around 1980 (Heinselman 1978, 1981; Kilgore 1981; Sando 1978). Descriptions of fire regimes are general because of fire's tremendous variability over time and space (Whelan 1995). Nevertheless, the fire regime is a useful concept because it brings a degree of order to a complicated body of knowledge. The fire regimes that have influenced North American ecosystems in an evolutionary sense are those of pre-Columbian times (prior to 1500), before diseases introduced by European explorers began to decimate populations of Native Americans (see Kay 1995). While knowledge of pre-Columbian fire regimes would be useful for understanding ecosystem patterns and processes today, little information is available from that era. Detailed information available about past fire regimes is mostly based on biophysical evidence, written records, and oral reports that encompass the time from about 1500 to the mid- to late-1800s. This was a time before extensive settlement by European Americans in most parts of North America, before extensive conversion of wildlands for agricultural and other purposes, and before fire suppression effectively reduced fire frequency in many areas. In this volume, we refer to the fire regimes of the past several centuries as "presettlement" fire regimes.

Fire frequency and severity form the basis for the commonly referenced fire regime classifications described by Heinselman (1978) and Kilgore (1981). Two concepts, fire return interval and fire rotation, describe the frequency with which fires occur on a landscape. *Mean fire return interval* is the average number of years between fires at a given location. *Fire rotation*, called by some authors the *fire cycle*, is the number of years that would be required to completely burn over a given area.

Fire severity describes the immediate effects of fire, which result from the rate of heat release in the fire's flaming front and the total heat released during burning. Fire severity determines in large part the mortality of dominant vegetation and changes in the aboveground structure of the plant community, so Kilgore (1981) refers to severe fires in forests as "stand-replacement" fires. The concept of stand replacement by fire applies to nonforest as well as forest areas. Fires in vegetation types such as prairie, tundra, and savannah are essentially all stand-replacing because the aboveground parts of dominant vegetation are killed (and often consumed) by fire. Most shrubland ecosystems also have stand-replacement fire regimes because fire usually kills the aboveground parts of shrubs. In this report, we refer to the following four kinds of fire regime:

1. *Understory fire regime* (applies to forest and woodland vegetation types)—Fires are generally not lethal to the dominant vegetation and do not substantially change the structure of the dominant vegetation. Approximately 80 percent or more of the aboveground dominant vegetation survives fires.
2. *Stand-replacement regime* (applies to forests, shrublands, and grasslands)—Fires kill or top-kill aboveground parts of the dominant vegetation, changing the aboveground structure substantially. Approximately 80 percent or more of the aboveground dominant vegetation is either consumed or killed as a result of fires.
3. *Mixed-severity regime* (applies to forests and woodlands)—Severity of fire either causes selective mortality in dominant vegetation, depending on different species' susceptibility to fire, or varies between understory and stand-replacement.
4. *Nonfire regime*—Little or no occurrence of natural fire (not discussed further in this volume).

See "Effects of Fire on Flora" (also in the Rainbow Series) for further discussion of fire regimes and comparison of this fire regime classification with others.

The literature demonstrates great local variation in fire effects on habitat, even within small geographic areas with a single fire regime. Fires theoretically should spread in an elliptical pattern (Anderson 1983; Van Wagner 1969), but the shape of burned areas and the fire severity patterns within them are influenced by fluctuations in weather during fires, diurnal changes in burning conditions, and variation in topography, fuels, and stand structure. Variable and broken topography and sparse fuels are likely to produce patchy burns, while landscapes with little relief and homogeneous fuels may burn more uniformly. It is no wonder then that fires shape a complex mosaic of size classes, vegetation structure, and plant species occurrence across the landscape, and this variety has a profound influence on the animals that live there.

Changes in Vegetation Structure

For animals, the vegetation structure spatially arranges the resources needed to live and reproduce, including food, shelter and hiding cover. Some fires alter the vegetation structure in relatively subtle ways, for example, reducing litter and dead herbs in variable-sized patches. Other fires change nearly every aspect of vegetation structure: woody plants may be stripped of foliage and killed; litter and duff may be consumed, exposing mineral soil; underground structures, such as roots and rhizomes, may be killed (for example, in most coniferous trees) or rejuvenated (for example, in many grass and shrub species, aspen, and oak). In this section, we summarize postfire structural changes according to the fire regimes described above.

Understory Fire Regimes

Understory fires change the canopy in two ways: by killing or top-killing a few of the most fire-susceptible trees, and by killing or top-killing a cohort of tree regeneration, also selectively according to fire resistance. Understory fires also reduce understory plant biomass, sometimes in a patchy pattern. Although the structural changes caused by any one understory fire are not dramatic, repeated understory fires shape and maintain a unique forest structure identified by O'Hara and others (1996) as "old forest, single stratum." It is characterized by large, old trees, parklike conditions, and few understory trees (fig. 2).

Figure 2—Mature longleaf pine forest, typical of forest structure maintained by frequent understory fire, in Francis Marion National Forest, South Carolina. This kind of habitat favors many fauna species, included red-cockaded woodpecker, Bachman's sparrow, northern bobwhite, fox squirrel, and flatwoods salamander. Photo by Robert G. Hooper.

Stand-Replacement Fire Regimes

Grasslands—In grasslands, the prefire structure of the vegetation reasserts itself quickly as a new stand of grass springs up from surviving root systems. Standing dead stems and litter are reduced. The proportion of forbs usually increases in the first or second postfire year. In about 3 years the grassland structure is usually reestablished (Bock and Bock 1990), and faunal populations are likely to resemble those of the preburn community. Repeated fires can convert some shrublands to grass, and fire exclusion converts some grasslands to shrubland and forest.

Shrublands—In shrub-dominated areas, including sagebrush, chaparral, and some oak woodlands, stand-replacing fires top-kill or kill aboveground vegetation. Canopy cover is severely reduced, but initial regrowth usually increases cover of grasses and forbs. Dead woody stems often remain standing and serve as perch sites for songbirds, raptors, and even lizards (fig. 3). Burning increases seed visibility and availability for small mammals but also increases their visibility to predators. Because cover for ungulates is reduced by fire, some species do not use the abundant postfire forage. Shrubs regenerate from underground parts and seed. The length of time required to reestablish the shrubland structure varies, from 2 years in saw palmetto scrub (Hilmon and Hughes 1965) to more than 50 years in big sagebrush (Wright 1986).

Forests and Woodlands—In tree-dominated areas, stand-replacing fires change habitat structure dramatically. When crown fire or severe surface fire kills most of the trees in a stand, surface vegetation is consumed over much of the area, and cover for animals that use the tree canopy is reduced. Crown fires eliminate most cover immediately; severe surface fires kill the tree foliage, which falls within a few months. Stand-replacing fires alter resources for herbivores and their predators. The habitat is not "destroyed," but transformed: The fire-killed trees become food for millions of insect larvae and provide perches for raptors. Trees infected by decay before the fire provide nest sites for woodpeckers and then for secondary cavity nesters (birds and mammals). As

Figure 3—Sagebrush 3 years after stand-replacing fire, east-central Idaho. Fire kills sagebrush but leaves dead stems that birds and reptiles use as perches. The photo shows early successional dominance by dense bluebunch wheatgrass. Photo by Loren Anderson.

these snags fall, other fire-killed trees decay and provide habitat for cavity nesters. For 10 to 20 years after stand-replacing fire, biomass is concentrated on the forest floor, as grasses and forbs, shrubs and tree saplings reoccupy the site. These provide forage and dense cover for small mammals, nest sites for shrubland birds, and a concentrated food source for grazing and browsing ungulates. In 30 to 50 years after stand-replacing fire, saplings become trees and suppress the early successional shrub and herb layers. The forest again provides hiding and thermal cover for ungulates and nesting habitat for animals that use the forest interior. The remaining fire-killed snags decay and fall, reducing nest sites for cavity-nesting birds and mammals but providing large pieces of dead wood on the ground. This fallen wood serves as cover for small mammals, salamanders, and ground-nesting birds. The fungi and invertebrates living in dead wood provide food for birds and small mammals (for example, see McCoy and Kaiser 1990).

In some northern and western coniferous forests, the initial postfire stand is composed of broad-leafed, deciduous trees such as aspen or birch. Conifer dominance follows later in succession. Some bird and mammal species prefer the broad-leafed successional stage to earlier and later stages of succession. As succession continues, conifers dominate and broad-leafed trees decay. This process creates snags and adds to dead wood on the ground, enhancing habitat for cavity nesters and small mammals. It also creates openings that are invaded by shrubs and saplings. Dense patches of shrubs and tree regeneration in long-unburned forests provide excellent cover for ungulates. Birds (for example, crossbills, nuthatches, brown creeper, and woodpeckers), tree squirrels, and American marten find food, cover, and nest sites within the structure of the old-growth coniferous forest.

In some Southeastern forests, the roles of pine and hardwood tree species are reversed. Many Southeastern forests regenerate to pine immediately after stand-replacing fire. In the absence of repeated understory fires, these pine stands are invaded and eventually dominated by broad-leafed deciduous species such as American beech, hickory, and southern magnolia (Engstrom and others 1984; Komarek 1968). As in the hardwood-conifer sere of the Western States, each structural stage supports a somewhat different assemblage of wildlife.

Mixed-Severity Fire Regimes

In mixed-severity fire regimes, fires either cause selective mortality of fire-susceptible species in the overstory or alternate between understory and stand-replacement, with overlapping burn boundaries. The net result is a fine grained pattern of stand ages and structures across the landscape. This pattern is accentuated in areas where variable topography and microclimate influence fire spread. Through feedback of fuel patterns into subsequent fire behavior, the variety in fuels and stand structures resulting from mixed-severity fire perpetuate the complex mosaic of ages and structures.

Snags and Dead Wood

It would be difficult to overestimate the importance of large trees, snags, and dead, down wood to North American birds and small mammals. According to Brown and Bright (1997), "The snag represents perhaps the most valuable category of tree-form diversity in the forest landscape." Fire and snags have a complex relationship. Fires convert live trees to snags, but fires also burn into the heartwood of old, decayed snags and cause them to fall. Fire may facilitate decay in surviving trees by providing an entry point for fungi, which increases the likelihood that the trees will be used by cavity excavators. Fire may harden the wood of trees killed during a burn, causing their outer wood to decay more slowly than that of trees that die from other causes. This "case-hardening" process reduces the immediate availability of fire-killed snags for nest excavation but slows their decay after they fall.

It is difficult to identify fire-injured trees that are likely to become snags, and it is also difficult to determine which snags may have the greatest "longevity," that is, may stand the longest time before falling. In ponderosa pine stands in Colorado, for example, the trees most likely to become long-lasting snags are underburned trees with moderate crown scorch that remain alive for at least 2 years after fire, a group that cannot be determined until 2 or 3 years after fire (Harrington 1996). According to Smith (1999), longevity of ponderosa pine snags is positively related to tree age and size at death. Fire-scarred trees may have greater longevity than trees never underburned (Harrington 1996).

The usefulness of snags to fauna is enhanced or reduced by the surrounding habitat, since cavity nesters vary in their needs for cover and food. Many cavity excavators require broken-topped snags because partial decay makes them easier to excavate than sound wood (Caton 1996). Some bird species nest only in large, old snags, which are likely to stand longer than small snags (Smith 1999). Pileated woodpeckers are an example. Some excavators and secondary cavity nesters prefer clumps of snags to individual snags, so the spatial arrangement of dead and decaying trees in an area influences their usefulness to wildlife (Saab and Dudley 1998).

Dead wood on the ground is an essential habitat component for many birds, small mammals (fig. 4), and even large mammals, including bears (Bull and Blumton 1999). Large dead logs harbor many invertebrates and are particularly productive of ants; they also provide shelter and cover for small mammals, amphibians, and reptiles. Fire both destroys and creates woody debris. While large, down logs are not always abundant in early postfire years, fire-killed trees eventually fall and become woody debris. Down wood from fire-killed trees often decays more slowly than wood of trees killed by other means (Graham and others 1994).

Figure 4—Vagrant shrew travelling in shelter of dead log, Lolo National Forest, western Montana. Large dead wood is an essential source of food and shelter for many small mammals. Photo by Kerry R. Foresman.

Notes

Edmund S. Telfer

Chapter 2: Regional Variation in Fire Regimes

To provide a context for discussion of fire effects on animals and their habitat, this chapter describes the vegetation, fire regimes, and postfire succession of several plant communities referred to in subsequent sections of this report. This description is not meant to be a complete survey of fire regimes in North America; such a survey is available in "Effects of Fire on Flora," also part of the Rainbow Series. Instead, it provides examples of plant communities and fire regimes throughout the continent, many of them described in earlier reviews, including Wright and Bailey (1982). These communities are divided according to the geographic regions used to describe fire effects on the flora in this series: northern ecosystems; eastern ecosystems, including the Great Plains; western forests; western woodlands, shrublands, and grasslands; and subtropical ecosystems.

Northern Ecosystems

Boreal Forest

Vegetation and Fire Regimes—The Boreal Forest was characterized in presettlement times by stand-replacement fire regimes, although understory fires were common in some dry forest types (Heinselman 1981; Johnson 1992). Most of the presettlement fire rotations reported in the literature were relatively short, ranging from 50 to 100 years (Heinselman 1981; Payette and others 1989; Wein 1993). Johnson (1992) determined that the fire rotation was between 40 and 60 years for Minnesota, Ontario, the Northwest Territories, and Alaska. Relatively short rotations probably occurred in dry continental interior regions; for example, a 39-year rotation in northern Alberta (Murphy 1985). Longer rotations occurred in floodplains (Heinselman 1981) and Eastern boreal forests (Viereck 1983). From 1980 to 1989, frequency of fires larger than about 500 acres (200 ha) in the Canadian boreal forest was greatest in the central part of the continent and decreased toward the east and northwest. Fires were more frequent during dry climatic periods than during wet periods (Clark 1988; Swain 1973).

Due to frequent fires in the Boreal Forest, there probably has been no time during the last 6,000 to 10,000 years when ancient or even old forest covered a high proportion of the region (Telfer 1993). Van Wagner (1978) and Johnson (1992) found that the distribution of forest area over age classes often approximates a negative exponential distribution, permitting prediction of the distribution of age class areas under various fire rotations. Based on this relationship, Johnson (1992) commented that a 40- to 60-year fire rotation, "…by definition, suggests that most (63 percent) stands will never live much beyond the age at which stand canopy closure occurs and very few will reach anything resembling old age."

Postfire Succession—Principal Boreal Forest trees include black spruce, white spruce, jack pine, and quaking aspen. All of these species regenerate well on burned sites. Most of the understory plants that occur in the Boreal Forest sprout from underground parts that can survive fire. Ahlgren (1974) does not consider any boreal shrub species likely to suffer substantial mortality due to burning.

Croskery and Lee (1981) examined plant regrowth at burned and unburned sites on a large May and June stand-replacing fire in northwestern Ontario. Existing trees, mostly black spruce and jack pine, were killed by the fire, and aboveground parts of shrubs and ground cover were mostly consumed. However, regrowth began immediately. By mid-July, ground cover in the burned area had rebounded to 50 percent of that in the unburned area, with an average of 14 species present compared to 21 on unburned sites. In the second growing season after fire, shrubs began to appear on the burn. By the fifth growing season, ground cover was 40 percent and mean height of deciduous species was 5 feet (1.5 m). Browse biomass was eliminated on severely burned areas for 2 years, then became available in small amounts. By the fifth year, browse was abundant.

Laurentian Forest

Vegetation and Fire Regimes—The Laurentian Forest constitutes a broad ecotone between the Eastern Deciduous Forest and Boreal Forest. It contains plant and animal species characteristic of both regions and some species, like eastern white pine, red pine, and red spruce, whose distributions are centered here. The forest consists of extensive pine forest and stands of northern hardwoods intermixed with eastern hemlock. Studies of charcoal and plant pollen in lake sediments show that fire has influenced species composition of the vegetation in the eastern portion of the Laurentian Forest during much of the past 10,000 years (Green 1986).

Overall, the most common kinds of fire in the Laurentian Forest were stand-replacement and mixed-severity fire, although understory fires occurred as well (Heinselman 1981). Stand-replacing fire predominated in jack pine, black spruce, and spruce-fir forests, with fire rotations in the 50- to 100-year range (Heinselman 1981). In red and white pine forests, mixed-severity fires predominated. Presettlement fire rotations in some coniferous forests were 150 to 300 years (Wein and Moore 1977). In Northern hardwood forests, fire rotations may sometimes have exceeded 1,000 years. The proportion of early successional stand area was small at any given time (Telfer 1993). Many fires were large, estimated at 1,000 to 10,000 acres (400 to 4,000 ha) (based on Heinselman 1981).

Postfire Succession—With so many species of both boreal and southern affinities in the Laurentian Forest, many combinations of species form in postfire succession. Long fire rotations create extensive stands of mature and old hardwoods (American beech, birches, and maples). Stand-replacement fires are followed by a flush of shrubs and saplings, including red and sugar maple, paper and gray birch, alders, quaking aspen and bigtooth aspen, and cherry and shadbush species. White and red pines are also prominent, especially on sandy soils.

Early in succession, northern red oak and bur oak often intermix with less shade-tolerant hardwoods and pines. Pole-sized trees may be dense. Balsam fir and red spruce invade and gradually increase in dominance. On dry ridges, sugar maple, red maple, American beech, and oaks eventually dominate. On uplands, sugar maple, yellow birch, and American beech dominate the usually long-lasting mature stage. Eastern hemlock dominates on mesic sites with red spruce, yellow birch, paper birch, and occasional eastern white pine. One particularly volatile combination of species occurs in the northern Laurentian Forest and the southeastern fringe of the Boreal Forest. There balsam fir is a dominant species that supports outbreaks of spruce budworm; budworm-killed forest is highly flammable.

Eastern Ecosystems and the Great Plains

Eastern Deciduous Forest

Vegetation and Fire Regimes—The Eastern Deciduous Forest had understory and stand-replacing fire regimes in the centuries before settlement by European Americans. Lightning-caused fires were common in the mixed mesophytic hardwood forests of the Appalachian uplands and the Mississippi Valley (Komarek 1974). Because precipitation was plentiful in most years, the fires usually burned small areas. Some areas in this forest type burned frequently, including those near the bluegrass grasslands of Kentucky, which supported herds of bison (Komarek 1974). Historians and anthropologists now suggest that a substantial proportion of this deciduous forest was kept in early successional stages through shifting cultivation, firewood cutting, and extensive burning by agricultural tribes of Native Americans (Day 1953; MacCleery 1993). Annual burning in these areas created parklike stands of large, open-grown trees, a high proportion of which were fire-resistant oaks and eastern white pines. These hardwood forests had little understory and many openings.

Stand-replacement and mixed-severity fires shaped most of the pine forests of Eastern North America, particularly the extensive stands of eastern white and red pine along the northern periphery of the Midwestern States and in southern Ontario (Szeicz and MacDonald 1990; Vogl 1970) and the pitch pine and eastern redcedar forests on the Atlantic Coastal Plain (Wright and Bailey 1982).

Postfire Succession—Pines are common early successional species in the Eastern Deciduous Forest (Komarek 1974). Hardwood species with vigorous sprouting ability, especially oaks, also tend to dominate after fire. Increased prominence of oaks is one of the most common results of disturbance in this kind of forest (Williams 1989). Shade-intolerant species, including tuliptree and sweetgum, regenerate well on burned land (Little 1974). Many herbaceous species invade burned areas aggressively. In southern parts of the region, repeated burning leads to a mixed ground cover of grasses and legumes amid patches of trees (Komarek 1974). Without repeated disturbance, hardwood trees reoccupy the land with oaks in the vanguard. Continued absence of fire permits Eastern deciduous forests to be dominated by sugar maple, red maple, eastern hemlock, and American beech.

Southeastern Forests

Vegetation and Fire Regimes—The Southeastern Pine Region extends in a great arc from eastern Texas around the Appalachian uplands to Virginia. The vegetation is characterized by the “southern pines”—longleaf, slash, loblolly, shortleaf, and sand pines (Komarek 1974; Wright and Bailey 1982). These pine species tolerate and even depend upon fire to different degrees, while most hardwood species in the Southeast are suppressed by fire. Protection from fire enables hardwood forests to develop.

The Southeastern Pine Region has a high incidence of lightning strikes. Lightning and ignitions by aboriginal peoples caused understory fires in most longleaf pine forests every 1 to 15 years during presettlement times (Christensen 1988; Myers 1990). Since many of the grass and forb species associated with these forests also depend upon frequent fires (Frost and others 1986), cattlemen, farmers, and hunters continued burning the southern pine forests until the widespread adoption of fire suppression practices in the 1930s. By that time, intentional burning to improve wildlife habitat was already recognized as a management tool; by 1950 it was a common practice (Riebold 1971). Longleaf pine dominated the Coastal Plain forests (except wetlands) until the early 1900s. Several factors, including alteration of the fire regime, have since favored dominance by loblolly and slash pines, which are somewhat less fire tolerant.

In eastern Oklahoma, shortleaf pine forests probably burned in large, low-severity understory fires at intervals of about 2 to 5 years prior to fire exclusion (Masters and others 1995).

The dominant vegetation in sand pine-scrub stands was killed or top-killed by fire every 15 to 100 years. One such fire burned 34,000 acres (14,000 ha) in 4 hours (Myers 1990). Maintenance of sand pine-scrub vegetation requires these infrequent, severe fires; more frequent fires can convert sand pine-scrub to longleaf pine (Christensen 1988).

Postfire Succession—The overriding impact of fire in the Southeastern Pine Region has been the maintenance of pine forest at the expense of hardwood forest. Relatively frequent understory fires shape a pine forest of variable density and well developed ground cover. Understory burning removes shrubs and small trees as sources of mast for wildlife, but it creates and maintains a vigorous understory of grasses, forbs, and fire-resistant shrubs (Wright and Bailey 1982). In the absence of fire, hardwood species invade the pine stands and deciduous forests develop. In much of the region, these are dominated by a mixture of oak and hickory in combination with many other deciduous species (Eyre 1980).

Prairie Grassland

Vegetation and Fire Regimes—The primeval prairie grasslands of North America stretched from the Gulf Coast in Texas north to central Alberta and from Illinois to western Montana. Precipitation increases from west to east, creating three north-south belts of vegetation—shortgrass prairie in the West, mixed prairie in the North and East, and tallgrass prairie in the Central and Eastern regions. Of all natural regions of North America, the Prairie Grassland has been most heavily impacted by human use. Tallgrass Prairie has been almost totally converted to agriculture. Development is somewhat less in westerly parts of the grassland. Substantial portions of the Shortgrass Prairie remain in use for cattle grazing.

The fire regime of the grasslands prior to settlement and development for agriculture was one of stand-replacing fires on a short return interval, every year in some areas (DeBano and others 1998; Wright and Bailey 1982). Ignitions due to lightning were common (Higgins 1984), and Native Americans ignited many fires (Wright and Bailey 1982). Prairie fires were often vast, burning into the forest margins and preventing tree invasion of grasslands (Reichman 1987).

Postfire Succession—In prairie grasslands, burning maintains dominance by fire-adapted grasses and forbs. Fire also maintains the productivity of grasslands, supplying fresh, nutritious vegetation that is used by herbivores. Fire effects are strongly influenced by season of burning and moisture conditions. Burning outside the growing season causes little change in biomass yield or species dominance; fire during the growing season is likely to reduce yield and change species dominance. Postfire recovery is delayed if a site is burned during drought or, where annuals dominate, before seed set (DeBano and others 1998).

All grassland communities are subject to invasion by shrubs and trees in the absence of fire. Invading species include oaks, pines, junipers, mesquite, and aspen (Wright and Bailey 1982).

Western Forests

Rocky Mountain Forest

Vegetation and Fire Regimes—The Rocky Mountain Forest Region occupies inland mountain ranges and plateaus from New Mexico to Alberta and British Columbia. Vegetation patterns are complex and varied due to climatic differences that arise from variation in elevation and topography and the great latitudinal extent of the region. The forests are mainly coniferous. Important dominant trees include ponderosa pine, lodgepole pine, spruces, and firs. West of the Rocky Mountains, from Idaho north into British Columbia, Douglas-fir, western larch, and grand fir are dominant tree species.

Rocky Mountain forests in past centuries had a variety of fire regimes: understory, mixed-severity, and stand-replacement. At low elevations, understory fires maintained large areas of ponderosa pine and Douglas-fir in an open, parklike structure for thousands of years prior to the 1900s. Fires on these sites increased grass and forb production. Stand-replacing fires and complex mixed-severity fires were common in subalpine spruce-fir and lodgepole pine forests; understory fires also occurred, especially on dry sites.

Presettlement mean fire return intervals in Rocky Mountain forests ranged from less than 10 years (Arno 1976) to more than 300 years (Romme 1980). Forests with a multistoried structure, including dense thickets of young conifers, were more likely to experience stand-replacing fire than open, parklike stands. When ignition occurred in lodgepole pine forests, old stands were more likely to burn than young stands (Romme and Despain 1989).

Postfire Succession—Stand-replacing fires were unusual in ponderosa pine during presettlement times but are now more common because of increased surface fuels and "ladder" fuels (shrubs and young trees that provide continuous fine material from the forest floor into the crowns of dominant trees). In presettlement times, repeated understory fire maintained an open forest with grasses and forbs on the forest floor and scattered patches of conifer regeneration. Fires occasionally killed large, old trees, creating openings where the exposed mineral soil provided a seedbed suitable for pine reproduction (Weaver 1974). Many forests were composed of multiple patches of even-aged trees.

Higher-elevation spruce-fir forests experience occasional stand-replacing fire. Conifer seedlings and deciduous shrubs sprout after being top-killed by fire and dominate regrowth within a few years after fire. Regenerating stands often produce large volumes of browse until the tree canopy closes, 25 or more years after fire. In the Northern Rocky Mountains, where lodgepole pine forests are mixed with spruce-fir, serotinous lodgepole pine cones open after being heated by fire. Fire thus simultaneously creates a good seedbed for pine and produces a rain of seed. The result is quick regeneration of lodgepole pine, often in dense stands.

Sierra Forest

Vegetation and Fire Regimes—Mixed conifer forests occur in the Sierra Nevada of California. Important species include Douglas-fir, incense cedar, sugar pine, white fir, and California red fir.

Sierra forests are famous for their groves of giant sequoia trees. Understory fires typically burned these forests at average intervals of 3 to 25 years. This fire regime produced an open structure with a grass and forb understory and scattered tree regeneration, similar to the structure of Rocky Mountain ponderosa pine forests.

Sierra Nevada forests also include ponderosa pine, with a presettlement regime of frequent understory fire; montane forests with a complex mixture of conifer species; and subalpine forests of lodgepole pine, whitebark pine, and California red fir. Montane and subalpine forests had a complex presettlement fire regime that included infrequent understory fire, mixed-severity fire, and stand-replacement fires of all sizes (Kilgore 1981; Taylor and Halpern 1991).

Postfire Succession—The understory fires characteristic of Sierra mixed conifer and ponderosa pine forests maintained open structures with little accumulation of debris on the ground (Kilgore 1981; Weaver 1974). Understory fire maintained dominance by pines and giant sequoias, with understory species including manzanita, deerbrush, wedgeleaf ceanothus, and bitter cherry (Wright and Bailey 1982). In the absence of fire, less fire-resistant species, including white fir and incense cedar, invade and develop into dense, tangled patches of young trees.

Pacific Coast Maritime Forest

Vegetation and Fire Regimes—The Pacific Coast Maritime Forest is the most productive forest type in the world (Agee 1993). The area is ecologically important because of the many species, including animals, that depend on old age classes of trees. The area is economically important because of its rapid rates of tree growth and biomass accumulation. The Pacific Coast region has wet winters and dry periods in the summer. In late summer, fire danger can become high, leading to stand-replacing crown fires with awesome intensity as described by Weaver (1974) for the 1933 Tillamook fire in Oregon.

Sitka spruce, western hemlock, and coast redwood dominate Pacific Coast Maritime forests. In past centuries, fires occurred infrequently in Sitka spruce and coastal forests of western hemlock, although most western hemlock forests show evidence that they were initiated following fire. Inland western hemlock forests probably burned in a regime of somewhat more frequent, mixed-severity fire. In redwood forests on relatively dry sites, fires of all kinds—understory, mixed-severity, and stand-replacement—were more common, occurring as frequently as every 50 years (Agee 1993).

Near the coast, long fire rotations in presettlement times resulted in a large proportion, probably about two-thirds, of the forest in mature and old age classes at any one time. There was thus ample habitat for flora and fauna that prefer or can survive in old growth.

Postfire Succession—Douglas-fir is important over much of the Pacific Coast Maritime Region because it is resistant to fire as an old tree, is able to colonize disturbed sites, and has a life span of several hundred years. On upland sites in the region, stand-replacing fire can be followed by dense shrub communities dominated by salmonberry, salal, red huckleberry, and vine maple (Agee 1993). Even where Douglas-fir becomes established immediately after fire, red alder may overtop it for many years (Wright and Bailey 1982). Postfire shrubfields sometimes persist indefinitely and sometimes are replaced by shade-tolerant conifers that regenerate beneath the shrub canopy.

Understory fires tend to eliminate most trees except large Douglas-fir and coastal redwood, if present (Agee 1993). Shade-tolerant trees regenerate under the remaining canopy. Mixed-severity fires produce gaps in which Douglas-fir regenerates and grows rapidly. Where redwood grows on alluvial sites, mixed-severity fire favors development of large, old redwood trees along with dense redwood regeneration.

Western Woodlands, Shrublands, and Grasslands

Pinyon-Juniper

Vegetation and Fire Regimes—Pinyon-juniper woodlands are dry, open forests occurring in the Southwestern United States. Prominent overstory species include the pinyon pines, Utah juniper, oneseed juniper, and alligator juniper. Pinyon-juniper woodland occupies elevations between higher oak woodlands and lower grass- and shrub-dominated areas (Wright and Bailey 1982). Because of the open nature of pinyon-juniper woodland, grasses and shrubs are prominent in the understory.

In presettlement times, stand-replacing fires probably occurred at intervals averaging less than 50 years in pinyon-juniper woodlands. Because of fire, areas with mature pinyon-juniper cover were somewhat restricted to locations with rocky soils and rough topography, which inhibited fire spread (Bradley and others 1992; Kilgore 1981; Wright and Bailey 1982). Where livestock grazing reduced herbaceous fuels, fire occurrence decreased and pinyon-juniper woodlands expanded. In mature, closed stands, fire spreads poorly because surface fuels are sparse. High winds and a high proportion of pine to juniper increase the potential for fire spread (Wright and Bailey 1982). Fire-caused tree mortality is likely to be great where fine fuels are dense or tumbleweeds have accumulated.

Postfire Succession—The impact of fire depends on tree density and the amount of grass and litter in the stand. For a few years after fire, pinyon-juniper woodlands present a stark landscape of dead trees and nearly bare soil. Annual plants become established in a few years. These are followed by perennial grasses and forbs. Invading plants often include weedy species, especially on bare soil. Junipers and shrubs typically reestablish in 4 to 6 years. After 40 to 60 years, the shrubs are replaced by a new stand of juniper (Barney and Frischnecht 1974; Koniak 1985). Development of a mixture of mature pinyons and junipers can require up to 300 years (West and Van Pelt 1987).

Chaparral and Western Oak Woodlands

Vegetation and Fire Regimes—Chaparral and western oak woodlands include broad-leafed shrub and tree species that are well adapted to fire. These plant communities occur in dry mountains and foothills throughout the Southwestern United States. The largest area is in southwestern California and the foothills of the Sierra Nevada, where chaparral is notorious for its frequent, fast-spreading, stand-replacing fires. Following fire, chaparral species sprout and also establish vigorously from seed. Many species have seed that germinates best after being heated by fire (DeBano and others 1998). In California chaparral, stand-replacing fires have occurred every 20 to 40 years for hundreds of years (Kilgore 1981). Fires were less frequent in Arizona chaparral, at higher elevations in California, and on northern aspects.

Oak woodlands are characterized by species that resprout vigorously after fire; Gambel oak dominates many such woodlands in Utah and Colorado. Oak woodlands had understory fire regimes with occasional stand-replacing fire in presettlement times (Wright and Bailey 1982). Fire frequency was reduced in areas where grazing reduced surface fuels (Bradley and others 1992).

Postfire Succession—Annual and perennial herbs flourish after fire in chaparral, along with seedling and resprouting shrubs. Herbs are gradually eliminated as the dense overstory of large shrubs matures (DeBano and others 1998).

Browse productivity in chaparral increases dramatically during the first 4 to 6 years after burning (Wright and Bailey 1982) but declines thereafter. For a decade or two after fire, chaparral is quite fire resistant (Wright 1986). Burning at 20- to 30-year intervals maintains a diverse mixture of species. If the fire return interval is longer, sprouting species will dominate, reducing plant species diversity.

In oak woodlands, fire either underburns or top-kills the dominant species and stimulates suckering at the bases of oaks. It thus changes the structure of oak woodlands, stimulates other shrubs, and produces a 2- to 3-year increase in productivity of grasses and forbs. Perpetuation of oaks and optimization of mast are wildlife management objectives in some locations because of widespread wildlife use of mast. In California, acorns are eaten by nearly 100 species of animals, including California quail, wild turkey, deer, and bear (McDonald and Huber 1995).

Sagebrush and Sagebrush Grasslands

Vegetation and Fire Regimes—Sagebrush dominates large areas in the Western United States in dense shrub stands and mixtures with grasslands. A common associate is antelope bitterbrush. Grasses and forbs are abundant. Sagebrush grasslands often intermix with forest cover, especially at higher elevations.

In presettlement times, fires burned sagebrush grasslands at intervals as short as 17 years and as long as 100 years (Wright and Bailey 1982). Fire severity in sagebrush varied, depending on the occurrence of sufficient grass and litter to carry fire. If fuel was sufficient, fires were stand-replacing and severe, burning through the shrub crowns. Where fuels were sparse, fires were patchy. Varied patterns of vegetation and seasonal differences in burning conditions produced substantial differences in fire severity and effects.

Cheatgrass, an exotic annual, is favored by frequent burns, especially spring burns (Wright and Bailey 1982). Cheatgrass provides an accumulation of fine fuel that burns readily, so it alters the fire regime in sagebrush grasslands to much more frequent, stand-replacing fire (Kilgore 1981; Knick 1999). This disturbance reduces shrub cover severely and eliminates the patchy pattern formerly characteristic of sagebrush-dominated landscapes.

Postfire Succession—Fires in sagebrush grasslands reduce woody shrub species. Big sagebrush, a valuable wildlife browse species, is highly susceptible to injury from fire. Its recovery depends on season and severity of burn, summer precipitation, and frequency of burning. Big sagebrush may take more than 50 years to recover preburn dominance (Wright 1986). Antelope bitterbrush may be killed or only top-killed by fire, depending on the ecotype present and fire severity (Bedunah and others 1995).

Many grass and forb species thrive after fire and may delay regrowth of shrubs. Fires occurring every few years reduce perennial grasses and favor annuals, including cheatgrass. Shrubs reinvade during wet years. Sagebrush grasslands occasionally undergo severe droughts, which provide a major setback to shrub vegetation even in the absence of fire (Wright 1986).

Deserts

Vegetation and Fire Regimes—North American deserts occur in two separate areas of dry climate. The larger of the two areas extends from Baja California north through the Great Basin to central Idaho and Oregon (Humphrey 1974). This large region supports three floristically distinct deserts: the Sonoran Desert in the south, the Mojave Desert in southeastern California and southern Nevada, and the large Great Basin Desert to the north. The second North American desert area is the Chihuahuan Desert, located in the northern interior of Mexico and southern New Mexico.

In deserts with woody plants and tall cacti, fire severity in presettlement times depended on fuel loading and continuity. Severe fire was possible only after a moist, productive growing season; mixed-severity fire was more likely at other times. Fire was most frequent and widespread in the Great Basin Desert because of its greater shrub biomass (sagebrush) and because grass biomass was usually sufficient to carry fire between clumps of shrubs

(Kilgore 1981). Next to the Great Basin Desert, the Chihuahuan Desert was the most prone to fire, while the Sonoran and Mojave Deserts only had enough ground cover to carry fire after occasional, unusually wet growing seasons (Humphrey 1974). In a review of fire effects on succulent plants, Thomas (1991) estimates that intervals between fires prior to European American settlement were as short as 3 years in some desert grasslands and more than 250 years in dry areas such as the Sonoran Desert.

Seasonal weather and grazing influence fire potential in deserts (Wright and Bailey 1982). A wet year produces large quantities of grasses and forbs, which provide fuel to carry fire. Grazing reduces these fine fuels, thus reducing potential fire spread.

Postfire Succession—Regrowth following fire depends on the availability of moisture. If burning is followed by a wet season, production of perennial grasses and some forbs may increase (Wright and Bailey 1982). In the most arid desert areas, fires may reduce density of shrubs and cacti for 50 to 100 years (Wright 1986). However, studies have shown substantial differences between species and also complex interactions among available moisture, grazing, and plant species (Cable 1967; MacPhee 1991; Wright and Bailey 1982). Several studies report increases in exotic annual grasses, including red brome and red stork's bill, after fire in desert ecosystems; both frequency and intensity of fires may have increased since the introduction of these grasses (Rogers and Steele 1980; Young and Evans 1973).

Subtropical Ecosystems

Florida Wetlands

Vegetation and Fire Regimes—The subtropical region of Florida is underlain by an expanse of limestone bedrock that is almost level and barely above sea level. Due to the area's flat surface and high annual rainfall—59 inches, 149 cm (Wright and Bailey 1982)—wetland covers much of the area. Lower places in the bedrock surface accumulate peat and support vegetation dominated by sawgrass. Where elevations are slightly higher, fresh water swamp or wet prairie vegetation occurs. Dry land occurs as knolls called "hammocks," which support mixed hardwood forest. Despite its extensive wetlands, fire has always influenced the ecology of southern Florida.

Postfire Succession—Burning has apparently maintained coastal marshes against mangrove invasion. Frequent fires in sawgrass kept fuel loadings low and prevented severe fires that would consume peat deposits. As peat accumulates, tree distribution expands out from the hammocks, increasing habitat for terrestrial fauna and decreasing habitat for wetland animals (Wright and Bailey 1982). Understory fires, occurring about five times per century on the average, maintained cypress stands by killing young hardwoods and suppressing hardwood regeneration (Ewel 1990). Severe fires after logging or draining swamps alter successional pathways, enhancing willows and eventual succession to hardwood forest.

Notes

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Chapter 3: Direct Effects of Fire and Animal Responses

This chapter summarizes current knowledge about the immediate and short term (days to weeks) effects of fire on terrestrial vertebrates: fire related mortality, emigration, and immigration. Within these topics, we describe fire effects mainly for two animal classes—birds and mammals. Information regarding reptiles, amphibians, and invertebrates is included if available in the literature.

Most animal species respond predictably to the passage of fire (Komarek 1969; Lyon and others 1978). These responses vary widely among species. Many vertebrate species flee or seek refuge, but some vertebrates and many insects are attracted to burning areas. Other behavioral responses to fire include rescuing young from burrows, approaching flames and smoke to forage, and entering recent burns to feed on charcoal and ash (Komarek 1969).

Injury and Mortality

Despite the perception by the general public that wildland fire is devastating to animals, fires generally kill and injure a relatively small proportion of animal populations. Ambient temperatures over 145 °F are lethal to small mammals (Howard and others 1959), and it is reasonable to assume the threshold does not differ greatly for large mammals or birds. Most fires thus have the potential to injure or kill fauna, and large, intense fires are certainly dangerous to animals caught in their path (Bendell 1974; Singer and Schullery 1989). Animals with limited mobility living above ground appear to be most vulnerable to fire-caused injury and mortality, but occasionally even large mammals are killed by fire. The large fires of 1988 in the Greater Yellowstone Area killed about 1 percent of the area's elk population (Singer and Schullery 1989). Fire effects on habitat influenced the species' population much more dramatically than did direct mortality. Because of drought during the summer of 1988 and forage loss on burned winter range, elk mortality was high in the winter of 1988 to 1989, as high as 40 percent at one location (Singer and others 1989; Vales and Peek 1996).

Fire may threaten a population that is already small if the species is limited in range and mobility or has specialized reproductive habits (Smith and Fischer 1997). The now extinct heath hen was restricted to Martha's Vineyard for many years before its extirpation, where scrub fires probably accelerated its demise (Lloyd 1938).

Season of burn is often an important variable in fauna mortality. Burning during nesting season appears to be most detrimental to bird and small mammal populations (Erwin and Stasiak 1979). Following the burning of a reestablished prairie in Nebraska, mortality of harvest mice in their aboveground nests was evident, and many nests of ground-nesting birds were found burned. Nestlings and juveniles of small mammals are not always killed by fire, however. Komarek (1969) observed adult cotton rats carrying young with eyes still closed out of an area while fire approached. While fire-caused mortality may sometimes be high for rodent species, their high reproductive potential enables them to increase rapidly in favorable environments and disperse readily into burned areas. Kaufman and others (1988b) describe this pattern for deer mouse and western harvest mouse populations in Kansas tallgrass prairie.

Birds

Fire-caused bird mortality depends on the season, uniformity, and severity of burning (Kruse and Piehl 1986; Lehman and Allendorf 1989; Robbins and Myers 1992). Mortality of adult songbirds is usually considered minor, but mortality of nestlings and fledglings does occur. In addition, a review by Finch and others (1997) points out that reproductive success may be reduced in the first postfire year because of food reductions from spring fires. Nest destruction and mortality of young have been reported for several ground-nesting species, including ruffed, spruce, and sharp-tailed grouse (Grange 1948), northern harrier (Kruse and Piehl 1986), and greater prairie-chicken (Svedarsky and others 1986). While eggs and young of ground-nesting birds are vulnerable to spring fires, long-term fire effects on bird populations depend partly on their tendency to renest. According to a review by Robbins and Myers (1992), wild turkeys rarely renest if their nests are destroyed after 2 to 3 weeks of incubation, while northern bobwhite may renest two or three times during a summer. For this reason, many biologists consider turkeys more vulnerable to fire. A mixed-grass prairie habitat in North Dakota was burned during the nesting season, but 69 percent of active clutches survived the fire and 37 percent eventually hatched. Nesting success was attributed in part to areas skipped by the fire as it burned in a mosaic pattern (Kruse and Piehl 1986). Underground nests, such as that of the burrowing owl, are probably safe from most fires.

In forested areas, fire effects on birds depend largely on fire severity. The young of birds nesting on the ground and in low vegetation are vulnerable even to understory fire during nesting season. Species nesting in the canopy could be injured by intense surface fire and crown fire, but this kind of fire behavior is more common in late summer than during the nesting season.

Mammals

The ability of mammals to survive fire depends on their mobility and on the uniformity, severity, size, and duration of the fire (Wright and Bailey 1982). Most small mammals seek refuge underground or in sheltered places within the burn, whereas large mammals must find a safe location in unburned patches or outside the burn. Lyon and others (1978) observe that small animals are somewhat more likely to panic in response to fire than large, highly mobile animals, which tend to move calmly about the periphery of a fire (fig. 5).

Most small mammals avoid fire by using underground tunnels, pathways under moist forest litter, stump and root holes, and spaces under rock, talus, and large dead wood (Ford and others 1999). Not all survive. Ver Steeg and others (1983), for instance, found numerous dead meadow voles after an early spring fire in Illinois grassland. Adequate ventilation inside burrows is essential for animal survival (Bendell 1974). Burrows with multiple entrances may be better ventilated than those with just one entrance (Geluso and others 1986). Small mammals living in burrows survived stand-replacing fire during summer in an ungrazed sagebrush-bunchgrass community in southeastern Washington (Hedlund and Rickard 1981). Most voles survived a prescribed burn in Nebraska grassland (Geluso and others 1986). Several retreated underground at the approach of the fire and returned to the surface after the fire had passed, apparently unharmed. Others remained aboveground, moving quickly through dense vegetation to outrun the fire. One individual sought refuge upon a raised mound of soil created by plains pocket gophers and was adequately sheltered there from heat and flame.

Small rodents that construct surface-level nests, such as brush rabbits, harvest mice, and woodrats (dusky-footed, desert, and white-throated), are more vulnerable to fire-caused mortality than deeper-nesting species, especially because their nests are constructed of dry, flammable materials (Kaufman and others 1988b; Quinn 1979; Simons 1991). Woodrats are particularly susceptible to fire mortality because of their reluctance to leave their houses even when a fire is actively burning (Simons 1991).

Direct fire-caused mortality has been reported for large as well as small mammals, including coyote, white-tailed deer, mule deer, elk, bison, black bear, and moose (French and French 1996; Gasaway and DuBois 1985; Hines 1973; Kramp and others 1983; Oliver and others 1998). Large mammal mortality is most likely when fire fronts are wide and fast moving, fires are actively crowning, and thick ground smoke

Figure 5—Bison foraging and resting near burning area, Yellowstone National Park. Photo by Jeff Henry, courtesy of National Park Service.

occurs. Singer and Schullery (1989) report that most of the large animals killed by the Yellowstone fires of 1988 died of smoke inhalation. Because mortality rates of large mammals are low, direct fire-caused mortality has little influence on populations of these species as a whole (French and French 1996). Animal mortality, of course, provides food for scavenger fauna (fig. 6). The largest group of fire-killed elk in Yellowstone National Park was monitored for several months after it burned. Grizzly bears, black bears, coyotes, bald

Figure 6—Fire-killed deer after stand-replacing fire, Yellowstone National Park. Note "whitewash" on the deer's flanks, evidence of use by scavenging birds. Photo courtesy of National Park Service.

eagles, golden eagles, and common ravens fed on the carcasses (French and French 1996). According to Blanchard and Knight (1996), the increased availability of carcasses benefited grizzly bears because drought had reduced other food sources.

The stand-replacing and mixed-severity fires of 1988 in the Greater Yellowstone Area, which occurred mostly in lodgepole pine-dominated forest, provided opportunities to study animal behavior during burns. Most thoroughly studied were large mammals, including bison, elk, bear, moose, and deer. French and French (1996) observed no large mammals fleeing a fire, and most appeared "indifferent" even to crowning fires. Singer and Schullery (1989) concluded that large mammals were sufficiently mobile to simply move away from danger during the fires. Bison, elk, and other ungulates grazed and rested within sight of flames, often 100 m or less from burning trees.

Reptiles and Amphibians

According to a review by Russell and others (1999), there are few reports of fire-caused injury to herpetofauna, even though many of these animals, particularly amphibians, have limited mobility. In a review of literature in the Southeastern States, Means and Campbell (1981) reported only one species that may suffer substantial population losses from fire, the eastern glass lizard. Another review (Scott 1996) mentions the box turtle as being vulnerable to fire, but there are many reports of box turtles and other turtle species surviving fires by burrowing into the soil (Russell and others 1999). No dead amphibians or reptiles were found after understory burning in a longleaf pine forest in Florida (Means and Campbell 1981). The vulnerability of snakes to fire may increase while they are in ecdysis (the process of shedding skin); of 68 eastern diamondback rattlesnakes marked before a fire in Florida, the only two killed were in mid-ecdysis.

Many reptiles and amphibians live in mesic habitat. Woodland salamanders in the southern Appalachians, for instance, use riparian sites and sites with plentiful, moist leaf litter. These sites are likely to burn less often and less severely than upland sites. The resulting microsite variation within burns may account for observations that fire has little effect on populations of these species (Ford and others 1999). Wetlands may provide refuge from fire, and activities such as breeding by aquatic species may be carried out with little interruption by fire (Russell and others 1999).

Many desert and semidesert habitats burned infrequently in past centuries because of sparse fuels. In these areas, as in mesic sites, patchy fire spread may protect herpetofauna from fire-caused injury and mortality. A growing concern is the conversion of vegetation in desert and semidesert, which burned infrequently in past centuries, to vegetation that now burns every few years due to invasion of weedy species (see "Effects of Altered Fire Regimes" in chapter 4). Animals in these ecosystems may not be adapted to avoid fire.

Invertebrates

The vulnerability of insects and other invertebrates to fire depends on their location at the time of fire. While adult forms can burrow or fly to escape injury, species with immobile life stages that occur in surface litter or aboveground plant tissue are more vulnerable. However, aboveground microsites, such the unburned center of a grass clump, can provide protection (Robbins and Myers 1992). Seasonality of fire no doubt interacts with phenology for many invertebrates. Research is needed on fire effects at all stages of insect life cycles, even though larval stages may be more difficult to track than adult stages (Pickering 1997).

An August understory burn in South Carolina forest reduced the soil mesofauna as measured the day after fire, but annually burned plots had generally higher populations of soil mesofauna than did plots that had not been burned in 3 years or more (Metz and Farrier 1971).

Escape and Emigration

A second popular concept regarding animals' response to fire is that they leave the area permanently as soon as fire is detected. While non-burrowing mammals and most birds do leave their habitat while it is burning, many return within hours or days. Others emigrate because the food and cover they require are not available in the burn. The length of time before these species return depends on how much fire altered the habitat structure and food supply.

Birds

Many birds leave burning areas to avoid injury. Some return to take advantage of the altered habitat, but others abandon burned areas because the habitat does not provide the structure or foods that they require to survive and reproduce. While some raptors are attracted to fire (see "Immigration" below), others move out of an area immediately after fire. After the large Marble-Cone fire in California, the spotted owls in Miller Canyon abandoned their habitat (Elliott 1985). Spotted owls in south-central Washington continued to use areas burned by understory fire but avoided stand-replacement burns, probably because their prey had been reduced (Bevis and others 1997). Structural features make recent burns unsuitable habitat for some species. Although stand-replacing fire in a Douglas-fir forest in western Montana favored birds that feed on insects, at least one insect feeder, the Swainson's thrush, abandoned the burn immediately (Lyon and Marzluff 1985), probably due to its need for cover.

Several studies report declines in bird abundance or species diversity in the first year or two after stand-replacing fire, but few reports are available for the months immediately following fire. After a late October fire in 1980 in coastal chaparral, California, fewer birds of all species were seen in November. Three months later, the bird population remained 26 percent below average (McClure 1981). The number of bird populations absent or declining in postfire years 1 and 2 has been reported to exceed the number of populations remaining stable or increasing after fires in Saskatchewan grassland (Pylypec 1991), Kansas shrub-grassland (Zimmerman 1992), California coastal sage scrub (Stanton 1986), and Wyoming spruce-fir-lodgepole pine forest (Taylor and Barmore 1980). Many bird species return to burned habitat 2 to 3 years after fire (Lyon and Marzluff 1985; Wirtz 1979).

Mammals

Because large mammals, such as moose and deer, depend on vegetation for forage, bedding, cover, and thermal protection, they abandon burned areas if fire removes many of the habitat features they need. Thus stand-replacing fires and understory burns that are severe enough to top-kill shrubs and young trees seem more likely to trigger high rates of emigration than patchy or low-severity fires. Woodland caribou in southeastern Manitoba avoided boreal forest burned by stand-replacing fire in favor of bog communities, lakes, and other unburned areas. Caribou may continue to avoid burns for 50 years or more, until lichens become reestablished in the new forest (Schaefer and Pruitt 1991; Thomas and others 1995). If recent burns provide some, but not all, habitat requirements for a species, the animals may stay near the edges of a burn. Immediately following large, stand-replacing fires in chapparal, Ashcraft (1979) reported mule deer grazing no farther than 300 feet (90 m) from cover.

Many small mammal species also leave burned habitats. Based on intensive trapping results, Vacanti and Geluso (1985) found that most voles survived a prescribed burn in Nebraska grassland but left the burned area until a new litter layer had accumulated, about two growing seasons later. Possible reasons for emigration included decreased protection from predators, decreased food availability, and more interactions among individuals. In the year after prescribed understory burns in conifer woodland with a shrubby understory in California, the abundance of small mammals was almost three times greater on unburned than burned plots, even though species composition did not vary significantly between burned and unburned areas (Blankenship 1982). Densities of western harvest mouse decreased the first year after tallgrass prairie was burned because their aboveground nests were destroyed and they left the area. During the same period, deer mice increased, apparently attracted by sparse ground cover that made seeds easy to find. Western harvest mouse densities in the burn increased the following spring and summer, with the populations on unburned sites serving as sources of dispersing individuals (Kaufman and others 1988b). In a southwestern Idaho shadscale-winterfat community, fire reduced the abundance of small mammals in the first postfire year. A decline in American badger numbers on the burn accompanied the small mammal decline (Groves and Steenhof 1988).

The effects of fire on mammal species are related to the uniformity and pattern of fire on the landscape. Fire has been cited by many authors as detrimental to American marten food and habitat (see Koehler and Hornocker 1977). However, a mixed-severity fire in an area of lodgepole pine, spruce, and fir in northern Idaho left a mosaic of forest types that supported a diversity of cover and food types favorable for marten (Koehler and Hornocker 1977). During summer and fall, American marten feed on ground squirrels, fruits, and insects in areas burned by stand-replacing fire. They require dense forest during most winters, but they use open forest during mild winters. Thus while large, uniform burns do not favor American marten, a mosaic of vegetation shaped in part by recent fire may do so.

Immigration

Many animals are actually attracted to fire, smoke, and recently burned areas. Some of the most interesting research regarding immigration in response to fire is in the field of insect ecology. The beetles of the subgenus *Melanophila* ("dark loving"), for instance, use infrared radiation sensors to find burning trees, where they mate and lay eggs (Hart 1998). Most birds and mammals that immigrate in response to fire are attracted by food resources.

Birds

A few bird species are attracted to active burns, and many increase in the days and weeks that follow fire. Parker (1974) reports that black vulture, northern harrier, red-shouldered hawk, and American kestrel were attracted to an agricultural (corn stubble) fire in Kansas. In the Southwest, raptor and scavenger species that are attracted to fire or use recent burns for hunting include northern harrier, American kestrel, red-tailed hawk, red-shouldered hawk, Cooper's hawk, and turkey and black vultures (Dodd 1988). After the large, severe Marble-Cone fire in California, western screech-owls moved into the burned area (Elliott 1985). Many species of birds were attracted to a 440-acre (180-ha) burn on the Superior National Forest, Minnesota. About 10 weeks after the fire, the area was alive with bird activity. Species included sparrows, American robin, barn swallow, common grackle, American kestrel, northern flicker, common raven, hairy woodpecker, great blue heron, eastern bluebird, and black-backed woodpecker (Stensaas 1989).

Predators and scavengers are often attracted to burns because their food is more abundant or more exposed than on unburned sites. During small prescribed burns in Texas bunchgrass and mesquite-grass stands, white-tailed hawks were attracted to grasshoppers chased from cover by the fires. Turkey vultures and crested caracaras fed on small mammals that had died in the fire (Tewes 1984). Stand-replacing and mixed-severity fire in a Douglas-fir forest in western Montana favored birds feeding on insects (Lyon and Marzluff 1985). Immediately after the fire, intense activity by wood-boring insects, parasites of wood borers, and predaceous flies occurred, accompanied by "almost frenetic" feeding by warblers and woodpeckers. In another study of grassland fire, American kestrel and red-tailed hawk increased after burning (Crowner and Barrett 1979). During a grassland fire in Florida, both cattle egrets and American kestrels foraged close to the flames. Apparently the egrets were attracted to vertebrates and invertebrates, and the kestrels were preying exclusively on insects as they flew out of the fire, into the wind (Smallwood and others 1982).

Several studies show that woodpeckers are particularly attracted to burned areas. Black-backed woodpeckers are almost restricted to standing dead, burned forests in the Northern Rocky Mountains (Caton 1996; Hutto 1995; Lyon and Marzluff 1985) (fig. 7). Schardien and Jackson (1978) found pileated woodpeckers foraging extensively on logs in an area in Mississippi that had burned 2 weeks earlier; an abundant food supply of wood-boring beetles appeared to be the primary attraction. Woodpeckers were attracted to a stand-replacement burn in coastal sage scrub, probably to feed on insects in the fire-killed cover (Moriarty and others 1985).

When small mammals are attracted to abundant new growth in the months following fire, predators and scavengers are attracted too. Abundant prey attracted golden eagles and peregrine falcons to recently burned areas in New Mexico and southern California (Lehman and Allendorf 1989). Following stand-replacing fire in chaparral, common raptors and ravens were studied for an increase in numbers. Only ravens increased, probably because of increased scavenging opportunities (Wirtz 1979). In Great Basin and Chihuahuan Desert shrubsteppe, patchy burns probably favor species that require perches and cover above the ground (Bock and Bock 1990).

Mammals

Most mammals travel at least occasionally to seek food and shelter, and some make lengthy migrations every year. Mammal species can readily move into burned areas. Some use burned areas exclusively, and some use them seasonally or as part of their home range.

Reports of mammals moving into burned areas immediately after fire are mainly anecdotal. Lloyd (1938) describes movement of large animals into burned areas to seek protection from insects. In California chaparral, mountain lions are attracted to the edges of recent burns where deer tend to congregate (Quinn

Figure 7—Male black-backed woodpecker at nest hole in fire-killed lodgepole pine. Photo by Richard L. Hutto.

1990). Crowner and Barrett (1979) report red fox hunting in a recent burn in an Ohio field.

Many studies describe movement by large mammals to recently burned areas because of food quantity or quality. Courtney (1989) reports migration of pronghorn to a northern mixed prairie in Alberta 2 months after a July fire. The pronghorn fed on pricklypear cactus, which was succulent and singed, with many of the spines burned off. The following spring, pronghorn moved into the burn because vegetation there began growing approximately 3 weeks earlier than on unburned range. When the Delta caribou herd had its calves in Alaska in 1982, the caribou preferred a recently burned snow-free area to an unburned snow-free area and a snow-covered area (Davis and Valkenburg 1983). Seven months after a stand-replacing fire in boreal forest, northern Minnesota, yearling moose had moved into the burn, apparently attracted by increased forage and a low-density resident moose population (Peek 1972, 1974). Moose density increased from 0.5 per square mile a few months after the fire to more than 2 per square mile two growing seasons after the fire. Moose temporarily left the area during the winter, when the forage that had sprouted in response to fire was covered with snow (Peek 1972).

Large mammals may move into burned habitat simply because of familiarity with the area before fire. A behavioral study of Alaskan moose after stand-replacing and mixed-severity fire indicated that increased use of burned areas depended heavily on prefire travel patterns and awareness by the moose population of the area (Gasaway and others 1989).

Visibility of predators may be another reason for large ungulates to move into burned areas. Desert bighorn sheep abandoned areas from which fire was excluded (Etchberger 1990). Mazaika and others (1992) recommend prescribed burning in the Santa Catalina Mountains, Arizona, to clear large shrubs and maintain seasonal diet quality for bighorn sheep.

Most small mammals are able to migrate readily in response to increased food supplies, so many species repopulate burns quickly after fire. Removal of litter and standing dead vegetation, rather than increased growth of vegetation, seemed to attract deer mice to burned prairie within 5 weeks of a spring fire (Kaufman and others 1988a). Increased food availability apparently outweighed the increased danger of predation (Kaufman and others 1988b). After fire in Arizona chaparral, recolonization was “rapid” for the species that prefer grassy habitat, including voles, pocket mice, and harvest mice (Bock and Bock 1990).

Two landscape-related aspects of fire, size and homogeneity, influence colonization and populations of small mammals on recent burns. Research by Schwilk and Keeley (1998) showed a positive relationship between deer mouse abundance and distance from unburned edge, perhaps in response to food provided by postfire annual plants growing in the middle of burned areas. The fires, which burned in California chaparral and coastal sage scrub, left some “lightly burned” patches in canyon bottoms. These refugia may have enabled small mammals to colonize severely burned sites during the first 6 months after fire (Schwilk and Keeley 1998).

Reptiles and Amphibians

Little is known about amphibian and reptile emigration and immigration after fire. A study of low-severity prescribed fires in hardwood-pine stands of the South Carolina Piedmont found no evidence that herpetofauna emigrated in response to fire (Russell 1999). Western fence lizards in chaparral take refuge under surface objects at the time of fire; after the fire, they invade the burned site from unburned patches (Lillywhite and North 1974). Komarek (1969) reports seeing southern diamondback rattlesnakes sunning themselves in areas blackened by recent fire. Frequent lightning-season fires promote growth of the bunchgrasses that flatwoods salamanders seek out for laying their eggs. Fire exclusion reduces the grasses in favor of closed slash or pond pine forest (Carlile 1997).

Notes

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Chapter 4: Fire Effects on Animal Populations

The literature describing animals' behavioral responses to fire, discussed in chapter 3, is limited. Furthermore, short-term responses do not provide insights about the vigor or sustainability of the species in an area. Studies of animal populations and communities are more helpful in providing such insights. Most research regarding fire effects on fauna focuses on the population level, reporting changes in abundance and reproductive success of particular species following fire. Population changes are the net result of the behavioral and short-term responses discussed in chapter 3, plus longer term responses (years to decades).

Numerous population studies report abundance and density of animals in relation to fire, but information on productivity and other demographic factors may be essential for understanding population responses. Research on the threatened Florida scrub-jay provides an example. The scrub-jay requires scrub oak associations (myrtle, Chapman, and sand live oak, ericaceous shrubs, and saw palmetto), often in areas with open pine cover (less than 15 percent), where pine densities are kept low by frequent understory fires. The best vegetation for the jays consists of a mosaic of different age classes of scrub, most of which have burned within the last 20 years. Optimum scrub height is about 4.5 feet (1.5 m), interspersed with shorter scrub (Breininger and others in press; Woolfenden 1973). Without fire, the oaks become too tall and the habitat too dense for the Florida scrub-jay because predators are not easily seen (Breininger and others 1995). Florida scrub-jay densities in areas with tall shrubs are sometimes greater than in areas with optimum-height shrubs. However, jay mortality in tall scrub exceeds reproductive success; the jay is unable to sustain a population in tall scrub, as it can in shorter scrub (Breininger and others in press).

Changes in Animal Populations

Birds

Bird populations respond to changes in food, cover, and nesting habitat caused by fire. The season of burning is important to birds in two ways: Fires during the nesting season may reduce populations more than fires in other seasons; and migratory populations may be affected only indirectly, or not at all, by burns that occur before their arrival in spring or after their departure in fall.

Most raptor populations are unaffected or respond favorably to burned habitat. Fires often favor raptors by reducing hiding cover and exposing prey populations. When prey species increase in response to postfire increases in forage, raptors are also favored. Dodd (1988) describes beneficial effects from fire on populations of burrowing owl in desert grassland, sharp-shinned and Cooper's hawk in chaparral, and northern goshawk and sharp-shinned hawk in ponderosa pine forest.

Fire effects on insect- and plant-eating bird populations depend on alterations in food and cover. The canyon towhee, which eats insects and seeds, increased after stand-replacing fire in chaparral, foraging for food in the recent burn (McClure 1981). Wirtz (1977) reports that swallows, swifts, and flycatchers were more abundant over burned than unburned chaparral during the first postfire year. California gnatcatchers in coastal sage scrub, however, require the structure and cover provided by mature scrub. They avoid burns for the first 4 to 5 years after fire (Beyers and Wirtz 1997). In the northern Rocky Mountains, Hutto (1995) found 15 bird species more abundant in communities recently burned by stand-replacing fire than in other habitat; most were bark-probing insect eaters. On a site burned 19 years previously by stand-replacing fire in Olympic National Park, hummingbirds were probably more abundant than anywhere else in the area because the burn provided abundant nectar-producing forbs and shrubs and also open space for courtship (Huff and others 1985). After mixed-severity and stand-replacement burns in central Idaho, lazuli buntings and chipping sparrows, both seed eaters, were the most abundant songbirds (Saab and Dudley 1998). Fire in marshes usually increases areas of open water and enhances forage for shorebirds and waterfowl (Vogl 1967; Ward 1968).

Bird nest site selection, territory establishment, and nesting success can be affected by season of fire. Spring burns may destroy active nests (Ward 1968). Duck nesting success in mixed-grass prairie in North Dakota was significantly lower in areas burned in spring than fall (Higgins 1986). Blue-winged teal, northern shoveler, and American wigeon showed the lowest nesting success on spring burns. The differences were short-lived, however. Duck nesting response to fall- and spring-burned areas was similar in the third postfire year.

Nesting success also depends on the quality of the habitat before fire. Most birds nesting in areas burned by stand-replacing fire in the northern Rocky Mountains used broken-topped snags that were present before the fire (Hutto 1995). Many species of woodpeckers show substantial population increases and disperse into areas burned by stand-replacing fire (Hejl and McFadzen 1998; Hutto 1995; Saab and Dudley 1998). After mixed-severity and stand-replacement burns in central Idaho, nest abundance for nine cavity-nesting species increased through postfire year 4. On burned, unlogged sites, all species had nesting success above 50 percent, and three Forest Service-sensitive species had 100 percent success (table 1) (Saab and Dudley 1998).

Ground-dwelling bird populations are likely to be affected by fires of any severity, whereas canopy-dwelling populations may not be affected by understory fire.

Table 1—Success of cavity-nesting species after stand-replacing and mixed-severity fires in ponderosa pine/Douglas-fir forest in central Idaho (Saab and Dudley 1998).

Species	No. nests/km surveyed in 1996, all treatments*	Nesting success, unlogged stands	Characteristics of preferred nesting habitat
		Percent	
Lewis' woodpecker**	0.70	100	Highest nesting success on standard logged sites, selected the largest, most heavily decayed snags
Hairy woodpecker	0.58	92	Highest nesting success on unlogged sites
Northern flicker	0.40	75	Highest nesting success on wildlife logged sites, selected the largest, most heavily decayed snags
Western bluebird	0.63	60	Highest nesting success on wildlife logged sites
Mountain bluebird	0.64	56	Highest nesting success on unlogged sites
American kestrel	0.29	not reported	Nested mainly on standard logged sites, selected heavily decayed snags
European starling	0.13	100	
White-headed woodpecker**	0.03	100	Selected heavily decayed snags
Black-backed woodpecker**	0.10	100	Favor unlogged sites, locations with high tree density, selected hard snags

* 1996 was postfire year 2 for sites in mixed-severity burn, postfire year 4 for sites in stand replacing burn. Three treatments were studied: standard salvage logged; wildlife logged (approximately 50 percent salvaged logged); and unlogged.

** Species listed as sensitive by Forest Service in Regions 1, 2, 4, or 6.

After a fall fire on prairie in Saskatchewan, populations of ground-dwelling birds declined significantly. Savannah sparrows and clay-colored sparrows, the two most common species, were both adversely affected by the burn. These species rely on shrubs, specifically western snowberry and silverberry, for nesting habitat (Pylypec 1991). The year after fire, the abundance of breeding pairs in the burn was less than half the abundance in unburned areas. The third postfire year, savannah sparrows had recovered to a breeding pair abundance 68 percent of that on unburned sites, but clay-colored sparrow abundance had not changed substantially.

Woodpeckers generally nest in snags or in the forest canopy. Reports indicate that populations of woodpecker using forests with understory fire regimes tend to be unaffected by underburns. Thinning from below, designed to emulate understory fire in reducing fuels in an old-growth forest in Oregon, did not alter use of the site by pileated woodpeckers or Vaux's swifts, another bird that uses the tree canopy in old-growth forests (Bull and others 1995). Pileated woodpeckers' ability to use underburned sites probably depends on fire severity. Fires that reduce logs, stumps, and snags could have adverse effects by decreasing insect availability. The endangered red-cockaded woodpecker inhabits open longleaf, loblolly and shortleaf pine forests with few hardwoods in the midstory. Winter and growing season understory fires every 2 to 5 years are essential for retarding the development of a hardwood midstory in red-cockaded woodpecker habitat (Carlile 1997; U.S. Department of the Interior, Fish and Wildlife Service 1985) (fig. 8). If a hardwood midstory does develop, the woodpecker abandons its territory (Loeb and others 1992). The most abundant red-cockaded woodpecker populations now occur in areas with a long history of aggressive prescribed burning (Costa and Escano 1989).

Bird populations may exhibit some plasticity in relation to postfire habitat use and nest site selection. Brewer's sparrows and sage sparrows have been described as specifically requiring large patches of dense sagebrush (Knick and Rotenberry 1995; Wiens and Rotenberry 1981), but evidence from burned areas suggests some adaptability. The Brewer's sparrow population declined after fire in big sagebrush in Idaho; however, this decline was neither severe nor long-lived (Petersen and Best 1987). Return rates of banded male Brewer's and male and female sage sparrows the first 4 years after fire did not differ between burned and unburned areas, except the second year after fire when fewer male sage sparrows returned to the burn. The burn may have benefited the sage sparrow population indirectly, since new males used the burn to establish their territories. Nest placement by Brewer's sparrow was examined in big sagebrush rangeland before and after a prescribed fire in southeastern Idaho (Winter and Best 1985). Before the burn, all nests were located in sagebrush canopies. The prescribed fire burned about 65 percent of the vegetation, leaving a mosaic of burned and unburned sagebrush. After fire, there was a significant shift in nest placement: 21 percent were placed close to the ground. Fire may have reduced the number of tall shrubs, influencing some sparrows to nest beneath shrubs to obtain cover and concealment. Eastern kingbird populations in Michigan forests show an adaptable response to stand-replacing fire. In undisturbed riparian areas, eastern kingbirds nest in woody vegetation, which provides foliage for concealment, but they also nest successfully in the charred trunks and branches of burned jack pines (Hamas 1983). Several nests occurred in cupped depressions formed by embers that burned into heartwood.

Figure 8—Prescribed fire to improve red-cockaded woodpecker habitat. Fire is backing past a cavity tree on the Osceola National Forest, Florida. Photo by Dale Wade.

Fires influence bird populations indirectly by altering the populations of associated invertebrate species. Chigger infestation in the bird community increased in a chaparral stand during the months following stand-replacing fire. Feather mites were reduced, perhaps because silicon dioxide in the ash killed the mites (McClure 1981).

Mammals

Most of the literature describing fire effects on small mammal populations is from studies of stand-replacement and mixed-severity fire. Like birds, mammals respond directly to fire-caused changes in cover and food. Spring fires may impact mammal populations more than fires in other seasons because of limited mobility of young. The species with the most vulnerable young are small mammals, most of which also have high reproductive rates; if postfire habitat provides food and shelter for them, their populations recover rapidly.

Ream (1981) summarized information in 237 references about small mammals and fire. She concluded that populations of ground squirrels, pocket gophers, and deer mice generally increase after stand-replacing fire. Kaufman and others (1982) also report that the deer mouse population increased after fire. They found more deer mice on 1- and 2-year-old burns in tallgrass prairie than in unburned areas. In the same study, western harvest mice were more abundant on unburned sites. One year after stand-replacing fire in shrub-steppe habitat in Idaho, the total number of small mammals was lower in burned plots than in unburned plots (Groves and Steenhof 1988), and most of the animals in the burn were deer mice.

Rabbits, showshoe hare, red squirrel, northern flying squirrel, and voles generally avoid recent stand-replacement burns, according to Ream (1981). Shrews avoid burned areas from which most of the litter and duff have been removed. Of 25 animal populations common in chaparral brushlands, two were more abundant in mature, closed chaparral than in recently burned sites: Townsend's chipmunk and dusky-footed woodrat (Biswell 1989). Northern red-backed voles avoided a stand-replacement burn in black spruce for 1 year and finally established a resident population in postfire year 4, coinciding with the first year of berry production in the burn (West 1982). In the first year after stand-replacing fire in California grassland and chaparral, populations of agile kangaroo rat, California pocket mouse, deer mouse, and California mouse were either unchanged or greater on burned than unburned areas. Populations of brush mouse, western harvest mouse, and woodrat species decreased or disappeared in burned chaparral and grasslands (Wirtz 1977). Mixed-severity fire had little impact on populations of small mammals in pitch pine forests of the southern Appalachians (Ford and others 1999).

Animals that are dormant or estivating in underground burrows during and immediately after fire are particularly well protected from direct fire effects. Populations of Townsend's ground squirrels, dormant below ground at the time of stand-replacing fire in a sagebrush-grass community in southeastern Washington, seemed unaffected by the fire (Hedlund and Rickard 1981). Research after a stand-replacing fire in chaparral found that the only burrowing rodents, Heerman and agile kangaroo rats, were also the only rodents to survive in substantial numbers, probably because their burrows protected them from heat (Quinn 1979).

Population responses of small mammals to fire are related to fire uniformity. Most reports of woodrat responses to fire indicate that they usually suffer relatively high mortality because their nests are above ground (Simons 1991). However, populations of woodrats were "unexpectedly high" in burned areas observed by Schwilk and Keeley (1998). These burns left patches of "lightly burned" vegetation in California chaparral and coastal sage scrub, which may have provided refugia for woodrat populations.

Ungulate species often benefit from increased food and nutrition on recent burns. Because ungulates are sensitive to alterations in vegetation structure, however, their net response to fire depends on its severity and uniformity. In Lava Beds National Monument, northern California, mule deer populations were little affected by fire; home ranges were neither abandoned nor extended as a result of burning (Purcell and others 1984). Mule deer populations in chaparral burned by stand-replacing fire often increase, benefiting from increased availability of browse. Mule deer density in climax chaparral was estimated at 25 per square mile, while density in a severely burned area was 56 per square mile (Ashcraft 1979). Fawn production the second spring after burning was 1.15 fawns per doe compared to 0.7 fawns per doe in climax chaparral. Biswell (1961) reported an even more dramatic increase: deer density in chamise chaparral rose from 30 deer per square mile in unburned brush to 120 deer per square mile the first year after stand-replacing fire. Density decreased each year after that until it reached preburn levels in 5 to 12 years. In contrast, Stager and Klebenow (1987) report that mule deer preferred pinyon-juniper stands 24 and 115 years after stand-replacing fire to recently burned stands. The difference may be attributable to the drier conditions in pinyon-juniper, which slow vegetation recovery from fire.

Most other large ungulates either respond neutrally or positively to postfire changes in habitat. Elk rely on browse in seral shrub fields during winter and use dense, pole-sized forest heavily in fall (Irwin and Peek 1983). Moose also rely on seral shrubs in many areas, especially where shrubfields are interspersed with closed-canopy forest. In two areas converted from sagebrush dominance to grassland with shrub patches, pronghorn were present after fire but not before; they had been absent from one site for 60 years prior to the burn (Deming 1963; Yoakum 1980). Bison may avoid burned areas for a short time, until regrowth of forage begins (Moore 1972). Several studies indicate that bison prefer foraging in recently burned areas the summer after fire (Boyce and Merrill 1991; Shaw and Carter 1990; Vinton and others 1993) (fig. 9). White-tailed deer prefer to browse on recent burns if cover is close by. Management recommendations for white-tailed deer for specific geographic regions often list a maximum opening size or minimum distance to cover (for example, see Ivey and Causey 1984; Keay and Peek 1980).

Large carnivores and omnivores are opportunistic species with large home ranges. Their populations change little in response to fire, but they tend to thrive in areas where their preferred prey or forage is most plentiful—often, in recent burns. Fire has been recommended for improving black bear (Landers 1987) and grizzly bear habitat (Hamer 1995; Morgan and others 1994) (fig. 10). In Minnesota, enough early postfire plant communities must exist within a gray wolf pack's territory to support a surplus of deer, moose, and American beaver for prey (Heinselman 1973).

American beaver populations invade streamside habitat where fire has stimulated regrowth of aspen or willow species (Kelleyhouse 1979; Ream 1981). Burned areas in New York had more beaver colonies and a higher annual occupancy than unburned areas (Prachar and others 1988).

Fire may indirectly reduce disease rates in large mammal populations. Following a stand-replacing fire in spruce-lodgepole pine and bunchgrass mosaic in Glacier National Park, Montana, bighorn sheep tended to disperse, which may have reduced lung-worm infections in the population (Peek and others 1985).

Reptiles and Amphibians

Fire-caused changes in plant species composition and habitat structure influence reptile and amphibian populations (Means and Campbell 1981; Russell and others 1999). In chaparral, reptiles were more abundant in recently burned areas than in areas with mature, dense cover. Individual populations responded to the developing structure of the vegetation (Simovich 1979). Species that preferred open sites increased slightly during the first 3 years after fire. During the same time, species that used or could tolerate dense

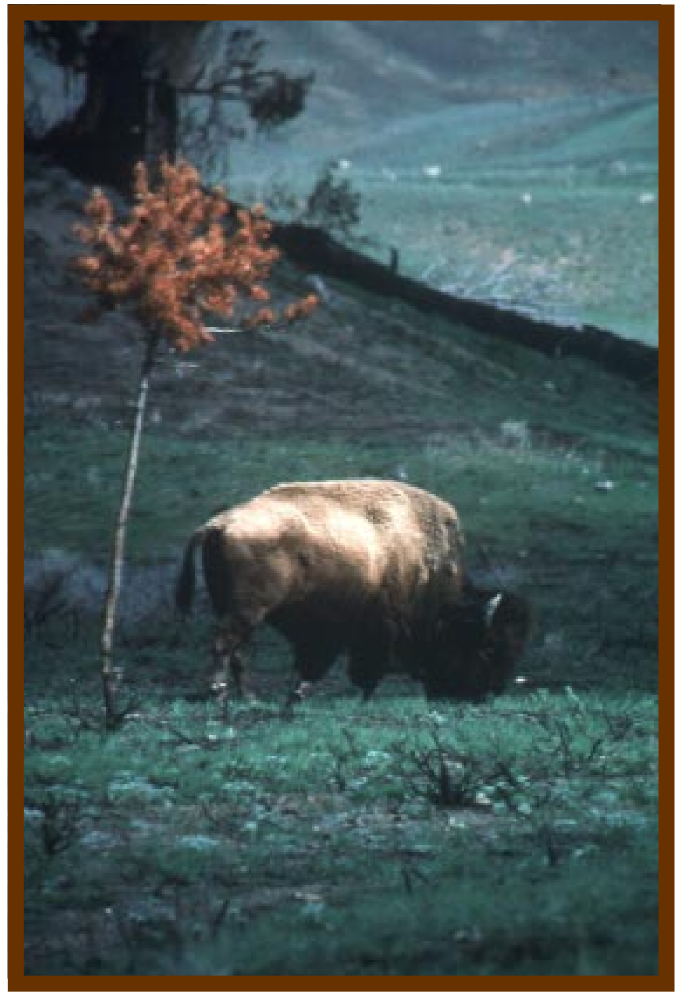

Figure 9—Bison grazing in area converted by stand-replacing fire from shrub-dominated to forb- and grass-dominated cover. Photo by Jim Peaco, courtesy of National Park Service.

vegetation decreased but were not eliminated. As the chaparral becomes a dense, mature layer, reptile abundance is likely to decrease.

Amphibians in forested areas are closely tied to debris quantities—the litter and woody material that accumulate slowly in the decades and centuries after stand-replacing fire. In forests of British Columbia, the proportion of nonmammalian vertebrates (mainly amphibians) using woody debris was positively correlated with the length of the fire rotation (Bunnell 1995).

Many herpetofauna populations show little response to understory and mixed-severity fire. After mixed-severity fire in pitch pine stands in the southern Appalachian Mountains, populations of woodland salamanders were generally unchanged (Ford and others 1999). Low-intensity underburns in hardwood-pine stands of the South Carolina Piedmont did not significantly alter species richness of herpetofauna;

Figure 10—Grizzly bears foraging in lodgepole pine regeneration following stand-replacing fire, Yellowstone National Park. Photo by Jim Peaco, courtesy of National Park Service.

amphibians were significantly more abundant on burned plots due to greater numbers of Fowler's toad and red-spotted newts (Russell 1999). Although the slash pine habitat of the flatwoods salamander in Florida was underburned during winter, its breeding season, the population showed no sign of decline (Means and Campbell 1981).

In longleaf pine forests and slash pine plantations in the Florida sandhills, the threatened gopher tortoise (fig. 11) requires a sparse tree canopy and open, grassy ground cover for optimum food and nesting (Carlile 1997; Means and Campbell 1981), conditions that are provided by understory burning. Fires during the growing season may increase nest sites and enhance food supplies for new hatchlings (Carlile 1997). More than 300 other species use the gopher tortoise's burrow, including numerous arthropods, reptiles, and amphibians, so fire effects on the tortoise impact many other populations in the faunal community (Carlile 1997; Means and Campbell 1981; Russell and others 1999; Witz and Wilson 1991).

A review by Russell and others (1999) explains that fire in isolated wetlands usually increases areas of open water and enhances vegetation structure favored by many aquatic and semiaquatic herpetofauna.

Invertebrates

At least 40 species of arthropods are attracted to fires (Evans 1971), alerted by stimuli including heat, smoke, and increased levels of carbon dioxide. Many use burned trees for breeding. When the larvae hatch, they feed on the burned trees.

Soil protects most soil macrofauna and pupae of many insects from fire. The level of protection depends on depth of the organism and depth of heat penetration, which in turn depend on duff consumption (Schmid and others 1981). Insect abundance above ground decreases immediately after fire in prairies but then increases as fresh, young plant growth becomes available (Robbins and Myers 1992).

Figure 11—Two gopher tortoises graze on new grass shoots after a prescribed fire. Photo by Larry Landers.

Effects of Altered Fire Regimes

Understory Fire Regimes

Exclusion of fire can cause changes in faunal abundance and community composition in forests adapted to understory fire, but studies designed to examine the long-term effects of fire exclusion are rare. In the Southeast, the red-cockaded woodpecker requires longleaf pine habitat with an open midstory, maintained in past centuries by frequent understory fire. When a dense hardwood midstory develops due to fire exclusion, the woodpecker abandons its territory (Loeb and others 1992). Bird populations were monitored for 15 years in a loblolly and shortleaf pine stand in northwest Florida, comparing a site underburned annually to one from which fire had been excluded (Engstrom and others 1984). After 15 years of fire exclusion, the unburned plot had 20 times more trees and less than one-third the ground cover of the annually burned plot. In the fire-excluded stand, the bird community changed continuously in response to structural changes. Species that require open habitat disappeared within 5 years of fire exclusion. During years 3 to 7, another group of species reached maximum numbers (common yellowthroat, indigo bunting, eastern towhee, white-eyed vireo, and northern cardinal). After saplings began to mature in the understory, species associated with mesic woods were observed. Populations of canopy-dwelling birds such as the eastern wood-pewee, great crested flycatcher, blue jay, and summer tanager were affected little by 15 years of succession.

Saab and Dudley (1998) hypothesize the effects of three future fire regimes on ponderosa pine-Douglas-fir forests with presettlement fire return intervals of 5 to 22 years (table 2). High intensity, stand-replacing fires would favor seven of the 11 cavity nesting bird species studied and would negatively affect four species. Continued fire suppression, accompanied by increasing forest density, would favor only two species. The third possibility discussed is a combination of silvicultural treatment and prescribed fire, which theoretically would favor eight species and negatively affect only two species. The table offers a framework for testing whether management to replace a presettlement regime of frequent understory fire with a combination of thinning and management-ignited understory fire can produce benefits similar to those from presettlement fire regimes to the species listed. To assess potential changes throughout the faunal community, such a table would need to include at least all indicator species and species of special concern.

Table 2—Predicted responses by cavity-nesting birds to three possible fire regimes compared with the presettlement low intensity, high frequency fire regime in Idaho ponderosa pine/ Douglas-fir forests (Saab and Dudley 1998), presented as a framework of hypotheses to be tested. + = more favorable than presettlement regime, 0 = no different, - = less favorable.

	Potential new fire regime		
Bird species	**High intensity stand-replacement fire**	**Complete fire suppression**	**Prescribed fire with stand management**
American kestrel	+	-	+
Lewis' woodpecker	+	-	+
Red-naped sapsucker	-	0	+
Downy woodpecker	-	0	+
Hairy woodpecker	+	0	+
Black-backed woodpecker	+	-	0
White-headed woodpecker	-	-	+
Northern flicker	+	+	-
Pileated woodpecker	-	+	-
Western bluebird	+	-	+
Mountain bluebird	+	-	+

Stand-Replacement Fire Regimes

Fire exclusion from areas with stand-replacing fire regimes has contributed to loss of habitat and population declines in several raptor and predator species. Examples include the golden eagle in the Appalachian Mountains (Spofford 1971) and short-eared owl along the eastern grassland-forest interface (Lehman and Allendorf 1989). A review by Nichols and Menke (1984) explains that several raptors (red-tailed hawks, Cooper's hawks, sharp-shinned hawks, and great horned owls) are more abundant in recently burned chaparral than in unburned areas due to greater visibility of prey.

Frequent, stand-replacing fire in presettlement times maintained a "virtually treeless landscape" on the Great Plains (Bidwell 1994). Fire exclusion, tree planting, and enhancement of waterways have encouraged woodlands to develop, fragmenting the prairies. The greater prairie-chicken, Henslow's sparrow, and upland sandpiper all decline where habitat is fragmented (Bidwell 1994). To increase abundance of the greater prairie-chicken in North Dakota, Kobriger and others (1988) recommend use of prescribed burning. To maintain nest and brood habitat for the prairie-chicken, Kirsch (1974) recommends burning large plots (at least 0.5 mile, 800 m, across) at 3- to 5-year intervals. Grasslands left undisturbed for more than 10 years are not desirable.

Prairie dog colonies once covered hundreds of thousands of acres of the Great Plains that burned frequently (Bidwell 1994). Prairie dogs prefer burned to unburned areas for feeding and establishing colonies (Bone and Klukas 1990). The prairie dog is essential prey for the black-footed ferret, and prairie dog colonies provide for needs of more than 100 other vertebrate species in some way (Scott 1996; Sharps and Uresk 1990). Prairie dog grazing and waste alter the soil and vegetation near colonies, favoring early successional forb species, stimulating growth of grass and forbs, and increasing the nitrogen content of forage (Bidwell 1994; Sharps and Uresk 1990). Improved grass forage attracts bison, and increased forb cover attracts pronghorn. Bison, in turn, trample the areas where they graze (Yoakum and others 1996). Bison impact on prairie dog colonies is reduced when recent burns are available for grazing (Coppock and others 1983).

Invasion by nonnative annual plants has increased fire frequency in many semidesert ecosystems that were characterized by stand-replacement fire regimes in presettlement times. Exotic annuals, particularly cheatgrass in sagebrush ecosystems, increase fuel load and continuity. The result is increased fire frequency, followed by greater area of bare soil that is colonized by greater numbers of exotic annuals (U.S. Department of the Interior 1996; Whisenant 1990). The impact of exotic annuals is exacerbated in sagebrush ecosystems because fire exclusion and overgrazing since the mid-1800s increased sagebrush dominance at the expense of native herbaceous species. Loss of sagebrush cover and disruption of the historic balance of shrubs, native grasses, and forbs threatens the viability of sage grouse, sage sparrow, Brewer's sparrow, and sage thrasher populations (Knick and Rotenberry 1995; Sveum and others 1998). In the Snake River Birds of Prey National Conservation Area, big sagebrush has declined from more than 80 percent cover in the 1800s to less than 15 percent in 1996 (U.S. Department of the Interior 1996). Areas that have burned in the last 15 years have less than 3 percent sagebrush cover. Models predict complete loss of shrublands in 25 to 50 years without fire suppression in cheatgrass areas. Loss of sagebrush is contributing to a steady decline in black-tailed jackrabbit populations and increased fluctuations in Townsend's ground squirrel populations. Prairie falcons and golden eagles rely on these two prey species, so increased fire frequency is reducing the density and reproductive success of both species (Wicklow-Howard 1989; U.S. Department of the Interior 1996). Other animals in Idaho that prey on the Townsend's ground squirrel—red-tailed hawks, American badgers, western rattlesnakes, and gopher snakes—may also be affected (Yensen and others 1992).

Lodgepole pine and aspen communities in the Western States provide two examples of effects of fire exclusion on forests with stand-replacement fire regimes. In lodgepole pine-spruce-fir forests, the most productive period for bird communities appears to be the first 30 postfire years. Thirteen species regularly breed only in the first 30 years after fire. Conversely, just two species breed exclusively in forests more than 30 years old (Taylor 1969, 1979; Taylor and Barmore 1980). Species that breed exclusively in the first 30 years after fire may be difficult to maintain in the ecosystem without fire. Fire exclusion and postfire salvage of dead trees after fire may reduce populations of these species over large geographic areas.

Aspen stands provide more forage and a greater diversity of understory plants than the spruce and fir communities that generally replace them in the absence of fire. Fires of moderate to high severity can regenerate aspen, but the moderate to high intensity fire necessary to stimulate vigorous suckering of aspen is often difficult to achieve (Brown and DeByle 1982; Severson and Rinne 1990).

DeMaynadier and Hunter's (1995) review points out that most research on effects of fire on amphibians and reptiles has been done in Florida. They and other authors (Russell and others 1999) caution against extending the results of this research to ecosystems where frequent fire was not part of the presettlement disturbance regime.

Mixed-Severity Fire Regimes

Not enough information is available to generalize about effects of changing fire regimes in areas with presettlement patterns of mixed-severity fire. Exclusion of fire from mixed-conifer and Douglas-fir forests in the Southwest has led to increased fuel loads and increasing risk of large, uniformly severe fire (Fiedler and Cully 1995; Lehman and Allendorf 1989; U.S. Department of the Interior, Fish and Wildlife Service 1995). Severe fire is likely to destroy nesting trees and the dense forest structure required by Mexican and California spotted owls. Prescribed understory fire has been recommended to reduce fuels in areas near spotted owl nest trees and to break up fuel continuity in large areas of continuous dense forest, reducing the likelihood of large, stand-replacing fires in the future (Fiedler and Cully 1995; U.S. Department of the Interior, Fish and Wildlife Service 1995; Weatherspoon and others 1992).

Grazing and fire exclusion have converted some desert grasslands to open woodlands. This constitutes loss of habitat for species such as pronghorn and Ord's kangaroo rat but increases habitat for mule deer (Longland 1995; MacPhee 1991).

Animal Influences on Postfire Habitat

Most of the literature regarding the relationship between fire and fauna focuses on fire-caused changes in vegetation and how habitat changes influence animal populations. A related topic is the effect of animal populations on the process of postfire succession. In this brief section, we provide a few examples of such relationships for animals and plants native to North America.

The jay-sized Clark's nutcracker (fig. 12) is responsible for most whitebark pine regeneration (Tomback 1986). The nutcracker prefers to cache seed in open sites with highly visible landmarks, conditions available within recent burns (Murray and others 1997) (fig. 13). Tomback and others (1996) studied whitebark pine regeneration after the 1988 fires in the Greater Yellowstone area. Areas burned by mixed-severity

Figure 12—Clark's nutcrackers cache seed of whitebark pines. Unrecovered seed from these caches accounts for most whitebark pine regeneration. Photo by Bob Keane.

Figure 13—Whitebark pine regeneration in an area burned by stand-replacing fire 30 years previous to the photo. Photo by Stephen F. Arno.

and stand-replacement fire had greater whitebark pine regeneration than did unburned sites.

Bison not only prefer burned to unburned grassland for grazing during the growing season, they also contribute to the pattern of burning in prairie. In tallgrass prairie in northeastern Kansas, bison selected patches with low forb cover dominated by big bluestem, and grazed larger patches in burned than unburned habitat (Vinton and others 1993). Ungrazed forbs in areas adjacent to heavily grazed patches were thriving, producing greater biomass than in larger, ungrazed portions of the study area. The increased variability in vegetation productivity may act as feedback to fire behavior, increasing variation in patchiness and variable severity of subsequent fires. During the centuries before European American settlement, bison populations may have been controlled by Native American hunting, which would have reduced the effects of grazing on fuel continuity (Kay 1998).

Kangaroo rats and pocket mice may enhance postfire dominance of Indian ricegrass in sagebrush grassland ecosystems. These rodent species gather and hoard large numbers of seed, with a clear preference for Indian ricegrass. On burned sites with abundant populations of these rodents, Indian ricegrass seed had been deposited in scatter-hoards before fire even though the species was not dominant. Indian ricegrass dominated soon after fire. Six years after fire, density of Indian ricegrass was more than tenfold greater on burned than unburned sites (Longland 1994, 1995).

Although fire causes high mortality for antelope bitterbrush, it also creates litter-free sites, in which bitterbrush germination rates are high. Most antelope bitterbrush seedlings originate in rodent seed caches, and rodents apparently retrieve fewer seed from sites with limited cover (such as burned areas) than from sites with better protection (Bedunah and others 1995; Evans and others 1983).

Grazing and browsing on postfire sites, whether by wild ungulates or domestic grazers, can alter postfire succession. For example, if aspen is treated by fire to regenerate the stand but then repeatedly browsed by wildlife, it often deteriorates more rapidly than without treatment (Bartos 1998; Basile 1979). Such intense effects of feeding by large ungulates only occur where the animal populations are food limited. Where Native American predation kept populations of these animals in check, such effects are unlikely (Kay 1998).

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Chapter 5: Fire Effects on Animal Communities

Many animal-fire studies depict a "reorganization" of animal communities resulting from fire, with increases in some species accompanied by decreases in others. Descriptions of faunal communities after fire, however, are much less prevalent than descriptions of population changes. The literature about fire and bird communities is more complete than the literature about other kinds of animals. In this chapter, we use the literature about fire and birds to search for response patterns in the relationship between fire regime and changes in bird community composition. The literature does not at this time provide enough studies of mammal communities to complete a similar analysis.

Each animal species in a community is likely to respond differently to fire and subsequent habitat changes. To synthesize information about these responses, we modified Rowe's (1983) classification of plant responses to fit animal responses to fire. Rowe's approach was to assign to each plant species an adaptation category based on reproduction and regeneration attributes in the context of fire. Using similar categories in our evaluation of the animal-fire literature, we classified species' responses (*not* species themselves) for a given study location using observed changes in animal abundance. Mean changes in species abundance before and the first few years after fire, or in burned versus unburned areas, can be classified into one of six categories (table 3). Possible community response patterns using these six categories include:

A. **Increasers predominate**: A high proportion of invader and/or exploiter responses. This pattern represents an upward shift in abundance, especially for opportunistic species.
B. **Decreasers predominate**: A high proportion of avoider and/or endurer responses. This pattern represents a downward shift in abundance and unsuitable or poor habitat conditions for species established prior to the burn.
C. **Most populations change**: An equitably high proportion of invader and/or exploiter responses and of avoider and/or endurer responses. This pattern represents a small change in total abundance but a large shift in abundance of many individual species.
D. **Few populations change**: A high proportion of resister responses and a low proportion of other responses. This pattern represents little change in species composition and relatively minor fire effects on the animal community.
E. **Intermediate change**: A high proportion of resister, endurer, and exploiter responses; low proportion of invader and avoider responses.

Table 3—Classification of changes in bird abundance into response categories.

Response category	Before fire	After fire
Invader	Not detected	Detected (minimum number)
Exploiter	Detected	>50% increase
Resister	Detected	≤50% increase or decrease
Endurer	Detected	>50% decrease
Avoider	Detected	Not detected or very low numbers
Vacillator	Detected/not detected	Inconsistent, wide fluctuations

This chapter presents bird community responses to fire according to the fire regimes as described in chapter 1: understory, stand-replacement, and mixed-severity. Understory fire regimes occur only in forest cover types. Stand-replacing fire regimes are divided according to vegetation type: grassland, shrub-grassland, shrubland, and forest. Finally, we discuss mixed-severity fires (also limited to forest types) that leave at least 40 percent cover of large trees.

Analysis of the literature using the framework described above shows that fire effects on bird communities are related to the amount of structural change in vegetation. In burned grasslands, bird communities tend to return to prefire structure and composition by postfire year 3. Postfire shrub communities are generally in flux until the shrub canopy is reestablished, often 20 years or more after fire. In forests, understory fire usually disrupts the bird community for 1 year or less. Stand-replacing fire generally alters bird communities for 30 years or more. However, variation is great. Many bird communities conform only loosely to this pattern.

Many studies of fire effects on bird communities report species richness or other indices of diversity. Conserving all species is obviously essential for sustaining ecosystem patterns and processes, but *maximizing* diversity in a given location does not necessarily sustain the ecosystem (Telfer 1993). Bird responses to fire in Southeastern scrub communities provide an example. Many bird species (for instance, the Carolina wren and northern cardinal) are negatively affected by regimes of frequent fire in these scrub communities. Increasing fire frequency may reduce these species, thus reducing species richness. But the populations reduced by frequent fire represent forest edge species common in Eastern North America. In contrast, increasing fire frequency favors the threatened Florida scrub-jay and other scrub specialists, which have a narrow geographic range and are the species that make Florida scrub habitat unique (Breininger and others in press). Their habitat is declining because fire frequencies have declined, and these changes have long-lasting effects on habitat structure even when fires later return to the system (Duncan and others 1999).

Understory Fire Regimes

Understory fires burn beneath the tree canopy, mostly through surface and understory fuels. Prescribed understory burns are commonly used to reduce fuel hazards and maintain open forest structure in areas that had high-frequency, low-intensity fire regimes in presettlement times, such as southeastern pines and ponderosa pine (see Biswell 1989). Understory fires often disrupt the bird community during the first postfire year, but by postfire year 2, underburned forests are generally returning to preburn bird community structure and composition.

The time since burn and the interval between understory fires influence fire effects on bird populations. In oak scrub and slash pine communities along the central east coast of Florida, for example, Carolina wren and white-eyed vireo had highest densities in areas that had not burned for more than 10 years. Common yellowthroat and rufous-sided towhee preferred areas burned 4 years previously, and few shrub-dwelling birds used understory burns less than 2 years old (Breininger and Smith 1992). Positive correlations between densities of shrub-dwelling birds and mean shrub height suggest that some shrub dwellers would decline under a regime of fire every 7 years or less. However, this decline would not be expected if some patches of habitat remained unburned (Breininger and Schmalzer 1990). Much scrub occurs as patchy mosaic within other vegetation types that have a greater propensity to burn, so burns are naturally patchy.

Frequent Understory Fires

Understory fires occurring at short (5- to 10-year) intervals usually cause minor changes to vegetation composition and structure and likewise to bird communities. Several studies have shown that many bird species resist changes in abundance in frequently underburned forests. Emlen (1970) reports few changes in the bird community during the first 5 months after understory burning in a 20-year-old slash pine forest in Everglades National Park. The fire removed most of the ground cover, dead grass, and litter; defoliated most shrubs; and scorched small trees. Trees in the middle and upper canopy were undamaged. Grass and herbs recovered quickly. Over 70 percent of bird species responses were classified as resister, showing little or no change in abundance. No species showed invader responses after the fire. In southeast Arizona ponderosa pine stands, moderately intense prescribed understory fires (with flame lengths up to 4 feet, 1.2 m) consumed nearly half of all snags more than 6 inches (15 cm) dbh, resulting in a net 45 percent decrease in potential nest trees the first year after treatment (Horton and Mannan 1988). Cavity-nesting bird species abundances changed little, however. In contrast to the above studies, a review by Finch and others (1997) reports considerable community change after "cool" understory burns in ponderosa pine. Seed eaters, timber drilling birds, and some aerial insect eaters increased, while timber and foliage gleaners generally decreased.

In the first 2 years after "cool" prescribed understory fires in the Black Hills, the bird community showed mainly resister and exploiter responses (Bock and Bock 1983). Bird abundance in postfire year 1 was nearly twice that in the unburned area, yet in postfire year 2 abundances were similar in burned and unburned areas. Such rapid shifts could not be explained by changes in vegetation structure or composition. Most likely, temporary, rapid increases in food resources attracted bird species to burned areas and resulted in a quick surge in their abundances.

The severity of understory fire affects the composition and abundance of the bird community after fire. In loblolly pine-bottomland hardwood forests of Alabama's Piedmont, high-intensity understory fire removed vegetation from the middle of the canopy down, while low-intensity understory fires had a "patchy" effect, with live and dead understory vegetation interspersed. Significantly more birds used the low-intensity burns than the high-intensity burns in the 4 months after treatment (Barron 1992). Bark, canopy, and shrub gleaners were more abundant on the low-intensity burn, while ground foragers were more abundant on the high-intensity burn.

Infrequent Understory Fires

More substantial changes in forest structure and the bird community may occur after fire in areas with infrequent understory fire (intervals greater than 10 years). Populations of the most common breeding birds decreased after severe understory fires in Yosemite National Park, while less common species increased substantially (Granholm 1982). Two understory fires were examined: a prescribed fire in white fir-mixed conifer forest and a naturally ignited understory fire in a California red fir forest. In presettlement times, these forest types underburned every 17 to 65 years (Taylor and Halpern 1991). Trees up to 40 feet (12 m) tall were killed by the fires. Bird communities in the two burns showed similar responses. The highest proportion of species responses was in the resister category. No species avoided the burns, and more than 70 percent of the responses were classified as resister, exploiter, or invader. Hermit thrush and Hammond's flycatcher populations were reduced most by the fires, and woodpecker populations increased most.

Vegetation usually responds more slowly after fire in dry forests, including pinyon-juniper, than in more productive, frequently burned forests. The bird community may likewise be slow to return to its prefire composition and structure. In pinyon-juniper forests of Nevada, understory fires occurred in the past much less frequently (once every several decades) than in the southern pines and ponderosa pine (Wright and Bailey 1982). In central Nevada, more than 60 percent of bird species the first 2 years after prescribed understory fire showed vacillator (showing wide population fluctuations) or exploiter responses (Mason 1981). Species using resources near the ground increased most after burning. Savannah and black-chinned sparrows were found only on burned areas.

Stand-Replacement Fire Regimes

Research in the literature indicates that bird communities are disrupted for at least 2 years by stand-replacing fire. A few studies show signs that the community is returning to its preburn structure in postfire years 3 and 4, but others do not. The changes can be positive for insect-eating and seed-eating species and negative for species that require a dense, closed canopy such as bark and foliage gleaners.

Grasslands

Grasslands with few or no shrubs have a relatively simple aboveground vegetation structure, which is consumed almost completely by fire. Vegetation change following fire is rapid. Conditions similar to preburn vegetation composition and structure reestablish by postfire year 2 or 3 (for example, see Launchbaugh 1972). Although grasses dominate the vegetation, forbs often increase in density and cover immediately after fire, so plant diversity may be highest within the first 2 years after fire. Bird species that nest and use grasslands seem to be well adapted to rapid, predictable changes in habitat characteristics associated with fires, even though such fires often remove avian nest substrates and hiding cover.

Bird communities in a South Dakota prairie 2 to 3 months after fire showed dramatic population changes, with a high proportion of invader, endurer, and avoider responses (Huber and Steuter 1984). This was the only grassland study that showed such a high proportion of invader responses, which may be due to the short duration of the study and the fact it was conducted soon after fire. Upland sandpiper and western meadowlark showed substantial increases compared to unburned areas, while grasshopper sparrow and red-winged blackbird had much lower abundances on the burn.

Other research on postfire bird communities was done over longer periods than the above study. During the first 2 years after grassland fires in southeastern Arizona, most bird populations changed, but few species abandoned or were completely new to the area (Bock and Bock 1978). Nearly 75 percent of the species responses were classified as vacillator, endurer, and exploiter.

In Saskatchewan, the bird community also changed in the first 2 years after grassland fire (Zimmerman 1992). More than half of the bird populations showed resister responses. No responses were classified as avoider, and only a few responses were invader and exploiter. Abundance of key species such as clay-colored and savannah sparrows were still substantially below the unburned levels in year 3, so overall abundance was consistently lower in the burned area. Recovery was slower than in other grasslands studied. The cool climate and short growing season of Saskatchewan may slow the recovery process for some prairie species.

The same bird species may respond differently to fire in different habitats. For example, field sparrows in central Illinois prefer to breed in grasslands overgrown with shrubs and young deciduous trees (shrub-grassland), but they also breed in grasslands without brush and in open woodlands (Best 1979). After burning, field sparrows used shrub-grassland more and burned grassland less than they had during the same period the previous year. Thus the response of field sparrow populations in grasslands was endurer, and the response in shrub-grasslands was exploiter. Fire evidently caused field sparrows to use the preferred habitat more intensively than the less-preferred habitat.

Climatic interactions with fire and habitat suitability are not well understood, but adaptation to periodic drought may be essential for a bird species to persist in grass-dominated communities (Zimmerman 1992). In average and wet years, food resources increased in Kansas prairie after fire, yet bird abundance did not. This indicated that the bird community was saturated (Zimmerman 1992). When drought and fire overlapped and resources were reduced, even drought-adapted species decreased in abundance, although no species disappeared from the community.

Shrub-Grasslands

We differentiate between grasslands and shrub-grasslands because grass-dominated areas with shrubs have more complex habitat structure than grasslands. The only shrub-grasslands discussed here are those in which shrubs were present before fire or in unburned areas used as controls. Shrub-grasslands are likely to have more niches available to birds and to recover their preburn structure more slowly after fire than grasslands. The two 1-year studies examined here indicate that annual burning causes substantial changes in bird communities in shrub-grasslands.

Annual burning of a Kansas prairie for more than 10 years led to a significant decrease in bird species richness. Annual burning maintained the prairie with low coverage of woody vegetation, rendering it unsuitable for woody-dependent core species and most other species. Among the bird species present every year, response to fire was almost 90 percent resister, endurer, and avoider (Zimmerman 1992). Annual burning virtually eliminated habitat characteristics needed by Henslow's sparrow and common yellowthroat.

Most species abundances changed in response to fire on a southwestern Florida dry shrub-grassland. In the first postfire year, most species responses on burned plots (with shrub cover ranging from 34 to 82 percent) were invader and avoider, compared to plots without fire for more than 15 years that had a closed shrub canopy (Fitzgerald and Tanner 1992). Species showing an invader response were mostly ground feeders (for example, Bachman's sparrow and common ground-dove), whereas shrub-dwelling species showed the avoider response (for example, northern cardinal and gray catbird). Burned plots provided better avian habitat than mechanically treated plots (in which shrubs were chopped). Birds colonized the burned plots much sooner than the mechanically treated plots. Shrubs killed by fire provided a more complex habitat structure than shrubs in the mechanical treatment. Annual burning would ultimately exclude shrubs, so the bird community response would probably resemble that after mechanical treatment.

The importance of shrubs as perches in shrub-grasslands is illustrated by a study in Kansas tallgrass prairie (Knodel-Montz 1981). Forty artificial perches were placed in burned and unburned prairie. Comparisons were made among plots annually burned and unburned, with and without artificial perches. Artificial perches on the burn were used nearly twice as often as those on the unburned plot, although the difference was not statistically significant. In the unburned plot, birds seemed to prefer natural perches to artificial ones.

Shrublands

Shrublands usually occur in dry environments and are characterized by sparse to dense shrubs with few or no trees. Examples are the extensive sagebrush lands of the Interior West and California chaparral. Fuels in shrublands tend to burn rapidly. Fires typically move swiftly and are difficult to control. Most aboveground vegetation is consumed by fire, so the structure of the vegetation is altered dramatically. Recovery time ranges from years to decades, depending largely on the resprouting ability of the species burned. Bird populations often decline after shrubland fires, but declines may be offset by populations that rebound if fire spread is patchy, leaving some areas unburned, and if species usually associated with grassland communities invade the burn.

Numbers of bird species and individuals were much lower where fire burned a California coastal sage scrub community dominated by California sagebrush than in an unburned area nearby (Stanton 1986). This was most noticeable the first 18 months after the fire (Moriarty and others 1985). The fire killed all but a few large shrubs and trees. Among the 37 bird species observed by Stanton (excluding raptors), more than 70 percent responded as resister and endurer, and few as avoider or invader. Significant differences in foraging activity between seasons and between burned and unburned areas were observed. All birds except the flycatchers spent more time actively foraging in the unburned than in the burned area. Permanent residents foraged more in the burned area during spring and early summer than during the rest of the year. Birds tended to perch rather than forage in the burned area.

Lower bird populations also predominated after a fire in big sagebrush in south-central Montana that killed nearly 100 percent of the sagebrush (Bock and Bock 1987). In postfire year 3, grass and herb cover were much higher on the burn than in a similar unburned area, but no recruitment of new sagebrush was detected. Of the few species detected, responses were mostly avoider, plus either endurer or resister. Lark sparrow, lark bunting, and Brewer's sparrow all avoided the burn. During the breeding season, these three species occupied patches of significantly more shrub canopy in the unburned area than available randomly. Grasshopper sparrows were classified as endurer. The only resister response was by the western meadowlark, an adaptive grassland bird.

Because sagebrush does not sprout from underground buds after fire, sagebrush communities require several decades to establish postfire vegetation composition and structure similar to that on unburned sites. Incomplete burning, characteristic of sagebrush stands, appears to be important to the development of these communities. Unburned islands of sagebrush are important sources of sagebrush seed after fire and retain habitat features vital to species associated with shrubs, such as sage grouse and Brewer's sparrow.

In southeastern Idaho, more than 50 percent of the species responses were classified as resister for a postburn bird community in big sagebrush (Petersen and Best 1987), where the prescribed fire was incomplete, killing about 50 percent of the shrubs. The first year after fire, total bird abundance declined significantly (22 percent). In years 2 and 3, there were no significant abundance differences between burned and unburned areas. In year 4, significantly more birds were detected on the burn. Species showing resister responses may have used different parts of the patchy postfire habitat. No species avoided the burn during the 4 years of the study.

Nonuniform burning was used to explain bird community changes after a fire in sagebrush in north-central Utah; 3 and 4 years after the fire, few bird species showed appreciable declines. Bird abundance in the burned area (with about 90 percent of aboveground vegetation burned and 80 percent of shrubs killed) was compared to abundance in an unburned site plowed 17 years before the study (Castrale 1982). Total bird density and number of breeding species were similar on the two sites. Breeding bird responses 3 to 4 years after fire were primarily exploiter and resister, with no avoider or invader responses. Burning was associated with increases of western meadowlark, a grassland species. Brewer's sparrow and sage thrasher, which nest above the ground in shrubs, were associated with unburned islands of sagebrush and did not use the grassy portions of the burned site. If the fire had killed all the shrubs, Brewer's sparrow, which can be eliminated from sagebrush habitats with chemical control of shrubs (Schroeder and Sturges 1975), probably would have been absent.

Completeness of burn influenced fire effects in central Florida's oak scrub (Breininger and Schmalzer 1990). During the winter and spring following a November burn, stations with more than 95 percent of the vegetation burned had low numbers of permanent residents, while stations with 10 to 25 percent of the vegetation burned had bird counts similar to unburned stations.

The nature of habitat adjacent to burned shrublands sometimes influences bird community responses. Lawrence (1966) sampled bird communities in interior California chaparral dominated by buckbrush before a prescribed fire and 3 years afterward. Observation transects crossed chaparral, grassland, and pine-oak woodland. Most chaparral species responses were classified as resister and endurer, with no avoider or exploiter responses. California quail and scrub jay declined sharply after fire.

Research in California shrublands indicates that fire does not reduce species diversity but does alter species composition. During the year following a stand-replacing fire in coastal sage scrub, southern California, the species richness of birds in the burned area gradually increased. By the end of the first year, species richness on the burn was 70 to 90 percent similar to that on an adjacent unburned area (Moriarty and others 1985). The species most abundant in the burn were those typically associated with open areas, whereas the species most abundant in unburned areas typically avoid open areas.

Forests and Woodlands

Stand-replacing fires in forests and woodlands are either severe surface fires or crown fires; more than 80 percent of the trees are top-killed or killed. The contrasts between prefire and postfire environments are much sharper than after understory fire, and the time needed for the vegetation to develop structure and composition resembling the preburn forest is measured in decades to centuries. During this time, many forces can alter the trajectory of succession, so the mature forest may differ substantially from the preburn forest. A stand-replacing fire is likely to result in many or most of the bird species present before fire being replaced by new species (Finch and others 1997). Some species use habitat that occurs only for a short time after stand-replacing fire. In Yellowstone and Grand Teton National Parks, more species were unique to the postfire period (1 to 17 years) than to later stages of succession (111 to 304 years after stand-replacing fire) (Taylor and Barmore 1980).

In this section we first describe bird response to fire in the short term (less than 5 years after fire) and then in the long-term (5 years or longer). Short-term studies typically included control plots, either sampled before the fire or after the fire in a similar, unburned area. Long-term studies covered early to late stages of vegetation succession. Some examined succession from 6 to 60 years after fire, when canopy closure occurred. Others examined a chronosequence of similar sites at different locations from early to late seral conditions.

Short term—The few studies available indicate that changes in habitat characteristics caused by stand-replacing fire cause postfire avian communities to differ substantially in the short term from prefire communities. High turnover occurs in the first 5 years after stand-replacing fire. The predominant response categories are invader and avoider. These responses usually describe 50 to 90 percent of postfire bird populations. Few species responses are classified as resister after crown fire, often less than 20 percent of the species present in the first 2 years postfire; some studies show no resister responses to fire. This community response to fire differs substantially from the response generally observed in understory fire regime types, where a high proportion of the postfire bird community consists of resister species. Most studies of understory fire regimes showed at least a third of the species responses as resister, with some over 70 percent.

In western hemlock forests of western Washington, which has a stand-replacing fire return interval spanning several centuries, more than half the bird populations showed invader and avoider responses during the first 2 years after a severe crown fire. The bird community composition shifted from domination by canopy-dwelling species to species nesting and foraging near the ground (Huff and others 1985).

Bird community response to stand-replacing fire in ponderosa pine forests of Arizona was similar to that in western hemlock forests (Lowe and others 1978), even though the climate and presettlement fire regimes of the two communities differ. Nearly 60 percent of the species responses were classified as invader and avoider 1 year after fire.

Substantial species turnover also characterized a dense 200-year-old spruce-fir-lodgepole pine forest in Grand Teton National Park, Wyoming, which burned in stand-replacing fire. More than 80 percent of bird population responses were avoider and invader during the first 3 years postfire (Taylor and Barmore 1980). Few species showed resister responses. As in western Washington, a shift in the bird community from canopy dwellers to ground/brush dwellers occurred. Patterns observed nearby in Yellowstone National Park were similar (Pfister 1980). In 250-year-old lodgepole pine-spruce-fir forest, about three-fourths of the bird community responses were classified as invader in years 2 to 3 after stand-replacing fire. The increased bird diversity in comparison with unburned forest was associated with rapid changes in forest structure and composition after the fire, which attracted several species uncharacteristic of the unburned forest.

A shift from canopy dwelling to ground- and shrub-dwelling species also occurred after stand-replacing fire in northern Minnesota. Apfelbaum and Haney (1981) sampled birds before and after crown fire in a 73-year-old jack pine/black spruce forest. The fire burned severely in an upland pine-dominated area while only lightly burning the hardwood draws. The number of breeding territories decreased by more than half the first year after fire. Tree canopy dwellers were most abundant before the fire, while ground- and shrub-dwelling species predominated afterward. The bird community showed high species turnover; 70 percent of species responses were avoider and invader. The black-backed woodpecker was an important species showing the invader response, comprising about 13 percent of total bird abundance after the fire. Ovenbird, the most important ground- and brush-dwelling species prior to fire, avoided the burned area, where the moss ground cover was replaced by lush herbs and jack pine seedlings.

Long term—Oliver and others (1998) show how a "landscape" disturbance is likely to affect bird abundance in three groups of species: those that reside in structurally complex old-growth stands with abundant understory, those that prefer edges between dense and open vegetation, and those that prefer open habitat (fig. 14). The diagram reflects some patterns reported in long-term studies of birds in forested ecosystems, although it does not account for the complex role of fire in producing and destroying snags (see "Snags and Dead Wood" in chapter 1). The predictions are in agreement with Finch and others' (1997) review of the general pattern of species change in southwestern ponderosa pine forests, whether burned by understory or stand-replacing fire: Granivores, tree drilling birds, and some aerial insectivores usually increase after fires, while tree- and foliage-gleaning birds generally decrease. Birds more closely tied to foliage availability, such as hermit thrush and blue-headed vireo, begin recovering as foliage volume increases in subsequent years. Finch and others (1997) add that woodpecker abundance often peaks in the first decade after fire, then gradually declines.

Figure 14 depicts a period early in succession after stand-replacement fire when birds are abundant, and also a time of transition when dominance by open- and edge-using species gives way to dominance by understory- and canopy-using species. Three studies of bird community dynamics after stand-replacing fire provide some insight regarding the species and habitat requirements that account for these changes. Research on bird community response to succession in the long-term requires either commitment to a long-term research program or use of a chronosequence, a series of sites similar in all characteristics except time since fire. The former method was used for a study in the California Sierra Nevada (Bock and Lynch 1970; Bock and others 1978; Raphael and others 1987). The two other studies discussed here are based on chronosequences. These three studies indicate that (1) early seral conditions foster high bird diversity, (2) more bird species breed exclusively in early seral stages than in mature forests, and (3) snags are a key habitat feature for avian diversity and abundance.

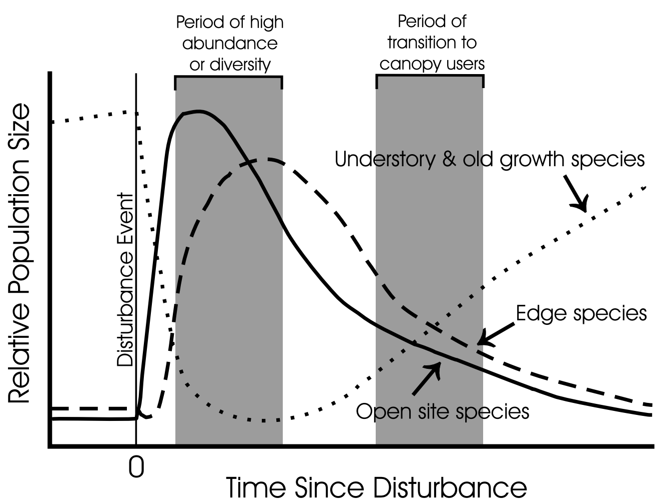

Figure 14—Hypothetical patterns of change in populations of species dependent on three features of forest structure: dense understory/old growth, edge, and open sites. Shaded areas are discussed in the text. Adapted from Oliver and others (1998).

In the Sierra Nevada study, burned and unburned plots were established in 1966, 6 years after a large (approximately 37,000 acres, 15,000 ha) stand-replacing fire in a mixed-conifer forest dominated by Jeffrey pine and white fir. The fire killed nearly all the overstory and understory trees, although small pockets of trees were alive in postfire year 6. Birds were sampled every year except one for the next 20 years.

Changes in the avian community in the burn were primarily related to changes in vegetation structure with succession (Raphael and others 1987). In postfire years 6 to 8, bird abundance on burned plots was similar to that on unburned plots, but species composition differed. Species nesting and foraging on living trees were most abundant on unburned plots, while species characteristic of low brush and open ground predominated in the burned area (Bock and Lynch 1970). Primary cavity excavators (woodpeckers) were more abundant on the burn; even higher numbers may have been present during the 6 years before the study was initiated. Of 32 regularly breeding species, 28 percent were unique to the burn, while 19 percent were unique to the unburned area.

Bird diversity decreased in the burn from postfire years 8 to 15, to less than in the unburned area (Bock and others 1978). By postfire year 15, fewer dead snags were standing and the ground cover was more dense, resembling a shrubfield. Six of 11 hole-nesting species declined during this period. Species that require some open ground, such as dark-eyed juncos—the most abundant breeding species in postfire year 8—were replaced by species indicative of shrubfields, including fox sparrows.

Shrub cover doubled between postfire years 8 and 23, snag density declined 90 percent, and cover of herb and grass-like species decreased significantly. At the end of this period, large snags (more than 15 inches, 38 cm dbh) were 2.5 times more abundant in the unburned area than in the burn. Birds that feed and nest in shrubs increased in abundance by more than 500 percent. Woodpeckers declined steadily. At the end of the 25 years, woodpecker abundance on the burn was similar to that in the unburned area. At postfire year 25, vegetation characteristic of a closed-canopy forest still had not developed in the burn. The transition from open- to closed-site species (postulated in fig. 14) was beginning, but the bird community was likely to continue changing and not likely to closely resemble either the unburned area or the burn anytime soon.

Western hemlock forests of different ages (times since stand-replacing fire) were sampled in western Washington (Huff 1984; Huff and others 1985). The ages of stands in this chronosequence were 1 to 3 years, 19, 110, 181, and 515 years. Year 19 of the sere had the highest bird diversity and least resembled other successional stages examined. Lowest bird diversity and abundance occurred at the 110-year-old site where, comparatively, the tree vertical structure was simple, snag density low, and understory composition and structure poorly developed. This stand age may represent the transition from open to closed structure depicted in figure 14. Huff and others (1985) note that rate of forest reestablishment may be slower in western hemlock forests than in the Jeffrey pine-white fir forests of the Sierra Nevada (described above). If so, a longer period of high diversity associated with early seral conditions can be expected for the western Washington sere. Once a full canopy develops in the western hemlock sere, few changes occur in bird species composition. Because the fire return interval is long, species composition may change relatively little for centuries.

A large-scale examination of avian successional relationships after crown fire was conducted in Yellowstone and Grand Teton National Parks, Wyoming, by comparing recent burns to older burns and to areas unburned for at least 300 years (Taylor 1969, 1979; Taylor and Barmore 1980). The most obvious changes in species composition the first few years after fire were surges in abundance of black-backed and northern three-toed woodpeckers. (Prior to the 1974 Waterfall Canyon fire, the black-backed woodpecker was not even known to occur in the Grand Tetons.) Breeding bird density in postfire years 5 to 29 was more than 50 percent greater than in lodgepole pine stands more than 40 years old with closed canopy. In postfire years 5 to 25, following an influx of cavity excavators, the number of secondary cavity nesters increased rapidly. Two of these species, the tree swallow and the mountain bluebird, dominated the avifauna. They consistently comprised 30 percent or more of postfire birds during the first 30 years after fire. In the second decade after fire, they comprised 55 to 64 percent of the total bird population. By about postfire year 30, mountain bluebirds and tree swallows started to decline at a rate that depended on the loss of standing snags with nest cavities. During this period, vegetation structure and succession made a transition from shrubland to young forest.

The most important event in succession for the postfire bird community was the transition from open to closed canopy, which occurred between postfire years 30 and 50. With the onset of this event, species abundance decreased by more than 60 percent. Species characteristic of later seral stages gradually appeared as the trees got taller. From about postfire year 50 to year 100, change in forest composition and structure stagnated. Over the next 200 years, lodgepole pine in the canopy gave way to shade-tolerant spruce and fir. The bird community changed little during this 250 years, with bird abundance lower than that in earlier successional stages. Bird density and diversity in 300-year-old and older spruce-fir forest is higher than in the previous 250 years.

Mixed-Severity Fire Regimes

Little is known about the effects of fire on bird populations in mixed-severity fire regimes. One might expect the bird community response to mixed-severity fire to be intermediate between responses to understory and stand-replacement fire. Both mixed-severity and stand-replacement fire occurred in Grand Teton National Park, Wyoming, in a 250 year old spruce-fir forest (Taylor and Barmore 1980). Half the species responses were invader and exploiter for the first 3 years after fire. Some canopy-dwelling species typical of unburned areas occurred in the mixed-severity burn but were absent from the stand-replacing burn. These included western tanager, golden-crowned kinglet, red-breasted nuthatch, and yellow-rumped warbler. The mixed-severity burn had less species turnover than the stand-replacement burn in the first 2 years postfire. Almost half the species responses to the stand-replacement fires were avoider, yet no avoider responses were recorded in the mixed-severity burns.

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Chapter 6: Fire Effects on Fauna at Landscape Scales

Studies of disturbance and succession have been a major focus of ecology over the past century (McIntosh 1985). These are studies of temporal pattern. The study of the spatial patterns associated with temporal patterns has blossomed only since about 1990 (Turner 1990). At the landscape scale (25,000 acres, 10,000 ha or more), a complex web of interactions and relationships unfolds. Interactions at this scale are widely accepted as important aspects of ecosystems (see, for example, Agee 1998; Lerzman and others 1998). However, knowledge gained at finer scales of resolution (for example, stand or homogeneous patch) is often difficult to apply at a landscape scale (Schmoldt and others 1999). Including landscape considerations in management demands new approaches to planning, analysis, and design (Diaz and Apostol 1992).

Landscapes are spatially heterogeneous, characterized by structure, function, and temporal variation (Forman and Godron 1986). Landscape structure encompasses the spatial characteristics of biotic and abiotic components in an area and is described by the arrangement, size, shape, number, and kind of patches (homogeneous units). Landscape function is defined by interactions among biotic and abiotic components. Temporal variation of a landscape is expressed by changes in structure and function over time. Configuration of patches affects the occurrence and spread of subsequent fires, so landscape-level feedback is an important part of fire effects at landscape scales (Agee 1998).

Fire's most obvious function in landscapes is to create and maintain a mosaic of different kinds of vegetation (Mushinsky and Gibson 1991). This includes size, composition, and structure of patches, as well as connectivity (linkages and flows) among patches. Within a large (200 mi^2, 500 km^2) burn in Alaska, Gasaway and DuBois (1985) reported substantial variation in fire severity and many unburned patches, resulting in variation in plant mortality and perpetuation of the mosaic nature of the landscape. Over time, a mixture of a few large burns with many small burns and variation within them produces relatively small homogeneous areas. One study in northern Manitoba reported an average stand size of 10 acres (4 ha) (Miller 1976 in Telfer 1993). Stand-replacing fires in boreal forest may skip as much as 15 to 20 percent of the area within their perimeters. The 1988 fires in the Greater Yellowstone Area, well publicized because of their size and severe fire behavior, actually consisted of a complex patchwork containing areas burned by crown fire, areas burned by

Figure 15—Aerial photo shows variation in fire severity over the landscape after the 1988 fires in the Greater Yellowstone Ecosystem. Black patches were burned by crown fire. Most of these are surrounded by red and gray areas where trees were killed by severe surface fire. Green cover represents a combination of unburned forest and areas burned by understory fire. Photo by Jim Peaco, courtesy of National Park Service.

severe surface fire, underburned sites, and unburned areas (Rothermel and others 1994) (fig. 15, table 4). The majority of severely burned area was within 650 feet (200 m) of unburned or "lightly burned" areas.

Landscape-scale fire effects on fauna include (1) changes in availability of habitat patches and heterogeneity within them, (2) changes in the composition and structure of larger areas, such as watersheds, which provide the spatial context for habitat patches, and (3) changes in connections among habitat patches. During the course of postfire succession, all three of these landscape features are in flux.

Fire changes the proportions and arrangement of habitat patches on the landscape. When fire increases heterogeneity on the landscape, animal species have increased opportunities to select from a variety of habitat conditions and successional stages. Fires often burn with varying severity, increasing heterogeneity. Adjacent unburned areas (which may surround or be embedded in the burn) serve as both sinks and sources for animal populations, and also influence animal emigration and immigration patterns (see Pulliam 1988). Bird diversity after stand-replacing fire may be higher on patchy or small burns than on large, uniform burns because the small areas are accessible to canopy and edge species as well as species that use open areas. A small (300 acre, 122 ha) stand-replacing fire in Douglas-fir forest in western Montana

Table 4—Proportion of area burned at four severities within the perimeter burned each day in the Greater Yellowstone Area, 1988 (Turner and others 1994).

Severity level	Percent of area burned daily	
	June 1-July 31	Aug. 20-Sept. 15
Unburned	29.3	28.2
Underburned, "light" burn	18.9	14.5
Severe (stand-replacing) surface fire	26.6	24.4
Crown fire	25.1	32.8

left many unburned patches. The burn attracted wood-boring insects, woodpeckers, and warblers. The burn itself was not used by Swainson's thrushes, but they remained abundant in nearby unburned areas (Lyon and Marzluff 1985).

Two management examples show how understanding of the relationship of individual species to landscape heterogeneity can be applied. The Karner blue butterfly (fig. 16) requires wild lupine, a forb growing in fire-dependent oak savanna and prairie, to complete its life cycle. The larva itself (fig. 17), however, is very sensitive to fire. To protect the butterfly at Indiana Dunes National Lakeshore, managers divide the landscape so that every burn area contains patches from which fire is excluded; these patches serve as refugia from which the butterfly can repopulate the burn (Kwilosz and Knutson 1999).

The sage grouse is sensitive to fire effects on the arrangement of habitat components on the landscape. Stand-replacing fire in sagebrush changes the proportions and arrangement of sage grouse habitat components. It is this arrangement that determines whether fire benefits or damages the species. Sage grouse use various successional stages of the sagebrush sere as lekking, nesting, brooding, and wintering grounds. Forb and insect availability are the driving factors in sage grouse productivity (Drut and others 1994). Fires increase openings, which often increases forb production. Fires may also enhance the nutritional value of browse and provide new lekking sites (Benson and others 1991; Martin 1990; Pyle and Crawford 1996). If burns cover large tracts of sagebrush or remove sagebrush from key wintering areas, however, they may damage sage grouse populations (Fischer and others 1996; Gregg and others 1994; Klebenow 1969, 1973; Welch and others 1990). Neither extensive dense sagebrush nor extensive open areas constitute optimal habitat for the species. While burning sometimes succeeds in restoring the balance of plant community components in sage grouse habitat, it is accompanied by the risk of increasing cheatgrass productivity, which may cause the area to reburn before sagebrush recovers (Crawford 1999).

Researchers in many ecosystems recommend addressing the size and spatial arrangement of patches in planning for specific objectives. In Southeastern forests, Dunaway (1976) recommends interspersion of underburned areas in longleaf pine, which have low ground cover and provide successful foraging for northern bobwhite chicks, with unburned areas for

Figure 16—Karner blue butterfly, an endangered species whose larval form feeds exclusively on a the fire-dependent wild lupine. Photo by Robert Carr, courtesy of the Michigan Chapter, The Nature Conservancy.

Figure 17—Karner blue butterfly larva feeding on its sole food source, the fire-dependent wild lupine. Ants protect the larvae from predation and feed on "honeydew," a high-sugar liquid exuded by the larvae. Photo by Catherine Papp Herms, courtesy of the Michigan Chapter, The Nature Conservancy.

escape cover and sheltering broods. In the Western States, Belsky (1996) suggests that a mosaic of pinyon-juniper woodland, grassland, and intermediate seral communities would optimize biodiversity in arid Western ecosystems. In the Southwest, Reynolds and others (1992) quantify the proportions of a landscape in ponderosa pine forests, characterized in presettlement times by understory fire regimes, that seem desirable for sustaining northern goshawk populations (fig. 18). Recommendations for sustaining habitat and prey for the northern goshawk in Utah and the Rocky Mountains include increasing the predominance of early-seral and midseral species, increasing the numbers of large trees in the landscape, and maintaining connectivity among habitat patches (Graham and others 1997, 1999).

Some animals require habitat that contains different features at different scales. Wright (1996) found many patches of old-growth ponderosa pine and Douglas-fir in western Montana that seemed suitable for occupation by flammulated owls, but the owls occupied fewer than half of them. The explanation lies in the landscape context for the patches of old growth. Occupied patches (fig. 19) were embedded in a landscape with many grassy openings and some dense thickets of Douglas-fir; unoccupied patches (fig. 20) were typically embedded in a landscape of closed, mature forest. Understory fire may enhance old growth for nesting, openings for foraging, and the landscape context for nest sites. However, a homogeneous underburned landscape without Douglas-fir thickets would reduce the quality of habitat for the owl.

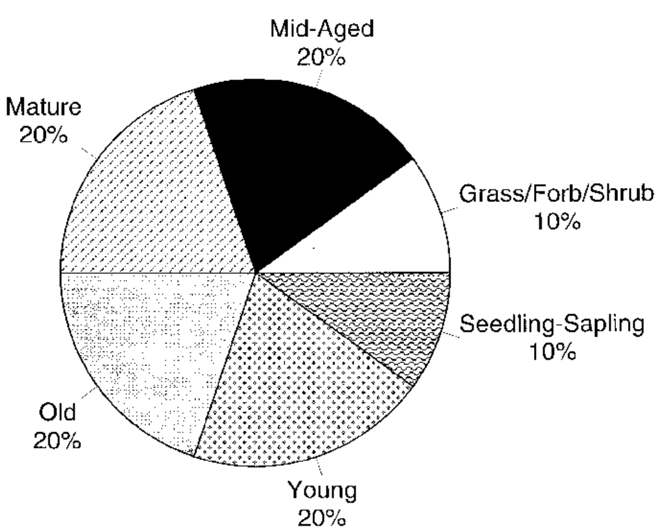

Figure 18—Proportions of a home range or landscape desirable for sustaining goshawks in forests with understory fire regimes (Reynolds and others 1992), example of a set of recommended stand descriptors to be implemented at landscape scales.

Nesting patterns of the Florida scrub-jay provide a second example of habitat requirements that vary according to scale. Optimum habitat for Florida scrub-jays consists of open oak patches 10 years or more after fire containing openings that often result from more recent fires. Scrub-jays use these openings for caching acorns (Breininger and others 1995). The oak patches preferred for nesting occur within a matrix of pine-scrub habitat, which is not used directly by the jay but indirectly serves its needs by providing prey species, enabling jays to see predatory birds from a long distance, and spreading fires into oak-dominated areas, which often burn poorly. Management that favors open oak without considering the more flammable adjacent habitat can result in a loss of openings and an increase in shrub height and tree densities, and eventually a Florida scrub-jay decline (Breininger and others 1995).

Corridors and connectivity influence habitat use by migratory fauna such as bison (Campbell and Hinkes 1983) and caribou (Thomas and others 1995), and for many predators, including fisher (Powell and Zielinski 1994), lynx (Koehler and Aubry 1994) and spotted owl (Laymon 1985; Thomas and others 1990). Connectivity is a crucial consideration for aquatic fauna as well (Rieman and others 1997). Although research design considerations may make it difficult to demonstrate conclusively that wildlife corridors benefit fauna (Beier and Noss 1998), some species definitely require landscapes with little fragmentation and high connectivity (Bunnell 1995). Bighorn sheep in Alberta foraged in spruce-pine forest burned 10 years previously by stand-replacing fire significantly more than in unburned areas, probably because the burned sites had a more open structure adjacent to escape areas (Bentz and Woodard 1988). Connectivity accounted in part for the expansion of a bison herd in Alaska after fire. Stand-replacing fire in black spruce forest produced extensive sedge-grasslands, a type that bison depend upon for winter range (Campbell and Hinkes 1983). The authors comment that winter range expansion was enhanced because the burned area was contiguous with summer range and areas used for winter range prior to the fire, so access to burned range was relatively easy. Where black spruce and shrublands fragment sedge-grasslands, bison have difficulty accessing their winter range because of deep snowpack.

In landscapes that contained a fine-grained mosaic of structures and age classes during presettlement times, native fauna could readily find most kinds of habitat. In contrast, in landscapes where large, stand-replacing fires were common, fauna sometimes traveled great distances in search of habitat. Ecosystems with large, stand-replacing fire regimes and

Figure 19—Not all habitat that seems suitable for flammulated owls at the stand level is occupied by the owl; suitability at the landscape is also important. This photo depicts a typical landscape in western Montana where flammulated owls *were* detected. Photo by Vita Wright.

Figure 20—Typical landscape where flammulated owls were *not* detected, western Montana. Photo by Vita Wright.

ecosystems that are now highly fragmented probably require more attention to connectivity than areas retaining a fine-grained mosaic. As fire and fire exclusion further alter landscapes, corridors, and entry/exit areas near corridors, fauna that require large, continuous areas of any structure—whether early seral or old growth—may not readily find new habitat.

Despite the tendency of natural fire regimes to provide habitat with a variety of structures at a variety of successional stages, one cannot assume that landscapes prior to European American settlement were at equilibrium, even at a landscape scale (Agee 1998). Ecosystems characterized in past centuries by infrequent large, severe fire are especially unlikely to exhibit a steady-state age structure because large fires have a long-term effect on the distribution of age classes on the landscape. Examples include aspen-black spruce forests in Alberta (Cumming and others 1996) and lodgepole pine in the Greater Yellowstone Area (Turner and others 1994). Presettlement fire regimes are an important part of the context for management. The spatial and temporal variability in these regimes, though difficult to identify and apply, may be a crucial aspect of effective management (Lertzman and others 1998).

Effects of Altered Fire Regimes

Excluding fire from a landscape, unless it is being intensively managed for fiber production, has two major effects on animal habitat. First, it increases the abundance and continuity of late-successional stages. Second, it changes fuel quantities and fuel arrangement, at least for a time.

Extensive changes in habitat associated with decades of fire exclusion are most evident in areas influenced by frequent fires during presettlement times (Gruell and others 1982). Understory fire regimes in southeastern forests provide one example. Fire exclusion increases dominance by less flammable vegetation, converting pine to hardwood forest (Engstrom and others 1984). Loss of early seral structures on sandy sites contributes to the decline of several reptile species, including sand skinks, six-lined racerunners, mole skinks, and central Florida crowned snakes (Russell and others 1999).

Kirtland's warbler/jack pine ecology in Michigan provides another example of fire exclusion's effects at landscape scales. Jack pine forests were characterized in presettlement times by relatively frequent, stand-replacement fire. The Kirtland's warbler (fig. 21)

Figure 21—Kirtland's warblers at nest. Photo by Betty Cotrille, courtesy of the Michigan Chapter, The Nature Conservancy.

nests on the ground in dense jack pine regeneration 5 to 24 years after stand-replacing fire or harvesting (Mayfield 1963; Probst and Weinrich 1993). The warbler was nearly extirpated during the 1960s and 1970s because of nest parasitism by the brown-headed cowbird, fire exclusion, and tree regeneration practices in jack pine forests of Michigan (Mayfield 1963, 1993). Extensive use of fire and harvesting to provide breeding habitat have kept the Kirtland's warbler from extinction, although uncertainty still exists about habitat attributes that actually limit population growth (nest sites, lower branch cover for fledglings, and foliage volume for foraging). Habitat modeling and management planning need to integrate habitat requirements, dynamics of disturbance and succession over large areas, and population dynamics of the warbler itself (Probst and Weinrich 1993).

In many Western ecosystems, landscape changes due to fire exclusion have changed fuel quantities and arrangement, increasing the likelihood of increased fire size and severity (Lehman and Allendorf 1989). In interior ponderosa pine/Douglas-fir forest, for example, exclusion of understory fire has led to the development of landscapes with extensive ladder fuels, nearly continuous thickets of dense tree regeneration, and large areas of late successional forest infested with root disease (Mutch and others 1993). These changes not only constitute habitat loss for species that require open old-growth stands and early seral stages; they also may increase the likelihood of large, severe fires in the future. Where fire exclusion has caused a shift in species composition and fuel arrays over large areas, subsequent fires without prior fuel modification are not likely to restore presettlement vegetation and habitat (Agee 1998).

The effects of fire exclusion on fauna that require late-seral and old-growth habitat originally established by fire are largely unknown. Although pileated woodpeckers do not nest in recent stand-replacement burns, they do prefer to nest in western larch, a fire dependent tree, in the Northern Rocky Mountains (McClelland 1977). If altered fire regimes reduce the abundance of large, old western larch, they are likely to impact the woodpecker as well. In presettlement times, the spotted owl occupied landscapes that consisted of large areas of forest at different stages of succession, characterized by Gaines and others (1997) as a "very dynamic" landscape. The owl prefers old-growth forest within this landscape, so fire exclusion has enhanced owl habitat, at least in some parts of the owl's range (Thomas and others 1990). Large, severe fires now would reduce the species' habitat and reduce connectivity between remaining old-growth stands (Thomas and others 1990). Protection of the owl may include fuel reduction in areas adjacent to occupied stands to reduce the likelihood of stand-replacement fire.

Several ecosystems in Western North America experience more frequent fire now than they did in the past because of invasive species. Where cheatgrass dominates areas formerly covered by large patches of sagebrush and grassland, for example, fires now occur almost annually and shrub cover is declining. Knick and Rotenberry (1995) report that site selection by sage sparrow, Brewer's sparrow, and sage thrasher is positively correlated with sagebrush cover. In addition, sage sparrow and sage thrasher prefer large to small patches of shrubs. The sage grouse requires mature sagebrush as part of its habitat, so extensive stand-replacing burns are likely to reduce its populations (Benson and others 1991). Increased fire frequency and cheatgrass cover have increased landscape-level heterogeneity by reducing sagebrush cover and patch size, lowering the value of even remnant sagebrush patches as habitat for native birds (Knick 1999).

Notes

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Chapter 7: Fire Effects on Wildlife Foods

Fire's influence on wildlife food is probably the most thoroughly researched aspect of the relationship between fire and fauna. It has generated a vast literature showing a great variety of results. The literature is not balanced among faunal classes or geographic regions, since most studies focus on a species of concern to managers in a particular geographic area. To summarize this literature, we return to the vegetation communities described in chapter 2, organizing the discussion according to the five geographic regions presented there. This structure also corresponds to the broad outline of "Effects of Fire on Flora" in the Rainbow Series.

For each vegetation type, we summarize what is known about changes in quantity and quality of forage following fire. In addition, where information is available, we summarize changes in availability of seeds, mast, and insects.

The factors that affect postfire changes in vegetation quantity and nutritional quality include soil, vegetation type, age and structure of vegetation prior to burning, rainfall before and after burning, severity of the fire, season of burning, time since fire, and presettlement disturbance regime. In general, the literature regarding fire effects on wildlife food indicate that:

- Burning sets back plant development and succession, often increasing or improving forage for wildlife from a few years to more than 100 years, depending on vegetation type.
- Fires usually increase habitat patchiness, providing wildlife with a diversity of vegetation conditions from which to select food and cover.
- The biomass of forage plants usually increases after burning in all but dry ecosystems.
- The production of seeds by grasses and legumes is usually enhanced by annual or biennial fires. Mast production is usually enhanced by a 5-year or longer burning cycle.
- Burning sometimes, but not always, increases the nutritional content and digestibility of plants. This effect is short-lived, typically lasting only one or two growing seasons.
- Some wildlife species select a more nutritious diet from burned areas even though the average nutrient content of burned plants does not differ from that of unburned plants.

It is impossible to generalize about fire effects on wildlife foods that apply throughout all of North America. Furthermore, while improving and increasing food for particular wildlife species may be an important objective, it is important that these goals not be accomplished at the expense of ecosystem sustainability (Provencher and others 1998).

Northern Ecosystems

Boreal Forest

Fire and the Quantity of Forage—During the first two growing seasons after a mixed-severity fire with large areas of stand-replacement in Minnesota boreal forest, most herbaceous and low shrub species increased rapidly in biomass. Production leveled off during the next 3 years (Irwin 1985; Ohmann and Grigal 1979).

Stand-replacing fire in boreal forest can greatly increase the production of woody browse for moose (MacCracken and Viereck 1990; Oldemeyer and others 1977; Wolff 1978). Prefire stand age and species composition play a significant role in plant response to fire (Auclair 1983; Furyaev and others 1983; MacCracken and Viereck 1990; Viereck 1983). Aspen stands that were 70 years old before stand-replacing fire produced 10 times as much browse in the first postburn year than did birch and spruce stands that were 180 years old before fire. Spruce stands that were 70 years old before fire produced three times as much browse after burning than did similar stands 180 years old before burning. The benefits of burning to moose may peak 20 to 25 years after stand-replacing fire (MacCracken and Viereck 1990; Oldemeyer and others 1977) and last less than 50 years (Schwartz and Franzmann 1989).

In boreal forests, stand-replacing fire reduces the lichens that caribou use as forage in winter; lichens may be reduced for up to 50 years after fire. Caribou prefer open forests burned 150 to 250 years ago. Their preference is related not only to abundance of food, but also to snow cover, visibility of predators and other herd members, and nearness to traditional travel routes (Thomas and others 1995). Lichens decline in old stands (200 years or more), indicating that fires of moderate to high severity may be essential for maintaining forage for caribou in the long-term (Auclair 1983; Klein 1982; Schaefer and Pruitt 1991).

Fire and Nutritional Quality—Stand-replacing fire in boreal forest increases the protein, phosphorus, calcium, magnesium, and potassium content of woody browse for moose, but this effect is probably gone by the third growing season (MacCracken and Viereck 1990; Oldemeyer and others 1977).

Laurentian Forest

Fire and the Quantity of Forage—Quaking aspen and paper birch, two of the most important browse plants for white-tailed deer and moose in the Northern and Eastern States, both sprout well after most fires. Paper birch reaches peak browse production 10 to 16 years after stand-replacing fire (Safford and others 1990). Twenty-five years after prescribed fire in quaking aspen in northern Minnesota, aspen productivity was 111 percent of productivity on unburned stands (Perala 1995).

Fire and Nutritional Quality—Low-intensity understory fires in aspen stands in southern Ontario increased the levels of nitrogen, calcium, phosphorus, magnesium, and potassium in aspen leaves the first growing season after burning (James and Smith 1977). The level of potassium in twigs was lower in burned than unburned stands.

Increases in some nutrients have been reported after severe fire. Ohmann and Grigal (1979) reported the effects of a mixed-severity fire (with large areas of stand-replacement) in forests of jack pine, quaking aspen, and paper birch in northern Minnesota. Concentrations of potassium, calcium, and magnesium increased during the first 5 years after fire, generally exceeding levels on unburned sites. Phosphorus on burned sites also exceeded that on unburned sites. Nitrogen concentrations were higher on burned than unburned sites but declined during the first five growing seasons after fire.

Eastern Ecosystems and the Great Plains

Eastern Deciduous Forests

Fire and the Quantity of Forage—Biomass of herbs and shrubs usually increases after fire in Eastern deciduous forests. The fire frequency needed for maximum productivity differs among vegetation types. Prescribed understory burns at 10- and 15-year intervals did not affect shrub or herbaceous cover in New Jersey pine stands in comparison with unburned sites (Buell and Cantlon 1953). As intervals between burns decreased to 5, 4, 3, 2, and 1 years, however, the shrub cover decreased and cover of herbaceous plants, mosses and lichens increased. Two and three growing seasons after late winter-early spring prescribed burns in oak-hickory stands in West Virginia (Pack and others 1988), herbaceous vegetation increased, although response to burning varied considerably. Results suggested that thinning stands prior to prescribed burning was necessary to increase understory productivity.

Fire and Nutritional Quality—Many studies from the Northeast and Midwest report increased nutrition of wildlife foods after fire, but the duration of increases varies. Prescribed understory burns in April increased the July levels of crude protein in scrub oak foliage in central Pennsylvania for 4 years (Hallisey and Wood 1976). Crude protein also increased in blueberry foliage, but only during the first growing season. Phosphorus, magnesium, and calcium levels were higher in scrub oak foliage, and magnesium was higher in teaberry the first growing season after fire. During the growing season after an April understory burn in mixed-oak forests in Wisconsin, the concentration of nitrogen, phosphorus, and potassium in leaves increased (Reich and others 1990). The increase was thought to be due to increased availability of nutrients in the soil. For most nutrients in most plant species, the effect decreased throughout the growing season.

Southeastern Forests

Fire and the Quantity of Forage—Several studies indicate that, in general, understory fire in Southern forests does not increase the biomass of forage but often increases the proportion of herbage to browse (Evans and others 1991; Stransky and Harlow 1981; White and others 1991) (fig. 22). Other studies have found that biomass increased after burning only under some conditions. Gilliam (1991) reported an increase in herbaceous biomass after burning a pine-bluestem range in Louisiana that had not been burned in 40 years. Prescribed burning combined with herbicides significantly increased the amount of forage in oak-hickory stands in northeastern Oklahoma (Thompson and others 1991). Grasses and legumes increased after fire reduced the canopy in oak-pine stands in Oklahoma and Arkansas (Masters and others 1993). Understory burning at 1- and 2-year intervals favored herbaceous cover, while understory burning at 3- and 4-year intervals favored a mixture of herbs and shrubs.

Fire and Nutritional Quality—Based on a review of 16 studies of fire in Southern forests, Stransky and Harlow (1981) propose several generalizations about the effects of fire on plant nutrition. Burning typically increases the crude protein and phosphorus content of grasses, forbs, and browse the first postfire year. Increases in nutritive quality are greatest at the beginning of the growing season and decline rapidly, so protein and phosphorus levels are usually similar in burned and unburned areas by winter. Calcium content of plants after burning is highly variable among studies. The palatability of forage generally improves after fire, at least until growth stops and lignin content increases. Seldom are effects of burning on

Figure 22—Biennial prescribed burn plots, St. Marks National Wildlife Refuge, Florida. Area in foreground is burned every other August and dominated by runner oak, a mast-producing species. Area in background is burned every other April and shows wiregrass flowering. This is a species favored by seed-eating animals. Photo courtesy of Dale Wade.

the nutritional content of plants detected after the first growing season (Christensen 1977; Stransky and Harlow 1981; Thill and others 1987). However, high-intensity fires in the Florida Keys, oak communities, and pine-oak communities in the Southeast have extended fire's positive effects on plant nutrition for at least 1 year beyond the first growing season (Carlson and others 1993; DeWitt and Derby 1955; Thackston and others 1982). Exceptions to the pattern of nutrient increases after fire include reports from Florida sandridge habitat (Abrahamson and Abrahamson 1989) and eastern Texas pine-hardwood (O'Halloran and others 1987). These studies report no substantial increase in plant or fungus nutrient levels after fire.

Fire and the Quantity of Seeds and Mast—Seed and mast production generally increases after fire in Southern forests. According to Harlow and Van Lear (1989), seed production by legumes, grasses and spurges is significantly greater on annually burned areas than on areas burned less frequently. Production of berries, drupes, and pomes peaks 2 to 4 years after burning for most of 20 species of shrubs and small trees. A stand-replacement fire in pine and hardwood in the mountains of Virginia greatly increased the production of blueberries the second growing season after burning. Production declined by year 5 but remained higher than that on unburned plots (Coggins and Engle 1971). Season and frequency of burning influence berry production. Waldrop and others (1987) found that annual and biennial summer fires in loblolly pine forests on the Coastal Plain of South Carolina reduced the numbers of blueberry plants after 30 years of burning, whereas winter burning did not. According to a review by Robbins and Myers (1992), frequent growing season burns reduce mast-producing species except runner oak and some blueberry species.

Hard mast in the Southeast is used by a variety of birds and mammals, including northern bobwhite, wild turkey, sapsuckers, squirrels, black bear, and white-tailed deer. The current lack of frequent fires (both understory and stand-replacement) in the southern Appalachians is thought to be responsible for the replacement of oaks by other species (Van Lear 1991; Van Lear and Watt 1993). Small oaks resprout after fires, and large oaks have fire-resistant bark that enables these large trees to survive fire better than their competitors if frequently underburned. However, when fires are excluded for long periods, competing species such as tuliptree also develop fire-resistant bark. These competitors can survive fire, but they have much less potential than oaks for producing hard mast. Thirty years of prescribed burning in the Coastal Plain of South Carolina had no effect on the number of mast-producing hardwoods more than 5 inches (12.5 cm) dbh (oaks, hickories, blackgum and others) (Waldrop and Lloyd 1991). Annual summer fires nearly eliminated small (less than 1 inch, 2.5 cm dbh) hardwoods, but all other burning treatments produced increases in the mast-producing species (Waldrop and others 1987).

Recommendations for use of prescribed fire often focus on specific wildlife-related objectives. Johnson and Landers (1978) recommend understory fire at 3-year intervals to optimize fruit production in open slash pines in Georgia, with some use of longer intervals to promote mast-producing species. The animals favored by this practice include white-tailed deer, common gray fox, northern bobwhite, wild turkey, raccoon, and songbirds. Hamilton (1981) recommends understory burning pine-hardwood habitats in winter at 5- to 10-year intervals to provide ample berries and mast for black bears. Harlow and Van Lear (1989) suggest that annual burning may not be desirable for the majority of wildlife species because mast production for most shrubs and small trees peaks 2 to 6 years after burning. However, annual burning would benefit northern bobwhite, mourning doves, some songbirds, and rodents.

Fire and Availability of Invertebrates—Fire effects on invertebrates relate not only to preburn conditions and fire severity but also to the life cycles and population patterns of the specific invertebrates studied. In the sandhills of the Florida panhandle, longleaf pine stands with dense turkey oak and sand live oak in the understory were underburned during the growing season. Arthropod density and biomass increased significantly, especially populations of grasshoppers, which constituted more than 90 percent of the arthropod biomass (Provencher and others 1998). Such increases are likely to benefit the northern bobwhite, a bird that feeds in the ground and herb layer, and hawking birds such as the loggerhead shrike and kestrels. Dunaway (1976) found that annual understory burning in longleaf pine did not increase the number of insects available to foraging birds, but may have provided open conditions conducive to successful hunting by northern bobwhite chicks. In central Florida sandhills, increased frequency of understory burning was positively correlated with the colony density of southern harvester ants (McCoy and Kaiser 1990). Understory fire in loblolly pine-shortleaf pine forest in east-central Mississippi increased invertebrates available to northern bobwhite and turkeys for up to 3 years (Hurst 1971, 1978).

While some insects are attracted to fire and increase rapidly in burns, fire reduces others. For example, fire is used in the southern Appalachian Mountains to control insects that prey on oak seedlings and mast (Van Lear and Watt 1993).

Prairie Grassland

Fire and the Quantity of Forage—In presettlement times, frequent fires in grasslands kept tree cover in check. Studies in many regions describe the invasion of prairie by trees in the absence of fire (Gruell 1979; Reichman 1987; Sieg and Severson 1996). Where prairie fires eliminate trees, fires increase the amount of forage available to fauna simply by increasing the area covered by prairie. In addition, grassland fire can cause early green-up of warm-season grasses, improved seed germination, and greater production of grasses and forbs (Hulbert 1986, 1988; Svejcar 1990). Dramatic increases in yield during the first postfire year have been reported for dominant prairie grasses including prairie dropseed, big bluestem, western wheatgrass, bluebunch wheatgrass, and Indiangrass (Bushey 1987; Dix and Butler 1954; Hulbert 1988; Svejcar 1990). Many studies that report increased yield also describe some circumstances under which yield is reduced. In general, fires followed by drought and fires in areas with less than 11 inches (300 mm) of summer rainfall may cause decreased forage production (Kucera 1981).

Fire and Nutritional Quality—Fire often increases the percentage of protein and minerals in prairie grasses and shrubs, although effects vary with season of burning (Daubenmire 1968). Forage quality in mountain shrub and grassland communities is enhanced by increased availability of mineral nitrogen (Hobbs and Schimel 1984). The effects of fire on grassland nutrients interact with the effects of grazing. Grazed patches in a tallgrass prairie contained less biomass than ungrazed patches and therefore lost less nitrogen to volatilization by fire (Hobbs and others 1991). The differences were substantial enough that grazing may control whether burning causes net increases or decreases in nitrogen on a site. Grazing also increases heterogeneity in grasslands, contributing to patchy fuels and thus variation in fire behavior and severity.

Fire and Availability of Invertebrates—Reed (1997) reviewed studies of fire effects on prairie arthropod communities. She found that fire modified these communities, and the communities continued to change with time after fire. Prairies with fires initiated in different years and different seasons are likely to promote species richness. Fire in oak savannas, studied over a 30-year period, did not significantly alter arthropod diversity (Siemann and others 1997). Fires in Texas grasslands did not significantly alter arthropod abundance and availability to foraging birds (Koerth and others 1986). The density of arachnids and insect orders on Texas grasslands, however, differed significantly between burned and unburned areas at various times of year (table 5).

Beetle abundance declines immediately after fire in prairies but may return to prefire levels within a month (Rice 1932). On a tallgrass prairie in Kansas, arthropod biomass was greater on annually burned than unburned plots; cicada nymphs were more abundant on burned than unburned plots (Seastedt and

Table 5—Effects of fire on invertebrates in a Texas grassland after a January fire (Koerth and others 1986). Groups listed were significantly ($p < 0.05$) more (or less) abundant on burned than unburned areas, as indicated by density (number/m^2).

	First year after fire		Second year after fire	
Month	More abundant on burn	Less abundant on burn	More abundant on burn	Less abundant on burn
April			Orthoptera	Hemiptera
May		Hemiptera Homoptera Coleoptera		
June	Diptera	Arachnidae		Homoptera
July	Hymenoptera	Hemiptera Arachnidae		 Homoptera
August				Coleoptera Arachnidae
September	Hymenoptera	Coleoptera Arachnidae		
October	Hymenoptera	Hemiptera Arachnidae	Orthoptera Arachnidae	

others 1986). Cicadas respond positively to increased root productivity on burned sites, but they are relatively immobile so their feeding is unlikely to contribute to decline of their host plants.

Western Forests

Rocky Mountain Forest

Fire and the Quantity of Forage—Most studies of fire and wildlife foods in Western forests focus on ungulates. This research generally indicates that burning produces positive results for elk and mule deer. During the first 5 to 10 years following stand-replacing fire, grass and forb biomass generally increases. Grass and forb biomass decreased the first growing season after fire in aspen stands in Wyoming but increased the second and third growing seasons to above preburn levels (Bartos and Mueggler 1981). On "heavily burned" sites, grass recovered more slowly than forbs. Forage increased three-fold after both understory and stand-replacement fire in a ponderosa pine forest in Arizona (Oswald and Covington 1983). The increase persisted 9 years in underburned stands, but grazing, perhaps combined with severe fire effects, reduced forage after 2 years in areas burned by stand-replacing fire. Climax bunchgrass stands have been recommended for bighorn sheep winter range, but bighorn sheep in western Montana preferred seral forest with elk sedge and pinegrass openings (Riggs and Peek 1980).

Although total biomass of grasses and forbs often increases following fire, the quantity of useable forage may actually be less on burned areas if species composition shifts to domination by relatively unpalatable species. Prescribed understory burning failed to improve forage in some Southwestern ponderosa pine stands because, although herbage increased dramatically, flannel mullein, an unpalatable species, dominated the understory after fire (Ffolliott and Guertin 1990).

Burning brush fields in northern Idaho greatly increased the browse available to wintering elk the following year (Leege and Hickey 1971). In British Columbia, elk wintered primarily in postfire grass and shrub communities, except during severe weather when conifer stands were used (Peck and Peek 1991). In Idaho, mule deer foraged primarily in burned habitats in winter, while white-tailed deer foraged primarily in unburned habitats (Keay and Peek 1980). An intense prescribed fire in Douglas-fir in Idaho improved forage for mule deer and elk. The benefits were expected to last more than 20 years (Lyon 1971). Positive effects of fire on grazing and browse productivity generally last less than 30 years (Oswald and Covington 1983; Pearson and others 1972).

Mixed-severity and stand-replacement fires often increase berry-producing shrubs and their productivity 20 to 60 years after fire. These changes benefit birds, small mammals, and bear. Increased production of forb foliage and tuberous roots after the 1988 Yellowstone fires benefited grizzly bears (Blanchard and Knight 1996). A mathematical model predicts increased wintering populations of elk and bison in Yellowstone for 20 to 30 years postfire (Boyce and Merrill 1991). Large, intense burns may be necessary for long-term maintenance of natural forest succession patterns of some forest types and for habitat diversity in others (Finch and others 1997). While fires top-kill huckleberry plants and kill many whitebark pines, two species that provide important forage for grizzly bears, they also rejuvenate decadent huckleberry stands and prevent subalpine fir from replacing whitebark pine in many high elevation forests (Agee 1993).

Fire and Nutritional Quality—Fires usually increase some nutrients in Rocky Mountain forests and the pine forests of Arizona and New Mexico for 1 to 3 years (Severson and Medina 1983). Stand-replacement fall burning in Wyoming aspen stands increased crude protein and phosphorus of forage during the first summer after treatment (DeByle and others 1989). In vitro dry matter digestibility was also higher in burned areas, and calcium content was lower. By late summer, only crude protein levels were different and, in the second postfire year, forage quality was similar on burned and unburned areas. Burning improves the nutritional qualities of forage plants in ponderosa pine forest for one to three growing seasons (Meneely and Schemnitz 1981; Pearson and others 1972; Rowland and others 1983). In western larch/Douglas-fir stands in Montana that had been burned with understory fire 3 years previously, nutrient content of plants was compared with samples from stands not burned for 70 years (Stark and Steele 1977). Sodium levels were higher for several species in stands where at least half of the duff was consumed by fire. Iron concentration was significantly greater in some species on burned than unburned sites, and calcium and phosphorus were significantly lower. The plant species tested showed no significant differences in nitrogen, magnesium, or copper between burned and unburned sites. Scouler's willow in underburned ponderosa pine/Douglas-fir forests in Montana contained higher concentrations of phosphorus and crude protein, and lower lignin concentration, than willows in unburned stands (Bedunah and others 1995).

Some research reports no significant changes in nutrient levels after fire. Seip and Bunnell (1985) found no differences in the nutritive quality of forage on frequently burned alpine range and unburned range used by Dall's sheep in British Columbia. The authors thought that sheep on burned range were in better physical condition than those on unburned range because of the quantity of forage rather than its nutritive quality. Stand-replacing prescribed fire in Idaho aspen forests in September produced little change in the nutritive content of forage the first, second, and fourth growing seasons after burning (Canon and others 1987). However, elk preferred to forage in the burned areas, possibly because preferred species were consistently available and foraging was more efficient.

Sierra Forest

Fire and the Quantity of Forage and Seed—Wildlife forage species in Sierra Nevada forests include many plants that dominate in chaparral to the west and more mesic forests to the north. Deerbrush and greenleaf manzanita are chaparral species but are also important components of the understory of Sierra forests. Forage of deerbrush and other *Ceanothus* species, which is high quality food for ungulates (Sampson and Jesperson 1963; Stubbendieck and others 1992), is abundant after fire because it reproduces from seed that is scarified by burning (Burcham 1974). Early spring burning in the Sierra Nevada increases palatability of foliage for wildlife (Kauffman and Martin 1985). Thimbleberry is an understory species characteristic of mesic Sierra forests; it generally increases after fire (Hamilton and Yearsley 1988).

Pacific Coast Maritime Forest

Fire and the Quantity of Forage—Salmonberry, an important understory species in Pacific Coast forests, is used by numerous wildlife species. Deer, elk, mountain goats, and moose browse on its buds and twigs; songbirds, gallinaceous birds, bears, and coyote feed on its fruit. Salmonberry sprouts prolifically and grows rapidly in the first years after fire, although severe fire may reduce sprouting (Tappeiner and others 1988; Zasada and others 1989).

Western Woodlands, Shrublands, and Grasslands

Pinyon-Juniper

Fire and the Quantity of Forage—Severson and Medina (1983) and Severson and Rinne (1990) review the effects of fire on forage production and wildlife habitat in the Southwest. While they demonstrate the important role of fire in improving Southwestern vegetation types for wildlife, they emphasize the need for a balance between burned and unburned areas. Fire intensity varies greatly in pinyon-juniper woodlands, and the early successional effects of fires are difficult to predict (Severson and Rinne 1990). Often, fire may not have much effect unless combined with other treatments (Wittie and McDaniel 1990). When conditions are favorable for stand-replacing fire, burning kills most of the pinyon-juniper overstory and increases diversity in the plant community, with some effects lasting up to 115 years after fire (McCulloch 1969; Severson and Medina 1983; Severson and Rinne 1990; Stager and Klebenow 1987). Shortly after fire, burns are usually dominated by forbs, with grasses becoming abundant a few years later. In an Ashe's juniper community burned during a moist winter and spring, grasses recovered quickly and soil erosion was minimal (Wink and Wright 1973). Similar treatments during a dry winter and spring, however, reduced herbaceous yields and increased erosion.

Chaparral and Western Oak Woodlands

Fire and the Quantity of Forage—Intense fires in chaparral result in a flush of herbaceous plants and shrubs for 1 to 5 years (Biswell 1974; Christensen and Muller 1975; Klinger and others 1989; Taber and Dasmann 1958). In Gambel oak rangeland in Colorado, fire did not significantly change the biomass of forbs and shrubs 2, 5, and 10 years after fall mixed-severity fire, but grass biomass was greater on burned than unburned sites during postfire year 10 (Kufeld 1983).

Fire and Nutritional Quality—Most studies of postfire nutrients in Western ecosystems report some changes, but the plant species and the nutrients affected vary. Stand-replacing fires in chaparral increased the protein content of leaves for one to two growing seasons and the phosphorus content for up to 6 years (Rundel and Parsons 1980; Taber and Dasmann 1958). Two growing seasons after fall mixed-severity burns in Gambel oak rangeland in Colorado, zinc and copper levels were higher in plants on burned than unburned sites. However, no differences were found in the protein, lignin, calcium, or phosphorus content of forbs, grasses or shrubs growing on burned and unburned areas (Kufeld 1983).

Where postfire nutrient changes vary among the plant species available to fauna, animals may select the more nutritious foods. September prescribed burns in mountain shrub and grassland habitats in Colorado increased the level of protein and in vitro digestible organic matter in winter diets of bighorn sheep and mule deer (Hobbs and Spowart 1984). Burning had no detectable effect on spring diets. The effects of burning on crude protein in the diet persisted for 2 years in both communities. The effect on digestible matter was present only in the mountain shrub habitat the second year. The increase in the nutritional quality of diets was greater than the apparent increase in the quality of browse and forage, indicating that sheep and deer foraged selectively for the plants that were more nutritious.

Fire and the Quantity of Seeds and Mast—The acorns produced by Western oak woodlands are used by birds, small mammals, and ungulates. Oaks that have been severely damaged by fire may produce “massive” seed crops (Rouse 1986).

Sagebrush and Sagebrush Grasslands

Fire and the Quantity of Forage—Some studies report no increases in grass and sagebrush productivity due to fire but do report other changes favorable to ungulates. Burning big sagebrush-bluebunch wheatgrass winter range in Wyoming decreased sagebrush for the 4 years of study but did not increase wheatgrass. Annual forbs were more abundant on the burned area only the second year after burning. Nonetheless, bighorn sheep and possibly mule deer made greater use of the burned areas than the unburned areas (Peek and others 1979). Prescribed burning reduced plant litter that inhibited grazing by elk on a Montana fescue-wheatgrass winter range. Fire did not significantly change the forb, shrub, and grass standing crops, however, except that rough fescue, the preferred winter forage, was reduced the first year after burning (Jourdonnais and Bedunah 1990).

Fire and Availability of Invertebrates—Stand-replacing and mixed-severity fire in big sagebrush in Oregon did not affect populations of darkling beetles or June beetles (Pyle and Crawford 1996). Fire did not appreciably alter their food and cover on the ground surface.

Deserts

Fire and the Quantity of Forage—Fire reduces most shrubs in the Great Basin Desert for at least a few years (Humphrey 1974). In the first year after fire, perennial grasses and forbs have reduced vigor and annuals are abundant. By the third year, total herbage often reaches a maximum, exceeding production on unburned sites, and grasses and herbs flower profusely. The dominant grasses are thickspike wheatgrass, plains reedgrass, and bluebunch wheatgrass; other grasses, including bluegrass species and Idaho fescue, do not recover to preburn production until the second decade after fire.

Fire in the Mojave Desert is likely only after a season of heavy production by annual plants. The moisture levels of woody and perennial plants determines the level of mortality. If conditions are excessively dry, damage is severe.

In the Sonoran and Chihuahuan Deserts, fire is uncommon because of the widely spaced, open-branched vegetation. In wet years, fires occur in grasslands and their interface with desert, killing woody plants, such as velvet mesquite, and expanding the grassland. Fires that burn off the spines from cacti (cholla, pricklypear, and barrel cactus) make the plants available as forage for livestock and rabbits. Fires at the grassland-woodland ecotone may remove woody vegetation without increasing ground cover (Bock and Bock 1990). In desert grasslands, fire is likely to reduce yield for 1 to 2 years, with productivity recovering to preburn levels by the third year (Jameson 1962; Wright 1980). Where black grama is dominant, fire effects vary. Productivity may be reduced for 10 years or longer (Wright 1980). Tobosagrass production increased two to threefold after early spring burns followed by rain, but burning in a dry spring reduced yield (Wright 1973).

Subtropical Ecosystems

Florida Wetlands

Fire and the Quantity of Forage—In Florida wetlands, fires increase open aquatic areas and reduce the encroachment of pine hammocks, thus altering the balance between terrestrial and aquatic habitat. Burning opens up cattail stands by removing years of accumulated litter. Fire eliminates litter in sawgrass stands and reduces plant height for a year or two. To maintain fruit production for white-tailed deer, Fults (1991) recommends burning saw-palmetto understories every 3 to 5 years.

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Chapter 8: Management and Research Implications

Management Implications

Only a few places in North America, or the world, exist where fire has not shaped the vegetation or influenced the faunal community. In many areas of North America, managers have successfully prevented or limited the occurrence of this natural process for nearly 100 years, and that century of fire exclusion has probably caused many changes in habitat and wildlife populations of which we are not even aware. It is likely that some faunal populations and communities present in today's landscapes could not have developed under pre-1900 fire regimes. Many researchers and managers agree, however, that the success of fire exclusion cannot continue (Fiedler and others 1998; Fule and Covington 1995) and, indeed, is already beginning to fail (Barbouletos and others 1998; Wicklow-Howard 1989; Williams and others 1998). Fire is most likely to increase in wildlands in the future. This likelihood carries with it two broad implications for the relationships between fire and fauna.

One: Alternatives in Managing Fire

Managers are increasingly likely to have to choose among:

- Massive fire suppression (with increasing hazards and increasing costs).
- Uncontrolled, possibly uncontrollable fires.
- A combination of prescribed fires and wildland fires used to achieve resource objectives.

The implications of these choices for animal communities in North American wildlands are significant. Most North American fauna communities have developed under pressure from repeated fires of specific severities and frequencies. Alteration of that pressure for the past 100 to 500 years has changed the abundance and geographic distribution of many kinds of habitat and the animals that depend on it.

Even more important than changes in past centuries, however, is the likelihood that fires in the immediate future will deviate substantially from what might be considered normal or natural in many areas of North America. While restoration of presettlement fire regimes may be desirable for habitat protection, this may be impossible in many areas because of fuel accumulation, structural change due to fire exclusion, and climate change (see discussion of this topic in "Effects of Fire on Flora" in the Rainbow Series). Even if habitat restoration is successful, animal populations may be slow to colonize treated areas, so perpetuation of existing habitat is a more reliable management strategy than restoration of degraded habitat. Managers attempting to restore habitat by emulating presettlement fire regimes will not only encounter increased fuel loads and increased continuity of fuels, but also resistance from the public because of the immediate increased risks to human life, health, property, and welfare. The altered vegetation may need to be burned under conditions that would not normally incur extensive fire spread. For many fauna species, this practice can produce site and landscape conditions completely outside the range of those under which the species evolved. Because spatial and temporal variation are important aspects of presettlement fire regimes, management plans should address these features explicitly whenever possible (Lertzman and others 1998).

Considering the many variables and unknowns that impinge upon management choices in regard to fire, careful consideration of the science and monitoring of treatment results is important. As Rieman and others (1997) comment regarding fire effects on aquatic fauna, "There is undoubtedly a point where the risk of fire outweighs the risk of our management, but that point needs to be discovered through careful evaluation and scientific study not through the opposing powers of emotional or political rhetoric."

Two: Integrating Management Objectives

Objectives of prescribed fires and use of wildland fires for resource benefits must be clearly stated and integrated with overall land management objectives, addressing the potential for interaction among disturbances such as grazing, flood, windthrow, predation, and insect and fungal infestation. In the past 10,000 years, fire has never operated in isolation from other disturbances, nor has fire usually occurred independent of human influence (Kay 1998; Pyne 1982). During thousands of years prior to settlement of North America by European Americans, Native Americans influenced both fire regimes and animal populations. In fact, populations of large ungulates may have been limited by Native American predation rather than food (Kay 1998). As Kay (1995) states, "Setting aside an area as wilderness or a National Park today, and then managing it by letting nature take its course will not preserve some remnant of the past but instead create conditions that have not existed for the last 10,000 yr." As managers face ubiquitous needs for addressing fire in land management, and as they encounter increasing difficulty in managing habitat in conditions near those under which faunal species evolved, we believe it is of paramount importance to have clear objectives for use of prescribed fire, wildland fire for resource benefits, and fire suppression, based on understanding of past disturbance patterns and human influence. It is important to avoid, if possible, major deviations into ecological conditions outside the range of variability that occurred in the millennium prior to 1900.

When fire suppression and use are not integrated with overall management programs, the potential for unanticipated problems and failure increases. Management for aspen restoration and bighorn sheep range improvement provide two examples. If aspen is treated by fire to regenerate the stand but then repeatedly browsed by wildlife, it often deteriorates more rapidly than without treatment (Bartos 1998; Basile 1979). The choice of treatment and the size and distribution of treated sites must in this case be integrated with knowledge of wildlife use patterns and wildlife management. Prescribed fire can negatively affect bighorn sheep habitat when range condition is already poor, when the burn leaves inadequate forage for the winter, and when other species, especially elk, are attracted to the burned habitat (Peek and others 1985). Again, fire management needs to be integrated with wildlife information and management.

Understanding of fire history, potential fire behavior, and differing needs of multiple species must be integrated in planning for prescribed fire. For example, since many small mammals use tunnels under forest litter and in or near large pieces of dead wood as refugia (Ford and others 1999), managers can influence the impact of fire on small mammals by including moisture levels of these fuels in plans for fire use. Salvaged logged sites in stand-replacement burns in the Northern Rocky Mountains provide nesting opportunities for some cavity nesters (northern flicker, hairy woodpecker, and mountain bluebird). Other bird species (black-backed woodpecker, northern three-toed woodpecker, and brown creeper) occur almost exclusively in burned, unlogged patches (Hejl and McFadzen 1998). If salvage logging is considered after a wildland fire, the needs of the specific bird community in the area must be considered.

Because funding and other resources for management will always be limited, it is important to use objectives to shape clear priorities for fire suppression and fire use. Is it more important to use limited resources on small areas that will benefit small, but perhaps irreplaceable, populations of animals? Or is it more important to restore large areas and address the challenges of landscape-level planning? Only carefully thought-out objectives can guide such choices well.

Needs for Further Understanding

Research questions regarding fire effects on fauna fall into two categories: (1) those regarding fauna-habitat relationships and (2) those regarding presettlement fire regimes.

Fauna-Habitat Relationships

Information involving relationships between fire and animals is needed for all classes of fauna. Most of the information currently available focuses on vertebrates, particularly mammals and birds. Studies of landscape and community ecology are virtually limited to birds. Furthermore, most studies are limited to population descriptors, while measurement of productivity may be essential for understanding fire effects and predicting effects of management options. Given the relative lack of information about fire effects on herpetofauna and insects, studies in those areas may be especially important (Pickering 1997; Russell and others 1999). Future research should address microsite conditions, patchiness within burns, and seasonality of fire effects for specific ecosystems. Likewise, information about fire effects on aquatic fauna is sparse, much of it originating from only a few ecosystems (for example, see Bozek and Young 1994; Mihuc and others 1996; Minshall and others 1989; Rieman and others 1997). More information is needed regarding long-term effects, landscape effects, and effects of postfire succession on aquatic fauna. (See also discussion of this topic in "Effects of Fire on Soil and Water" in the Rainbow Series.)

The need to fill information gaps will increase as stands and landscapes continue to diverge from presettlement patterns and as managers increasingly use fire for vegetation management. To improve long-term management for sustaining ecosystems, information is needed about the effects of fire on many kinds of fauna, at different seasons and under different conditions, and over many decades. Information on the interactions of burning season with life cycles of animal species, especially insects and herpetofauna, is also important.

Site-Level Research Questions—At the site level, managers need detailed information on the use of fire to manage the structure of vegetation, especially in shrublands and forest understories. Objectives for this kind of management include maintaining nesting habitat for birds, ensuring habitat features needed for reproduction by herpetofauna and insects, providing cover for small mammals, and enhancing local community diversity.

Also at the site level, managers need better designed, more comprehensive studies of fire impacts on quantity and quality of forage for wildlife. A truly vast literature addresses this subject, but much of it is hard to apply because the investigators did not control for factors other than burning and did not describe fire severity or burning conditions in detail. Land managers in many localities currently use limited amounts of prescribed fire to enhance wildlife habitat, but more widespread use of fire in habitat management will require more comprehensive knowledge than is currently available.

Landscape-Level Research Questions—At the landscape level, we lack almost any knowledge of the combination of mosaics and patterns best suited to specific populations, and we have little understanding of how to maintain the total landscape for regional biodiversity. While habitat corridors are important for sustaining some wildlife species (Beier and Noss 1998; Oliver and others 1998), what are the implications of fire and succession in corridors and the locations that provide access to them? Some research of this kind is under way, but limitations of time and money will virtually assure that computer models rather than landscape-level experiments will provide the greatest progress (Schmoldt and others 1999).

Wildlife researchers often face a dilemma regarding research priorities: Should we invest time and resources in learning more about faunal habitat, or should we learn more about the species themselves? The answer depends on the ecosystem under study. Schultz and Crone (1998) developed a model for habitat change in the prairie habitat of the Fender's blue butterfly, a candidate for listing on the U.S. Endangered Species list. They report that lack of knowledge about postfire habitat change limited the certainty of the model's predictions more than lack of knowledge about the butterfly itself. In contrast, both Wright (1996) and Telfer (1993) state that information about the fauna species investigated (birds in both studies), especially nesting success, currently limits our ability to understand the effects of potential management choices, including those regarding fire.

Presettlement Fire Regimes

Important knowledge gaps remain about the distribution and structure of vegetation in presettlement times. Without this information, managers cannot decide what proportion of forest land should be in various age classes, structural classes, and cover types to maintain biodiversity. Furthermore, managers need methods for integrating current agricultural and infrastructural elements in the landscape with remaining wildlands at large scales, approximating the original fire-shaped mosaic and structure for an area as well as possible. With this information, wildlands can be used to the best advantage to maintain regional biodiversity, increase numbers of particular wildlife species, and achieve other environmental goals.

Human Dimension

Finally, researchers and managers need to collaborate in assessing the comparative merits and drawbacks of various kinds of fire for natural resource objectives across the landscape. What ecological and social risks occur with prescribed fires, wildland fires managed for resource objectives, and fire suppression? How can these risks be reduced? It is impossible to know all the consequences of intervening in an ecosystem, whether the intervention is active (prescribed fire, for example), or passive (such as fire exclusion or landscape fragmentation). Monitoring and comparison of monitoring results with predictions are essential. Communication among researchers, managers, and the public is also essential. Science cannot be used until it is shared with and understood by managers, whose job is to apply the results, and a substantial proportion of the public, who add the perspective of their values and experience. Policy, according to Pyne (1982), "has to be based on broad cultural perceptions and political paradigms, not solely on ecological or economic investigations; scientific research is only one component among many that contribute to it."

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Appendices

Appendix A: Common and Scientific Names of Animal Species

Taxonomy for birds is from Ehrlich and others (1988); for reptiles and amphibians, Conant and Collins (1991) and Stebbins (1985); for mammals, Jones and others (1992); for insects, Borror and White (1970).

Common name	Scientific name
Agile kangaroo rat	*Dipodomys agilis*
American Kestrel	*Falco sparverius*
American marten	*Martes americana*
American Robin	*Turdus migratorius*
American Wigeon	*Anas americana*
American badger	*Taxidea taxus*
American beaver	*Castor canadensis*
Arachnids (includes spiders and ticks)	*Arachnidae*
Bachman's Sparrow	*Aimophila aestivalis*
Bald Eagle	*Haliaeetus leucocephalus*
Barn Swallow	*Hirundo rustica*
Beetles	*Coleoptera*
Bewick's Wren	*Thryomanes bewickii*
Bighorn sheep	*Ovis canadensis*
Bison	*Bison bison*
Black bear	*Ursus americanus*
Black Vulture	*Coragyps atratus*
Black-backed Woodpecker	*Picoides arcticus*
Black-chinned Sparrow	*Spizella atrogularis*
Black-footed ferret	*Mustela nigripes*
Black-tailed jackrabbit	*Lepus californicus*
Blue Jay	*Cyanocitta cristata*
Blue-headed Vireo	*Vireo solitarius*
Blue-winged Teal	*Anas discors*
Box turtle	*Terrapene carolina*
Brewer's Sparrow	*Spizella breweri*
Brown Creeper	*Certhia americana*
Brown-headed Nuthatch	*Sitta pusilla*
Brush mouse	*Peromyscus boylii*
Brush rabbit	*Sylvilagus bachmani*
Bugs	*Hemiptera*
Burrowing Owl	*Athene cunicularia*
California Gnatcatcher	*Polioptila californica*
California Quail	*Callipepla californica*
California mouse	*Peromyscus californicus*
California pocket mouse	*Chaetodipus californicus*
Canyon Towhee	*Pipilo fuscus*
Caribou	*Rangifer tarandus*
Carolina Wren	*Thryothorus ludovicianus*
Cattle Egret	*Bubulcus ibis*
Central Florida crowned snake	*Tantilla relicta*
Chigger	*Acarina*
Chipping Sparrow	*Spizella passerina*
Cicadas, hoppers, whiteflies, aphids, scale insects	*Homoptera*

Appendix A

Common name	Scientific name
Clark's Nutcracker	*Nucifraga columbiana*
Clay-colored Sparrow	*Spizella pallida*
Common Grackle	*Quiscalus quiscula*
Common gray fox	*Urocyon cinereoargenteus*
Common Ground-dove	*Columbina passerina*
Common Raven	*Corvus corax*
Common Yellowthroat	*Geothlypis trichas*
Cooper's Hawk	*Accipiter cooperii*
Cotton rat	*Sigmodon* spp.
Coyote	*Canis latrans*
Crested Caracara	*Caracara plancus*
Crossbills	*Loxia* spp.
Dall's sheep	*Ovis dalli*
Dark-eyed Junco	*Junco hyemalis*
Darkling beetles	*Tenebrionoidea*
Deer mouse	*Peromyscus maniculatus*
Desert woodrat	*Neotoma lepida*
Desert tortoise	*Gopherus agassizii*
Diamondback rattlesnake	*Crotalus admanteus*
Downy Woodpecker	*Picoides pubescens*
Dusky-footed woodrat	*Neotoma fuscipes*
Eastern glass lizard	*Ophisaurus ventralis*
Eastern Kingbird	*Tyrannus tyrannus*
Eastern Wood-pewee	*Contopus virens*
Eastern Bluebird	*Sialia sialis*
Eastern Towhee	*Pipilo erythrophthalmus*
Elk	*Cervus elaphus*
European Starling	*Sturnus vulgaris*
Feather Mite	*Acarina*
Fender's blue butterfly	*Icaricia icarioides fenderi*
Ferruginous Hawk	*Buteo regalis*
Field Sparrow	*Spizella pusilla*
Flammulated Owl	*Otus flammeolus*
Flatwoods salamander	*Ambystoma singulatum*
Florida Scrub-Jay	*Aphelocoma coerulescens coerulescens*
Fowler's toad	*Bufo woodhousii*
Fox Sparrow	*Passerella iliaca*
Golden eagle	*Aquila chrysaetos*
Golden-crowned Kinglet	*Regulus satrapa*
Gopher tortoise	*Gopherus polyphemus*
Gopher snake	*Pituophis melanoleucus*
Grasshopper Sparrow	*Ammodramus savannarum*
Grasshoppers, katydids, crickets, mantids, walkingsticks, and cockroaches	*Orthoptera*
Gray wolf	*Canis lupus*
Gray Catbird	*Dumetella carolinensis*
Great Horned Owl	*Bubo virginianus*
Great Crested Flycatcher	*Myiarchus crinitus*
Great Blue Heron	*Ardea herodias*
Greater Prairie-Chicken	*Tympanuchus cupido pinnatus*
Grizzly bear	*Ursus arctos*
Ground squirrel	*Spermophilus* spp.

Appendix A

Common name	Scientific name
Hairy Woodpecker	*Picoides villosus*
Hammond's Flycatcher	*Empidonax hammondii*
Heath Hen	*Tympanuchus cupido cupido*
Heerman kangaroo rat	*Dipodomys heermanni*
Hermit Thrush	*Catharus guttatus*
Hooded Warbler	*Wilsonia citrina*
Indigo Bunting	*Passerina cyanea*
June beetles	*Melolonthinae*
Kangaroo rat species	*Dipodomys* spp.
Karner blue butterfly	*Lycaeides melissa samuelis*
Kirtland's Warbler	*Dendroica kirtlandii*
Lark Sparrow	*Chondestes grammacus*
Lark Bunting	*Calamospiza melanocorys*
Lazuli Bunting	*Passerina amoena*
Lewis' Woodpecker	*Melanerpes lewis*
Loggerhead Shrike	*Lanius ludovicianus*
Lynx	*Lynx lynx*
Meadow vole	*Microtus pennsylvanicus*
Mole skink	*Eumeces egregius*
Moose	*Alces alces*
Mountain lion	*Felis concolor*
Mountain goat	*Oreamnos americanus*
Mountain Bluebird	*Sialia currucoides*
Mourning Dove	*Zenaida macroura*
Mule deer	*Odocoileus hemionus*
Northern Flicker	*Colaptes auratus*
Northern flying squirrel	*Glaucomys sabrinus*
Northern Goshawk	*Accipiter gentilis*
Northern Harrier	*Circus cyaneus*
Northern red-backed vole	*Clethrionomys rutilus*
Northern Shoveler	*Anas clypeata*
Northern Bobwhite	*Colinus virginianus*
Northern Cardinal	*Cardinalis cardinalis*
Nuthatches	*Sitta* spp.
Ord's kangaroo rat	*Dipodomys ordii*
Ovenbird	*Seiurus aurocapillus*
Peregrine Falcon	*Falco peregrinus*
Pileated Woodpecker	*Dryocopus pileatus*
Pine Warbler	*Dendroica pinus*
Plains pocket gopher	*Geomys bursarius*
Pocket mouse species	*Perognathus* spp.
Pocket gopher species	*Thomomys, Geomys* spp.
Prairie dog	*Cynomys ludovicianus*
Prairie Falcon	*Falco mexicanus*
Pronghorn	*Antilocapra americana*
Rabbit	*Sylvilagus* spp.
Raccoon	*Procyon lotor*
Red squirrel	*Tamiasciurus hudsonicus*
Red fox	*Vulpes vulpes*
Red-bellied Woodpecker	*Melanerpes carolinus*
Red-breasted Nuthatch	*Sitta canadensis*
Red-cockaded Woodpecker	*Picoides borealis*
Red-eyed Vireo	*Vireo olivaceus*

Appendix A

Common name	Scientific name
Red-naped Sapsucker	*Sphyrapicus nuchalis*
Red-shouldered Hawk	*Buteo lineatus*
Red-spotted newt	*Notophthalmus viridescns*
Red-tailed Hawk	*Buteo jamaicensis*
Red-winged Blackbird	*Agelaius phoeniceus*
Ruffed Grouse	*Bonasa umbellus*
Sage Grouse	*Centrocercus urophasianus*
Sage Sparrow	*Amphispiza belli*
Sage Thrasher	*Oreoscoptes montanus*
Sand skink	*Neoseps reynoldsi*
Sapsucker species	*Sphyrapicus* spp.
Savannah Sparrow	*Passerculus sandwichensis*
Scrub-Jay	*Aphelocoma* spp.
Sharp-shinned Hawk	*Accipiter striatus*
Sharp-tailed Grouse	*Tympanuchus phasianellus*
Short-eared Owl	*Asio flammeus*
Shrew species	*Sorex* and *Blarina* spp.
Six-lined racerunner	*Cnemidophorus sexlineatus*
Snowshoe hare	*Lepus americanus*
Southern harvester ant	*Pogonomyrmex badius*
Spotted Owl	*Strix occidentalis*
Spruce budworm	*Choristoneura fumiferana*
Spruce Grouse	*Falcipennis canadensis*
Summer Tanager	*Piranga rubra*
Swainson's Thrush	*Catharus ustulatus*
Three-toed Woodpecker	*Picoides tridactylus*
Townsend's chipmunk	*Tamias townsendii*
Townsend's ground squirrel	*Spermophilus townsendii*
Tree Swallow	*Tachycineta bicolor*
Turkey Vulture	*Cathartes aura*
Upland Sandpiper	*Bartramia longicauda*
Vaux's Swift	*Chaetura vauxi*
Vagrant shrew	*Sorex vagrans*
Vole species	*Microtus*, *Clethrionomys*, and *Phenacomys* spp.
Western Bluebird	*Sialia mexicana*
Western fence lizard	*Sceloporus occidentalis*
Western harvest mouse	*Reithrodontomys megalotis*
Western Meadowlark	*Sturnella neglecta*
Western rattlesnake	*Crotalus viridis*
Western Screech-Owl	*Otus kennicottii*
Western Tanager	*Piranga ludoviciana*
White-eyed Vireo	*Vireo griseus*
White-headed Woodpecker	*Picoides albolarvatus*
White-tailed deer	*Odocoileus virginianus*
White-tailed Hawk	*Buteo albicaudatus*
White-throated woodrat	*Neotoma albigula*
Wild Turkey	*Meleagris gallopavo*
Wood Thrush	*Hylocichla mustelina*
Woodland salamanders	*Desmognathus aeneus*, *Desmognathus ochrophaeus*, *Eurycea wilderae*, *Plethodon jordani*
Woodrat species	*Neotoma* spp.
Wrentit	*Chamaea fasciata*
Yellow-rumped Warbler	*Dendroica coronata*

Appendix B: Common and Scientific Names of Plant Species

Names are from U.S. Department of Agriculture, Natural Resources Conservation Service (1999).

Common name	**Scientific name**
alder	*Alnus* spp.
alligator juniper	*Juniperus deppeana*
American beech	*Fagus grandifolia*
antelope bitterbrush	*Purshia tridentata*
Ashe's juniper	*Juniperus ashei*
balsam fir	*Abies balsamea*
barrel cactus	*Ferocactus* spp.
big bluestem	*Andropogon gerardii*
big sagebrush	*Artemisia tridentata*
bigleaf maple	*Acer macrophyllum*
bigtooth aspen	*Populus grandidentata*
bitter cherry	*Prunus emarginata*
black grama	*Bouteloua eriopoda*
black spruce	*Picea mariana*
blackgum	*Nyssa sylvatica*
blueberry	*Vaccinium* spp.
bluebunch wheatgrass	*Pseudoroegneria spicata*
bluegrass	*Poa* spp.
bluestem	*Andropogon* spp.
buckbrush	*Ceanothus cuneatus*
bur oak	*Quercus macrocarpa*
California red fir	*Abies magnifica*
cattail	*Typha* spp.
chamise	*Adenostoma fasciculatum*
Chapman oak	*Quercus chapmanii*
cheatgrass	*Bromus tectorum*
chinkapin oak	*Quercus muehlenbergii*
cholla	*Opuntia fulgida*
deerbrush	*Ceanothus integerrimus*
Douglas-fir	*Pseudotsuga menziesii*
eastern redcedar	*Juniperus virginiana*
eastern white pine	*Pinus strobus*
eastern hemlock	*Tsuga canadensis*
fir	*Abies* spp.
Gambel oak	*Quercus gambelii*
Geyer's sedge	*Carex geyeri*
giant sequoia	*Sequoiadendron giganteum*
grand fir	*Abies grandis*
gray birch	*Betula populifolia*
greenleaf manzanita	*Arctostaphylos patula*
hickory	*Carya* spp.
huckleberry (western species)	*Vaccinium* spp.
Idaho fescue	*Festuca idahoensis*
incense cedar	*Calocedrus decurrens*
Indian ricegrass	*Achnatherum hymenoides*
Indiangrass	*Sorghastrum nutans*
jack pine	*Pinus banksiana*
Jeffrey pine	*Pinus jeffreyi*
juniper	*Juniperus* spp.
loblolly pine	*Pinus taeda*
lodgepole pine	*Pinus contorta*

Appendix B

Common name	Scientific name
longleaf pine	*Pinus palustris*
manzanita	*Arctostaphylos* spp.
mesquite	*Prosopis* spp.
mullein species	*Verbascum* spp.
myrtle oak	*Quercus myrtifolia*
northern red oak	*Quercus rubra*
oak	*Quercus* spp.
oneseed juniper	*Juniperus monosperma*
paper birch	*Betula papyrifera*
pinegrass	*Calamagrostis rubescens*
pinyon pines	*Pinus cembroides*, *P. edulis*, *P. monophylla*
pitch pine	*Pinus rigida*
plains reedgrass	*Calamagrostis montanensis*
pond pine	*Pinus serotina*
ponderosa pine	*Pinus ponderosa*
prairie dropseed	*Sporobolus heterolepis*
pricklypear	*Opuntia* spp.
quaking aspen	*Populus tremuloides*
red alder	*Alnus rubra*
red huckleberry	*Vaccinium parvifolium*
red maple	*Acer rubrum*
red pine	*Pinus resinosa*
red spruce	*Picea rubens*
redwood	*Sequoia sempervirens*
rough dropseed	*Sporobolus clandestinus*
rough fescue	*Festuca altaica* (subspecies *F. hallii*, *F. campestris*)
runner oak	*Quercus margarettiae*
sagebrush	*Artemisia* spp.
salal	*Gaultheria shallon*
salmonberry	*Rubus spectabilis*
sand live oak	*Quercus geminata*
sand pine	*Pinus clausa*
saw palmetto	*Serenoa repens*
sawgrass	*Cladium* spp.
Scouler's willow	*Salix scouleriana*
sedge species	*Carex* spp.
shadbush	*Amelanchier arborea*
shortleaf pine	*Pinus echinata*
silverberry	*Eleagnus commutata*
Sitka spruce	*Picea sitchensis*
slash pine	*Pinus elliottii*
southern magnolia	*Magnolia grandiflora*
spruce species	*Picea* spp.
spurge species	*Euphorbia* spp.
subalpine fir	*Abies lasiocarpa*
sugar maple	*Acer saccharum*
sugar pine	*Pinus lambertiana*
sweetgum	*Liquidambar styraciflua*
thickspike wheatgrass	*Elymus macrourus*
thimbleberry	*Rubus parviflorus*
tobosagrass	*Pleuraphis mutica*
tuliptree	*Liriodendron tulipifera*
turkey oak	*Quercus laevis*

Appendix B

Common name	Scientific name
Utah juniper	*Juniperus osteosperma*
velvet mesquite	*Prosopis velutina*
vine maple	*Acer circinatum*
wedgeleaf ceanothus	*Ceanothus cuneatus*
western hemlock	*Tsuga heterophylla*
western juniper	*Juniperus occidentalis*
western larch	*Larix occidentalis*
western snowberry	*Symphoricarpos occidentalis*
western wheatgrass	*Pascopyrum smithii*
white fir	*Abies concolor*
white spruce	*Picea glauca*
whitebark pine	*Pinus albicaulis*
wild lupine	*Lupinus perennis*
willow species	*Salix* spp.
winterfat	*Krascheninnikovia lanata*
yellow birch	*Betula alleghaniensis*

Appendix C: Glossary

The definitions here were derived from the following: fuels and fire behavior from Agee (1993), Brown and others (1982), Helms (1998), National Park Service and others (1998), Ryan and Noste (1985); fire occurrence from Agee (1993), Johnson (1992), and Romme (1980); plant reproduction from Allaby (1992), Sutton and Tinus (1983); other terms from Lincoln and others (1998).

abundance: The total number of individuals of a species in an area or community.

climax: A biotic community that is in equilibrium with existing environmental conditions and represents the terminal stage of an ecological succession.

cohort: A group of individuals of the same age, recruited into a population at the same time; age class.

connectivity: Accessibility of suitable habitat from population centers. All patches of suitable habitat that can be reached and occupied are considered connected.

crown fire: Fire that burns in the crowns of trees and shrubs, usually ignited by a surface fire. Crown fires are common in coniferous forests and chaparral shrublands.

density: The number of individuals within a given area.

dominance (dominant): The extent to which a given species predominates in a community because of its size, abundance, or coverage.

duff: Partially decomposed organic matter lying beneath the litter layer and above the mineral soil. It includes the fermentation and humus layers of the forest floor (02 soil horizon).

duration of fire: The length of time that combustion occurs at a given point. Relates closely to downward heating and fire effects below the fuel surface as well as heating of tree boles above the surface.

fire cycle: Used in this volume as equivalent to **fire rotation.**

fire exclusion: The policy of suppressing all wildland fires in an area.

fire frequency: A general term referring to the recurrence of fire in a given area over time. Sometimes stated as number of fires per unit time in a designated area. Also used to refer to the probability of an element burning per unit time.

fire intensity: Used in this volume as equivalent to **fireline intensity.**

fire regime: General pattern of fire frequency, season, size, and prominent, immediate effects in a vegetation type or ecosystem.

fire return interval: Number of years between fires at a given location.

fire rotation: The length of time necessary for an area equal in size to the study area to burn.

fire severity: A qualitative measure of the immediate effects of fire on the ecosystem. Relates to the extent of mortality and survival of plant and animal life both above and below ground and to loss of organic matter.

fireline intensity: The rate of energy release per unit length of the fire front expressed as BTU per foot of fireline per second or as kilowatts per meter of fireline. This expression is commonly used to describe the power of wildland fires.

flame length: The length of flames in the propagating fire front measured along the slant of the flame from the midpoint of its base to its tip. Mathematically related to fireline intensity and the height of scorch in the tree crown.

fuel: Living and dead vegetation that can be ignited. For descriptions of kinds of fuels and fuel classification, see "Effects of Fire on Flora" in the Rainbow Series.

fuel continuity: A qualitative description of the distribution of fuel both horizontally and vertically. Continuous fuels readily support fire spread. The larger the fuel discontinuity, the greater the fire intensity required for fire spread.

fuel loading: Weight per unit area of fuel often expressed in tons per acre or tonnes per hectare. Dead woody fuel loadings are commonly described for small material in diameter classes of 0 to 1/4-, 1/4 to 1-, and 1 to 3-inches and for large material in one class greater than 3 inches.

ground fire: Fire that burns in the organic material below the litter layer, mostly by smoldering combustion. Fires in duff, peat, dead moss, lichens, and partly decomposed wood are typically ground fires.

herpetile: Amphibian or reptile.

ladder fuels: Shrubs and young trees that provide continuous fine material from the forest floor into the crowns of dominant trees.

litter: The top layer of the forest floor (01 soil horizon); includes freshly fallen leaves, needles, fine twigs, bark flakes, fruits, matted dead grass, and a variety of miscellaneous vegetative parts that are little altered by decomposition. Litter also accumulates beneath rangeland shrubs. Some surface feather moss and lichens are considered to be litter because their moisture response is similar to that of dead fine fuel.

mast: Fruits of all flowering plants used by wildlife, including fruits with fleshy exteriors (such as berries) and fruits with dry or hard exteriors (such as nuts and cones).

mean fire return interval: The arithmetic average of all fire intervals in a given area over a given time period.

mesic: Pertaining to conditions of moderate moisture or water supply.

mixed severity fire regime: Regime in which fires either cause selective mortality in dominant vegetation, depending on different species' susceptibility to fire, or vary between understory and stand replacement.

prescribed fire: Any fire ignited by management actions to meet specific objectives. Prior to ignition, a written, approved prescribed fire plan must exist, and National Environmental Protection Act requirements must be met.

presettlement fire regime: The time from about 1500 to the mid- to late-1800s, a period when Native American populations had already been heavily impacted by European presence but before extensive settlement by European Americans in most parts of North America, before extensive conversion of wildlands for agricultural and other purposes, and before fires were effectively suppressed in many areas.

rhizome: A creeping stem, not a root, growing beneath the surface; consists of a series of nodes with roots commonly produced from the nodes and producing buds in the leaf axils.

scatter-hoard: Seed cached in scattered shallow holes, a common behavior for kangaroo rats and pocket mice.

secondary cavity nester: Animal that lives in tree cavities but does not excavate them itself.

sere: A succession of plant communities leading to a particular association.

snag: A standing dead tree from which the leaves and some of the branches have fallen.

stand replacement fire regime: Regime in which fires kill or top-kill aboveground parts of the dominant vegetation, changing the aboveground structure substantially. Approximately 80 percent or more of the aboveground dominant vegetation is either consumed or dies as a results of fires. Applies to forests, shrublands, and grasslands.

succession: The gradual, somewhat predictable process of community change and replacement, leading toward a climax community; the process of continuous colonization and extinction of populations at a particular site.

surface fire: Fire that burns in litter and other live and dead fuels at or near the surface of the ground, mostly by flaming combustion.

top-kill: Kills aboveground tissues of plant without killing underground parts from which the plant can produce new stems and leaves.

total heat release: The heat released by combustion during burnout of all fuels in BTU per square foot or kilocalories per square meter.

underburn: Understory fire.

understory fire regime: Regime in which fires are generally not lethal to the dominant vegetation and do not substantially change the structure of the dominant vegetation. Approximately 80 percent or more of the aboveground dominant vegetation survives fires. Applies to forest and woodland vegetation types.

wildland fire: Any nonstructure fire, other than prescribed fire, that occurs in a wildland.

xeric: Having very little moisture; tolerating or adapted to dry conditions.

USDA United States Department of Agriculture

Forest Service

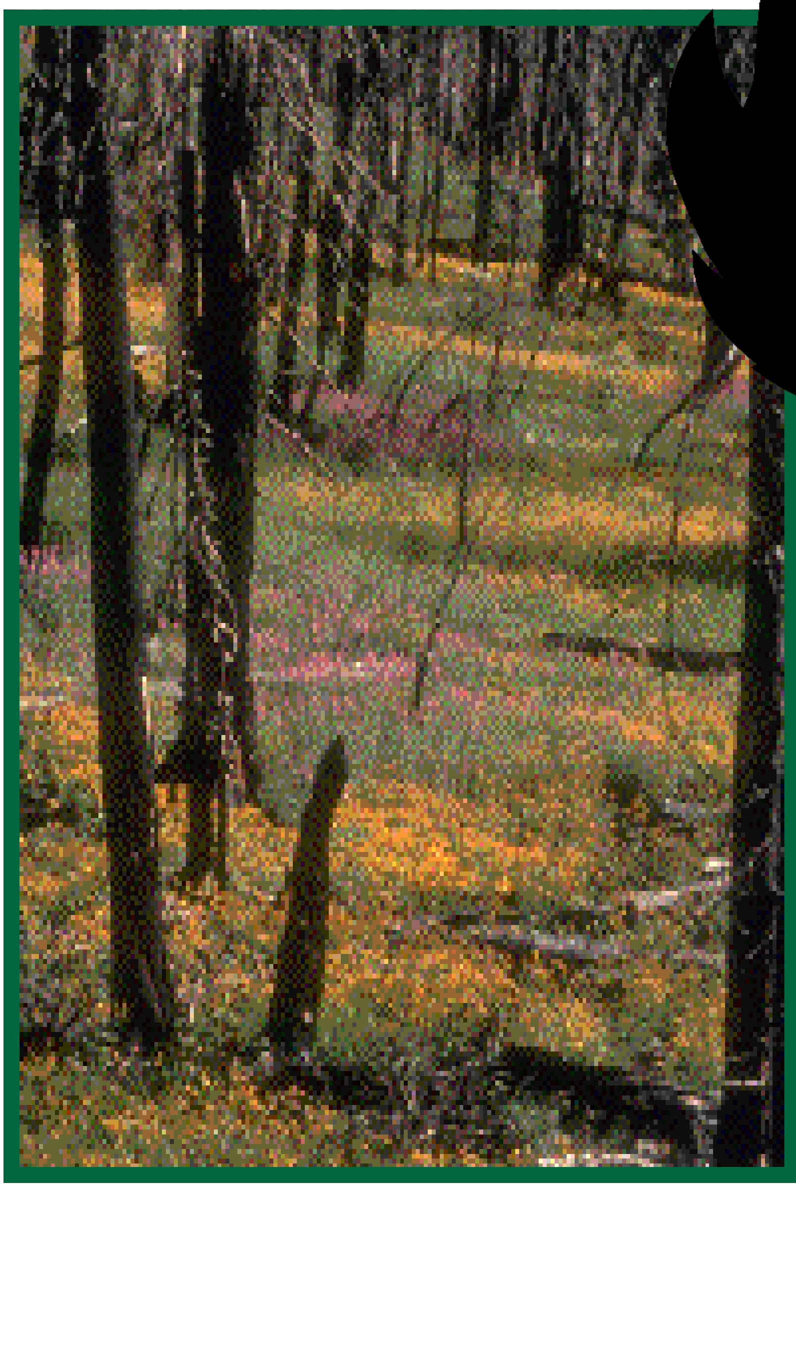
Rocky Mountain Research Station

General Technical Report RMRS-GTR-42-volume 2

December 2000

Wildland Fire in Ecosystems

Effects of Fire on Flora



Abstract

Brown, James K.; Smith, Jane Kapler, eds. 2000. Wildland fire in ecosystems: effects of fire on flora. Gen. Tech. Rep. RMRS-GTR-42-vol. 2. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 257 p.

This state-of-knowledge review about the effects of fire on flora and fuels can assist land managers with ecosystem and fire management planning and in their efforts to inform others about the ecological role of fire. Chapter topics include fire regime classification, autecological effects of fire, fire regime characteristics and postfire plant community developments in ecosystems throughout the United States and Canada, global climate change, ecological principles of fire regimes, and practical considerations for managing fire in an ecosytem context.

Keywords: ecosystem, fire effects, fire management, fire regime, fire severity, fuels, habitat, plant response, plants, succession, vegetation

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Wildland Fire in Ecosystems

Effects of Fire on Flora

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Cover photo—Arnica and fireweed flowers, Bob Marshall Wilderness, MT, 2 years after crown fire. Photo by Melanie Miller.

Preface

In 1978, a national workshop on fire effects in Denver, Colorado, provided the impetus for the “Effects of Wildland Fire on Ecosystems” series. Recognizing that knowledge of fire was needed for land management planning, state-of-the-knowledge reviews were produced that became known as the “Rainbow Series.” The series consisted of six publications, each with a different colored cover, describing the effects of fire on soil, water, air, flora, fauna, and fuels.

The Rainbow Series proved popular in providing fire effects information for professionals, students, and others. Printed supplies eventually ran out, but knowledge of fire effects continued to grow. To meet the continuing demand for summaries of fire effects knowledge, the interagency National Wildfire Coordinating Group asked Forest Service research leaders to update and revise the series. To fulfill this request, a meeting for organizing the revision was held January 4-6, 1993, in Scottsdale, Arizona. The series name was then changed to “The Rainbow Series.” The five-volume series covers air, soil and water, fauna, flora and fuels, and cultural resources.

The Rainbow Series emphasizes principles and processes rather than serving as a summary of all that is known. The five volumes, taken together, provide a wealth of information and examples to advance understanding of basic concepts regarding fire effects in the United States and Canada. As conceptual background, they provide technical support to fire and resource managers for carrying out interdisciplinary planning, which is essential to managing wildlands in an ecosystem context. Planners and managers will find the series helpful in many aspects of ecosystem-based management, but they will also need to seek out and synthesize more detailed information to resolve specific management questions.

— The Authors
October 2000

Acknowledgments

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Contents

Summary

This state-of-knowledge review about the effects of fire on flora and fuels can assist land managers in planning for ecosystem management and fire management, and in their efforts to inform others about the ecological role of fire. Chapter 1 presents an overview and a classification of fire regimes that is used throughout the report. Chapter 2 summarizes knowledge of fire effects on individual plants, including susceptibility to mortality of aerial crowns, stems, and roots; vegetative regeneration; seedling establishment from on-site and off-site seed sources; seasonal influences such as carbohydrates and phenological stage; and factors affecting burn severity.

Five chapters describe fire regime characteristics such as fire severity, fire frequency, and fire intensity, and postfire plant community responses for ecosystems throughout the United States and Canada. Typical fuel compositions, fuel loadings, and fire behavior are described for many vegetation types. Vegetation types including Forest-Range Environmental Study (FRES), Kuchler, and Society of American Foresters (SAF) types are classified as belonging to understory, mixed, or stand replacement fire severity regime types. The severity and frequency of fire are described for the pre-Euro-American settlement period and contrasted with current fire regimes. Historic fire frequencies ranged from a fire every 1 to 3 years in some grassland and pine types to a fire every 500 to 1,000 years in some coastal forest and northern hardwood types. In many vegetation types characterized by understory fire regimes, a considerable shift in fire frequency and fire severity occurred during the past century. Successional patterns and vegetation dynamics following disturbance by fire, and in some cases related grazing and silvicultural treatments, are described for major vegetation types. Management considerations are discussed, especially for the application of prescribed fire.

A chapter on global climate change describes the complexity of a changing climate and possible influences on vegetation, fuels, and fire. The uncertainty of global climate change and its interactions with vegetation means expectations for fire management are general and tentative. Nonetheless, manipulation of wildlands and disturbance regimes may be necessary to ensure continual presence of some species.

The last chapter takes a broader, more fundamental view of the ecological principles and shifting fire regimes described in the other chapters. The influences of fire regimes on biodiversity and fuel accumulation are discussed. Strategies and approaches for managing fire in an ecosystem context and sources of technical knowledge that can assist in the process are described. Research needs are broadly summarized.

James K. Brown

Chapter 1: Introduction and Fire Regimes

At the request of public and private wildland fire managers who recognized a need to assimilate current fire effects knowledge, we produced this state-of-the-art integrated series of documents relevant to management of ecosystems. The series covers our technical understanding of fire effects, an understanding that has expanded considerably since about 1980, along with an awareness that fire is a fundamental process of ecosystems that must be understood and managed to meet resource and ecosystem management goals. The "Rainbow Series" of documents stresses concepts and principles, provides an entry into the relevant literature, and discusses management implications. The volumes in the series are intended to be useful for land management planning, development of environmental assessments and environmental impact statements, training and education, informing others such as conservation groups and regulatory agencies, and accessing the technical literature.

Knowledge of fire effects has attained increased importance to land managers because fire as a disturbance process is an integral part of the concept of ecosystem management. Fire—a disturbance that initiates change—affects the composition, structure, and pattern of vegetation on the landscape. Disturbance is necessary to maintain a diversity of living things and processes. The old idea of plant communities and their broader ecological systems reaching equilibrium is being rejected by modern ecologists and resource managers (Botkin 1990; Morgan and others 1994).

Aldo Leopold (1949) recognized the principle of ecosystems decades ago, but it has only recently received widespread recognition as an important process to guide management of wildlands. Although the variety of governmental and private organizations responsible for management of natural resources disagree on an exact definition of ecosystem management, the goal of sustainability is central to most approaches (Christensen and others 1996). This goal focuses on delivery of goods and services. Ecosystem management defined by Christensen and others (1996) is *management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem structure and function.* This definition puts the primary focus on sustainability of ecosystem structures and processes necessary to deliver goods and services.

Ecosystem management broadens the focus of management from patches or stands to landscapes of variable scale. It moves the focus from individual ecosystem parts such as timber, water, range, fish, and wilderness, to how the parts fit together and function as a whole (Bormann and others 1994). It embodies other concepts such as conservation of biodiversity, sustained yield of multiple resources, and ecosystem health (Salwasser 1994). A guiding premise for sustaining ecosystems and protecting biodiversity put forth by Kaufmann and others (1994) is to manage ecosystems to conserve the structure, composition, and function of all elements, including their frequency, distribution, and natural extinction. Fire effects are woven through all aspects of this premise. An ecosystem can be defined as simply a place where things live (Salwasser 1994) or in more detailed terms that relate the interaction of organisms and physical environment through a flow of energy (Bormann and others 1994). Ecosystems contain components such as plants, vegetative communities, and landforms, and processes such as nutrient cycling. The dynamic nature of ecosystems and the scale of landscape patterns and processes are fundamental ecosystem characteristics that managers must consider in integrating knowledge of fire into the management of ecosystems.

Fire is a dynamic process, predictable but uncertain, that varies over time and the landscape. Fire has shaped vegetative communities for as long as vegetation and lightning have existed on earth (Pyne 1982). Recycling of carbon and nutrients depends on biological decomposition and fire. In regions where decay is constrained by dry and cold climates, fire plays a dominant role in recycling plant debris. In warmer, moist climates, decay plays the dominant role (Harvey 1994).

Lightning as a cause of fire over geologic time is widely appreciated. But humans also have been a major source of ignition, having used fire for various purposes during the past 20,000 years (Wright and Bailey 1982) and beyond. The pervasive influence of intentional burning by Native Americans during the past several centuries is probably not fully appreciated (Denevan 1992; Gruell 1985a). Human influence was particularly significant in grasslands and those communities bordering grasslands (Wright and Bailey 1982). Historically, fire caused by all ignition sources occurred over large areas covering more than half of the United States at intervals of 1 to 12 years; and fire occurred at longer intervals over most of the rest of the country (Frost 1998).

The land manager faces a complex challenge in managing fire to achieve beneficial effects and avoid unwanted results. Even attempts to eliminate harmful fire can over the long term cause undesirable consequences, such as increased risk of damaging fire and declining ecosystem health (Covington and others 1994; Mutch and others 1993). Thus, it is imperative that the immediate and long-term effects of fire be understood and integrated into land management planning.

Flora and Fuel Volume

The purpose of the Flora and Fuel volume is to assist land managers with ecosystem and fire management planning and in their efforts to inform others about the role of fire and the basis for including fire as an ecosystem management practice. The geographic area covered in this series volume includes Canada and all of the United States and adjoining Caribbean areas. The contents focus on principles, generalities, and broad scale fire effects on flora rather than on detailed site specific responses. Vegetative response to individual fires can vary substantially depending on a host of factors involving characteristics of the fire, existing vegetation, site conditions, and postfire weather. The value and indeed challenge in preparing this volume was in providing a summary of fire effects that was meaningful over broad areas even in view of highly variable responses. Those wishing a more detailed explanation of fire effects on flora are referred to several textbooks on the subject (Agee 1993; Johnson 1992; Wein and MacLean 1983; Wright and Bailey 1982).

Chapter 2 covers autecological effects of fire. Chapters 3 through 7 are about regional fire regime characteristics and postfire development of plant communities. Chapter 8 reviews the potential for climate change and implications for fire management. Chapter 9 provides an overview of the ecological principles underlying fire regimes, shifts in fire regimes, and related management considerations.

The regional fire regime chapters are organized by the following biogeographic regions:

- Northern ecosystems
- Eastern United States forests and grasslands
- Western forests
- Western shrublands, woodlands, grasslands
- Subtropical ecosystems

Each plant community chapter is organized by fire regime type (understory, mixed, stand-replacement) and similar subheadings. First, the fire regime characteristics are described, including fire severity, fire frequency, fire size and pattern, and fuels and fire behavior. This emphasizes the commonality of vegetation types that have similar fire regime characteristics based on the dominant vegetation undergoing similar structural changes. Next, postfire plant communities are discussed with emphasis on temporal changes in vegetation and fuels. Pre-1900 and post-1900 subheadings are often used to help distinguish between succession occurring before and after organized fire suppression. The 1900 date is an approximation of when fire suppression effectively reduced the extent of wildfire. In some vegetation types knowledge is insufficient to determine whether successional patterns differ between the two periods. Last, management considerations are described involving silvicultural practices, prescribed fire, ecosystem restoration, and other aspects of planned disturbance for maintaining healthy ecosystems.

The word "fuels" refers to live and dead vegetation that can potentially contribute to combustion. Fuel quantities can vary from a small portion to all of the aboveground biomass depending on a number of fuel properties especially particle size, moisture content, and arrangement. Although vegetation biomass increases predictably with time because of perpetual photosynthesis, changes in fuel biomass over time can be highly irregular due to the tradeoff between annual increment and decay and properties affecting fuel availability. In this volume fuels are described generally in terms of accumulation and flammability. Some information on fuel loadings is presented primarily to show typical values or a range in values that characterize various vegetation types. Fuel loading models for major vegetation types can be found in the First Order Fire Effects Model (FOFEM), which provides quantitative predictions of tree mortality, fuel consumption, smoke emissions, and soil heating (Reinhardt and others 1997). More detailed knowledge can be found in the literature referenced throughout this volume.

Plant community fire effects are discussed for broad vegetation types. Forest and Range Environmental Study (FRES) ecosystem types (Garrison and others 1977), which cover the 48 contiguous States, are used at the broadest scale. Society of American Forester cover types (Eyre 1980) are also used especially for Canada and Alaska, and where more resolution of fire effects knowledge is needed. FRES ecosystem types are based on an aggregation of Kuchler's (1964) Potential Natural Vegetation classes (American Geographical Society 1975). FRES, SAF, and Kuchler types that have similar fire regime characteristics are grouped to show synonymy (tables 3-1, 4-1, 5-1, 6-1, 7-1). Due to the subjective nature of defining vegetation types, some overlap occurs among types, particularly Kuchler and SAF types. For example, some SAF types may be reported in more than one Kuchler type. The correspondence of Kuchler and SAF types with FRES types is also described in the Fire Effects Information System User's Guide (Fischer and others 1996).

Scientific names of plant species referred to by common names throughout the volume are listed in appendix A. Appendix B describes the succession simulation models. And the glossary of fuel, fire, and plant reproduction terms is in appendix C.

Fire Regimes

"Fire regime" refers to the nature of fire occurring over long periods and the prominent immediate effects of fire that generally characterize an ecosystem. Descriptions of fire regimes are general and broad because of the enormous variability of fire over time and space (Whelan 1995). Classification of fire regimes into distinct categories faces the same difficulties and a dilemma that underlie any ecological classification. One difficulty is that putting boundaries around segments of biological processes that vary continuously involves some degree of arbitrariness. The dilemma is that for a classification to be useful to managers it must be practical and easily communicated, thus free of complexity. Yet to accurately reflect the nature of a biological process, such as response to fire, it must account for a complexity of interacting variables. A tradeoff between practicality and accuracy or between simplicity and complexity is required. The fire regime concept brings a degree of order to a complicated body of fire behavior and fire ecology knowledge. It provides a simplifying means of communicating about the role of fire among technical as well as nontechnical audiences.

Classifications of fire regimes can be based on the characteristics of the fire itself or on the effects produced by the fire (Agee 1993). Fire regimes have been described by factors such as fire frequency, fire periodicity, fire intensity, size of fire, pattern on the landscape, season of burn, and depth of burn (Kilgore 1987). The detail of a classification determines its best use. The more detailed classifications are primarily useful to ecologists and fire specialists attempting to describe and understand the more intricate aspects of fire. The simpler classifications are more useful for broadscale assessments and for explaining the role of fire to nontechnical audiences.

Heinselman (1978) and Kilgore (1981) produced the first classifications of fire regimes directed at forests. Two factors, fire frequency and intensity, formed the basis for their commonly referenced fire regime classifications (fig.1-1). A difficulty with fire intensity is that a wide range of intensities, including crown fire and surface fire, can cause stand-replacement because mortality to aboveground vegetation is complete or nearly complete. Fire intensity relates only generally to fire severity. Severity of fire reflects (1) the immediate or primary effects of fire that result from intensity of the propagating fire front and (2) heat released during total fuel consumption. Plant mortality and removal of organic matter are the primary fire effects. Kilgore emphasized fire severity in his modification of

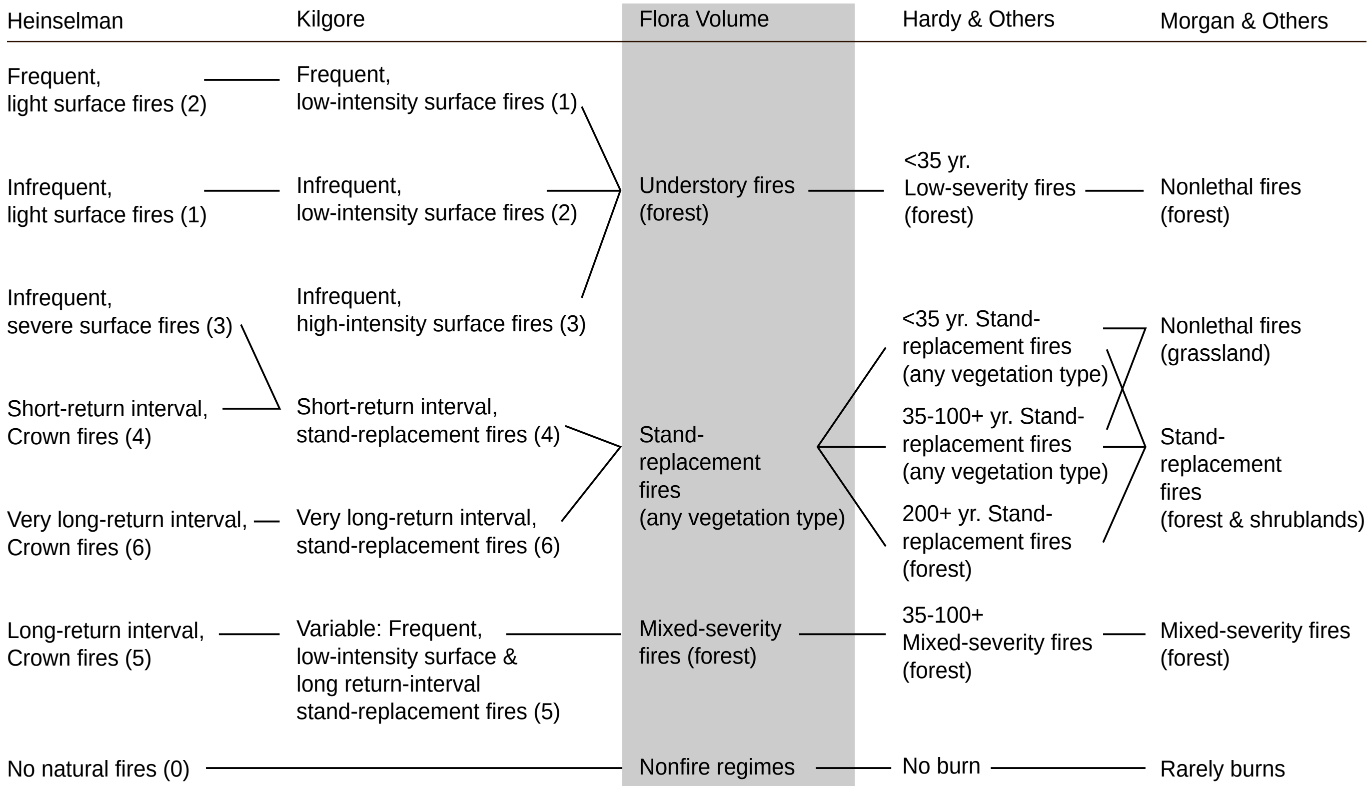

Figure 1-1—Comparison of fire regime classifications by Heinselman (1978), Kilgore (1981), Hardy and others (1998), Morgan and others (1998), and the Flora and Fuel Volume. Lines connect similar fire regime types. In parentheses, forest includes woodlands and grassland includes shrublands.

Heinselman's fire regimes by referring to mortality of the primary tree cover as stand-replacement.

Two recent fire regime classifications have proven useful for mapping extensive areas of forest, shrubland, and grassland vegetation at 1 km resolution. Morgan and others (1998) mapped historical and current fire regimes in the Interior Columbia River Basin based on four fire severity and five fire frequency classes (fig. 1-1). Hardy and others (1998) mapped fire regimes of the Western United States using fire severity and fire frequency combined into five classes. They keyed the fire regime classes to spectral images and biophysical data including elevation, hydrologic units, Kuchler's vegetation types, and Bailey's (1995) sections. Results were used to prioritize allocation of funds and resources as part of a national strategy for prescribed fire. For example, high priority for restoration using prescribed fire was assigned to areas where current and historical fire regimes have departed significantly such as in the ponderosa pine type.

Readers who wish to view a more complex ecological classification are referred to the detailed classification of fire regimes developed by Frost (1998). It incorporates periodicity of the fire cycle, primary season of burn, fire frequency, and fire effects on vegetation. It is very sensitive to fire frequency and its effects on understory herbaceous species. Sensitivity to frequency is provided by recognizing four frequency classes between 0 and 25 years. This separates some eastern and western vegetation into different fire regime types.

The fire regime classification employed in this volume is based on fire severity. Characteristic fire frequencies are reported but not combined with fire severity into classes. Use of fire severity as the key component for describing fire regimes is appealing because it relates directly to the effects of disturbance, especially on survival and structure of the dominant vegetation. It is intended for broadscale applications and for communication about fire's role among resource managers and others interested in natural resources.

Detailed information available about past fire regimes is based mostly on biophysical evidence, written records, and oral reports that encompass the period from about 1500 to late 1800, a time before extensive settlement by European-Americans in many parts of North America, before intense conversion of wildlands for agricultural and other purposes, and before fire suppression effectively reduced fire frequency in many areas. In this volume, we refer to the fire regimes of the past several centuries as "presettlement" fire regimes. The following describes the fire regime types used in the Flora and Fuel Volume:

1. *Understory fire regime* (applies to forests and woodlands)—Fires are generally nonlethal to the dominant vegetation and do not substantially change the structure of the dominant vegetation. Approximately 80 percent or more of the aboveground dominant vegetation survives fires.
2. *Stand-replacement fire regime* (applies to forests, woodlands, shrublands, and grasslands)—Fires kill aboveground parts of the dominant vegetation, changing the aboveground structure substantially. Approximately 80 percent or more of the aboveground dominant vegetation is either consumed or dies as a result of fires.
3. *Mixed severity fire regime* (applies to forests and woodlands)—Severity of fire either causes selective mortality in dominant vegetation, depending on different tree species' susceptibility to fire, or varies between understory and stand-replacement.
4. *Nonfire regime*—Little or no occurrence of natural fire.

In this volume, we consider all ecosystem types other than forest and woodland to have stand-replacement fire regimes because most fires in those ecosystem types either kill or remove most of the aboveground dominant vegetation, altering the aboveground structure substantially. Most belowground plant parts survive, allowing species that sprout to recover rapidly. This is true of tundra, grasslands, and many shrubland ecosystems. Morgan and others (1998) consider grasslands to have "nonlethal" fire regimes based on the criterion that structure and composition of vegetation is similar to the preburn condition within 3 years after a burn (fig. 1-1). Because fire radically alters the structure of the dominant vegetation for at least a short time, however, we consider grassland ecosystems to have stand-replacement fire regimes. Because grassland, tundra, and many shrublands are stand-replacement fire regime types, a more interesting aspect of fire regimes in these ecosystems is fire frequency, which can vary substantially and have a major influence on vegetation composition and structure.

The understory and mixed severity fire regimes apply only to forest and woodland vegetation types. The mixed severity fire regime can arise in three ways:

- Many trees are killed by mostly surface fire but many survive, usually of fire resistant species and relatively large size. This type of fire regime was described as the "moderate severity" regime by Agee (1993) and Heyerdal (1997).
- Severity within individual fires varies between understory burning and stand-replacement, which creates a fine-grained pattern of young and older trees. This kind of fire regime has not been recognized in previous classifications. It probably occurs because of fluctuations in weather during fires, diurnal changes in burning conditions, and variation in topography, fuels, and stand structure within burns (see chapters 5 and 6). Highly dissected terrain is conducive to this fire regime. In actuality, a blend of these two mixed severity types probably occurs.
- Fire severity varies over time with individual fires alternating between understory burns and stand-replacement. Kilgore (1987) described this as the "variable" regime and applied it to redwood forests. It also fits red pine forests (chapter 3).

The fire regime types were simplified from the classifications reported by Heinselman (1978) and Kilgore (1981). They are identical to the fire severity component utilized by Hardy and others (1998) except we use "understory" instead of "nonlethal" to depict that fire regime. We chose the term understory as a fire regime name because the term nonlethal is more easily misinterpreted when considering forest and nonforest ecosystems. Our fire regime classification is similar to that reported by Morgan and others (1998). To show how all of these classifications are related, equivalent or similar fire regime types are connected by lines in figure 1-1. The primary ecological knowledge imparted by fire regime types is whether fires leave the dominant aboveground vegetation standing and alive or result in stand-replacement. To reflect this, the fire regime types used in this volume, are characterized as nonlethal understory fire, stand-replacement fire, and mixed severity fire.

Fire severity is defined by what happens on areas that actually burned. In reality, unburned islands and patches of variable size and shape occur within the perimeter of fires. In studies of historical fire, it is difficult to separate burned from unburned patches. Thus, in applying the classifications, some nonlethal effects of fire can be attributed to unburned patches.

Forests of all types can be grouped into the understory, mixed, or stand-replacement fire regimes, which correspond to low, moderate, and high fire severity types described by Agee (1993). Some forest types occurring over a wide range of environmental conditions can fall into two fire regime classes. For example, most lodgepole pine and jack pine forests were characterized by stand-replacement fire. But some of the forests, typically on drier sites, reflect a mixed fire regime history. Evidence (Arno and others [in press]; Frost 1998) indicates that the mixed fire regime type was more prevalent than previously thought especially in coniferous forests. As fire moves across the landscape its behavior and effects can change dramatically due to variability in stand structure, fuels, topography, and changing weather elements. This can result in highly variable tree mortality and survival patterns within a fire's boundary. Generally, the severity and intensity of fire are inversely related to fire frequency (Swetnam 1993). For example, stand-replacement fires tend to occur in forests with low frequency, and understory to mixed severity fires tend to occur in forests with high fire frequency. Considerable variability exists within this generalization.

In this volume we consider grasslands and tundra fire regimes to be essentially all stand-replacement because the aboveground dominant vegetation is either killed or removed by fire. Also, many shrubland ecosystems are stand-replacement fire regime types because the dominant shrub layer is usually killed back to growing points in or near the ground. Stand-replacement fire in grass and sedge dominated ecosystems may be either lethal or nonlethal to aboveground vegetation. It is nonlethal if vegetative parts have already cured and exist as dead fuel, which is often the case in Western United States. But it is lethal if some of the aboveground grasses and sedges are living and are killed by fire as is commonly the case in marshes of eastern North America and in tundra. Fire is usually nonlethal to belowground plant parts allowing species that sprout to recover rapidly.

The natural role of fire can be understood and communicated through the concept of fire regimes. Significant changes in the role of fire due to management actions or possible shifts in climate can be readily described by shifts in fire regimes. It is increasingly recognized that knowledge of fire regimes is critical to understanding and managing ecosystems. To assist in this, fire regime types are identified for the major vegetation types in the United States and Canada (tables 3-1, 4-1, 5-1, 6-1, 7-1). The prevalence of each fire regime type within an ecosystem is characterized as being of major or minor importance. Fire frequency classes defined by Hardy and others (1998) are also tabulated along with a range in fire frequencies where there was sufficient knowledge.

To illustrate the extent and juxtaposition of various fire regimes across the landscape, presettlement fire regime types showing fire severity and fire return intervals were mapped for the lower United States (fig. 1-2). The mapping was based on a digitized atlas of Kuchler's Potential Natural Vegetation Types (Hardy and others 1998) and the fire regime types ascribed to the Kuchler types in chapters 3 to 7. In interpreting the map, keep in mind that Kuchler types represent broad classes; vegetative cover types and fire regime types can vary within the Kuchler types. In the figure legend, the overlapping of fire frequency classes such as 0 to 10 and 0 to 35 years means that the broader class encompasses more variability in fire return intervals and uncertainty of estimation.

The map illustrates the great expanse occupied by the short return-interval, understory fire regime type in the Eastern United States. It is important to note that much of the presettlement oak-hickory type was a savanna classified as forest having an understory fire regime, but it reasonably could have been classified as prairie having a short return-interval, stand-replacement fire regime. The pattern and frequency of the mixed fire regime type varies substantially between western conifers and eastern hardwoods. Although the mixed regime mortality is similar, the fire behavior and species fire resistance differ. Fires in conifers typically are more intense than in hardwoods, but conifers have a higher resistance to fire injury. Mapping of fire regime types and changes between current and historical periods can be useful for broad-scale fire management planning and for communicating with non fire managers about landscape fire ecology.

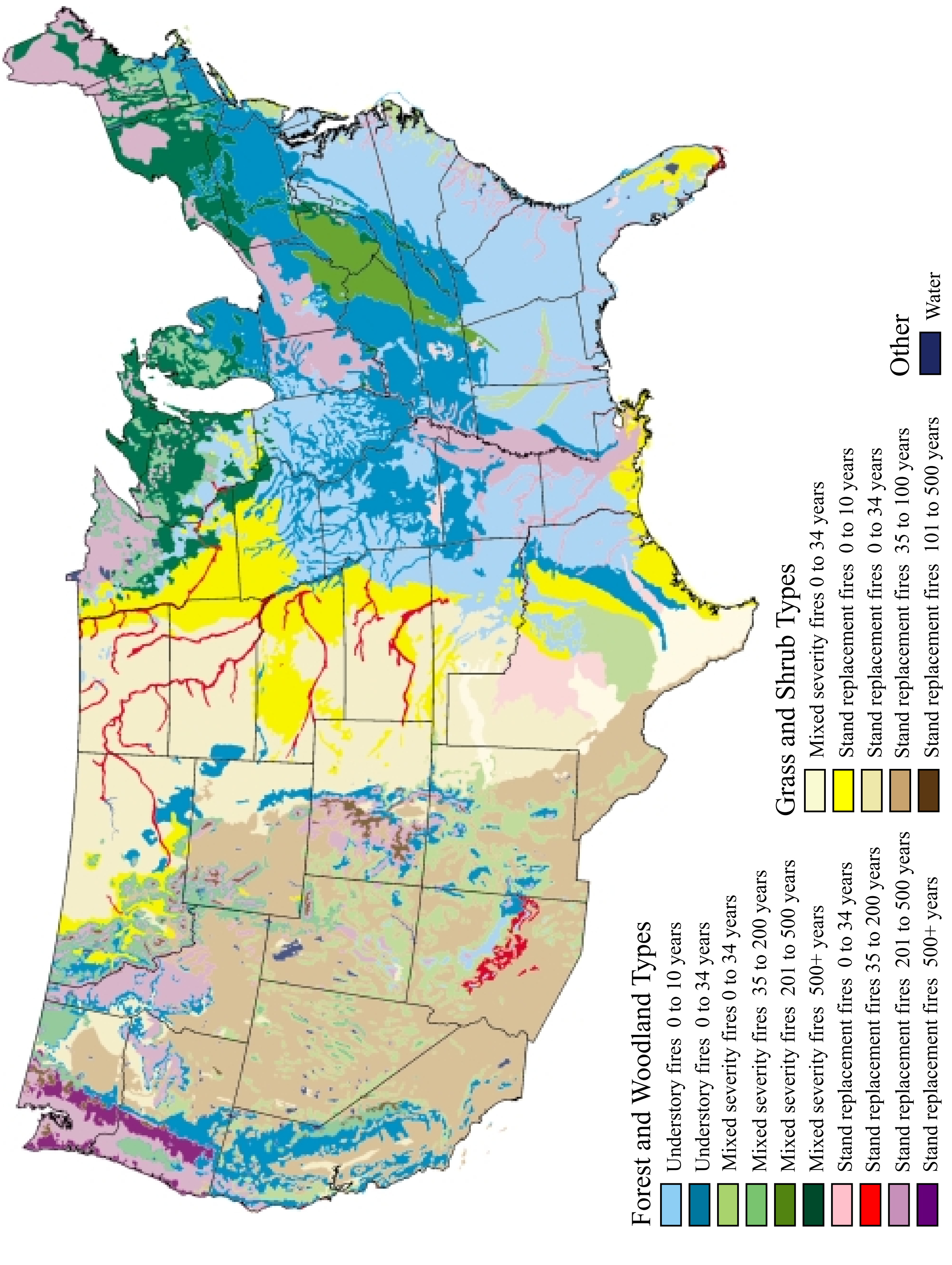

Figure 1-2—Fire regime types based on Kuchler's Potential Natural Vegetation types (prepared by Jim Menakis).

Notes

Melanie Miller

Chapter 2: Fire Autecology

Fire is a key ecological process within most ecosystems in the United States and Canada. An understanding of factors controlling the initial response of vegetation to fire is essential to its management. Fire effects on plants can vary significantly among fires and on different areas of the same fire. Fire behavior, fire duration, the pattern of fuel consumption, and the amount of subsurface heating all influence injury and mortality of plants, and their subsequent recovery. Postfire responses also depend upon the characteristics of the plant species on the site, their susceptibility to fire, and the means by which they recover after fire.

This chapter describes the key elements that explain fire effects on vascular plants, those plants with specific structures for gathering and transporting water and nutrients. Effects on mosses, lichens, liverworts, algae, and fungi are not discussed. The chapter addresses plant survival, resprouting, and seedling establishment in the initial stages of postfire recovery. Factors that affect a species presence or absence in the immediate postfire community will be described, but not those that affect productivity, such as changes in soil nutrient availability. The adaptations that allow survival, and the methods by which plants recover, are common to species found in almost all of the ecosystems discussed in this volume. The chapter describes principles in a general way, and provides specific examples from different ecosystems, although no attempt has been made to present examples from every system. Mechanisms operate in the same way no matter where they occur.

Plant Mortality

The likelihood of plant tissue being killed by fire depends upon the amount of heat it receives. The heat received by a plant is determined by the temperature reached and the duration of exposure. Most plant cells die if heated to temperatures between about 122 to 131 °F (50 to 55 °C) (Wright and Bailey 1982). Plant tissue withstands heat in a time-temperature dependent manner. Mortality can occur at high temperatures after a short period (Martin 1963), while death at lower temperatures requires a longer exposure (Ursic 1961). Additionally, some plant tissues, particularly growing points (meristems or buds) tend to be much more sensitive to heat when they are actively growing and their tissue moisture is high, than when their moisture content is low (Wright and Bailey 1982). The concentration of other compounds that vary seasonally such as salts, sugars, and lignins may also be related to heat tolerance of plants. Plant mortality depends on the amount of meristematic tissues killed. Susceptible tissue may not be exposed to heating by fire because it is protected by structures such as bark or bud scales, or is buried in duff or soil.

Plant mortality is often the result of injury to several different parts of the plant, such as crown damage coupled with high cambial mortality. Death may not occur for several years and is often associated with the secondary agents of disease, fungus, or insects. The resistance of plants to these agents is often lowered by injury, and wound sites provide an entry point for pathogens in conifers (Littke and Gara 1986) and hardwoods (Loomis 1973). A plant weakened by drought, either before a fire or after wounding, is also more likely to die.

Aerial Crown Mortality

A woody plant's structure affects the probability that the aboveground portion will be killed by fire. Important aerial crown characteristics include branch density, ratio of live to dead crown material, location of the base of the crown with respect to surface fuels, and total crown size (Brown and Davis 1973). Height enhances survival, as the aerial portions of small stature plants are almost always killed. Species of trees that self-prune their dead lower branches, such as red pine, are less likely to have a fire carry into their crowns (Keeley and Zedler 1998). Small buds are more susceptible to lethal heating than large buds because of their small mass (Byram 1948; Wagener 1961). Large buds, such as on some of the pines, are more heat resistant. The small diameter twigs and small buds of most shrub species make them fairly susceptible to fire. For conifers, long needles provide more initial protection to buds than short needles that leave the bud directly exposed to heat from the fire (Wagener 1961). Whether leaves are deciduous or evergreen affects crown survival in that deciduous trees are much less susceptible during the dormant than growing season.

In order for the aerial crown to survive fire, some buds and branch cambium must survive. For conifers with short needles and trees and shrubs with small buds, crown scorch is often equivalent to crown death because small buds and twigs do not survive (Wade 1986). The upper portions of the crown may survive on taller trees. Large buds shielded by long needles can survive fires that scorch adjacent foliage (Ryan 1990; Wade 1986). The large shielded buds of ponderosa pine, lodgepole pine, western white pine, and western larch can survive at a 20 percent lower height than that where foliage is killed (Ryan 1990). Crown consumption is a better indicator of crown mortality than scorch for fire-resistant conifers such as longleaf pine, which has long needles, large well protected buds, and thick twigs (Wade 1986). Crown characteristics that affect survival of trees after fire are listed in table 2-1.

The scorching of a tree crown is primarily caused by peak temperature heat fluxes associated with the passage of the flaming fire front (Van Wagner 1973). Long-term heating caused by burnout of fuel concentrations after the flaming front has passed can also scorch crowns. Whether the heat generated by fire is lethal to foliage also depends on the ambient air temperature (Byram 1958). For example, at a 90 °F air temperature without wind, the height of foliage scorch can be approximately 25 percent higher than it would be at 77 °F, because at higher air temperatures less additional heat is required to raise the foliage temperature to a lethal level (Albini 1976). Scorch is also affected by the degree to which heat is dissipated by wind (Van Wagner 1973). In western conifers, the percent of crown volume with scorched foliage is a better predictor of crown mortality than scorch height because it is a better measure of the amount of remaining live foliage (Peterson 1985). In southern pine species, nearly all trees can survive 100 percent crown scorch except during the fall when survival is about 95 percent (Wade 1985; Weise and others 1990). Heat-caused needle damage is detectable within a few days, sometimes within hours, and becomes more obvious over the next several weeks (Ryan and Wade 2000).

Stem Mortality

In fires where aerial crowns are not burned, trees and shrubs can be killed by girdling, caused by lethal heating of the cambial layer, the active growth layer just beneath the bark. Fire resistance of tree stems is most closely related to bark thickness, which varies with species, tree diameter and age, distance above the ground, site characteristics, and health and vigor of the tree (Gill 1995). Some species with thin bark have a fairly thick collar of bark at the base of the bole (Harmon 1984). The insulating quality of bark is also affected by its structure, composition, density, and moisture content (Hare 1965; Reifsnyder and others 1967), factors that vary among species. For example, among central hardwoods, bark of silver maple has a high specific gravity and thermal conductivity, and can transmit heat to cambial layers in less time than bark with a low specific gravity and conductivity, such as bur oak and eastern cottonwood (Hengst and Dawson 1994). Flame length (Brown and DeByle 1987), flaming residence time (Wade 1986), and stem char height (Regelbrugge and Conard 1993; Regelbrugge and Smith 1994) can be related to the amount of mortality of thin-barked trees. The cambium layer of thin-barked trees such as lodgepole pine and subalpine fir is usually dead beneath any charred bark (Ryan 1982). For Northwestern conifers in natural fuel situations, minimum bark thickness associated with consistent tree survival is about 0.39 inches (1 cm) (Ryan 1990). Wade and Johansen (1986) noted that bark as thin as 0.5 inch (1.25 cm) could protect young loblolly and slash pines during dormant season fires with low fireline

Table 2-1—Tree characteristics important to surviving fire and an overall species resistance to fire rating.[a]

Species	Basal bark thickness, mature trees	Branch density	Size of buds	Length of needles	Ability to regenerate vegetatively after fire	Size[b] when fire resistance is gained[c]	Fire resistance at maturity
Conifers							
Pines							
Digger pine	Medium	Low	Medium	Long	None	None	Medium
E.white pine	Thick	Low	Medium	Medium	None	Mature	Medium
Jack pine	Thin	Low	Medium	Short	None	None	Low
Jeffrey pine	Thick	Medium	Medium	Long	None	Pole	High
Loblolly pine	Thick	Medium	Medium	Long	Root crown[d]	Sapling	High
Longleaf pine	Thick	Medium	Large	Long	Root crown[d]	Seedling	High
Pinyon pine	Thin	Low	Large	Short	None	None	Low
Pitch pine	Thick	Medium	Medium	Medium	Root crown, Stump sprouts	Mature	Medium
Pond pine	Thick	Medium	Medium	Long	Root crown, Stump sprouts	Pole	High
Ponderosa pine	Thick	Medium	Large	Long	None	Sapling/Pole	High
Red pine	Thick	Low	Large	Medium	None	Pole	Medium
Rocky Mt. Lodgepole pine	Very Thin	Low	Medium	Short	None	Mature	Medium
Sand pine	Thin	Medium	Medium	Medium	None	Mature	Low
Shore pine	Thin	Medium	Medium	Medium	None	None	Low
Shortleaf pine	Thick	Medium	Medium	Medium	Root crown[d]	Sapling	High
Slash pine	Thick	Medium	Large	Long	None	Sapling	High
Sugar pine	Thick	Medium	Medium	Medium	None	Mature	Medium
Virginia pine	Thin	Medium	Medium	Short	None	None	Low
W. white pine	Medium	Medium	Medium	Medium	None	Mature	Medium
Whitebark pine	Very Thin	Medium	Medium	Medium	None	Mature	Medium
Firs							
Balsam fir	Thin	High	Small	Short	None	None	Low
Douglas-fir, coast	Very Thick	High	Medium	Medium	None	Pole/Mature	High
Douglas-fir, Rocky Mountain	Thick	High	Medium	Medium	None	Pole	High
Grand fir	Medium	High	Medium	Medium	None	Mature	Medium
Noble fir	Medium	Medium	Medium	Medium	None	Mature	Medium
Pacific silver fir	Medium	Medium	Medium	Medium	None	None	Low
Subalpine fir	Very Thin	High	Medium	Medium	None	None	Very Low
White fir	Medium	High	Medium	Medium	None	Mature	Medium

(con.)

Table 2-1—Con.

Species	Basal bark thickness, mature trees	Branch density	Size of buds	Length of needles	Ability to regenerate vegetatively after fire	Size[b] when fire resistance is gained[c]	Fire resistance at maturity
Junipers							
Alligator juniper	Thin/Med	Low	Small	Short	Root crown, Stump sprouts, Roots	Mature	Low/Med
E. redcedar	Thin	High	Small	Short	None	None	Low
Oneseed juniper	Thin/Med	Low	Small	Short	None	Mature	Low/Med
Utah juniper	Thin/Med	Low	Small	Short	None	Mature	Low/Med
W. juniper	Thin	Low	Small	Short	None	Mature	Low/Med
Other conifers							
Alaska-cedar	Very Thin	Medium	Small	Short	None	None	Low
Black spruce	Medium	High	Small	Short	None	Mature	Low/Med
Blue spruce	Thin	High	Medium	Medium	None	None	Low
Engelmann spruce	Thin	High	Medium	Medium	None	None	Low
Giant sequoia	Very Thick	Medium	Small	Short	None	Pole	Very High
Incense-cedar	Thick	High	Small	Short	None	Mature	Medium
Mt. Hemlock	Medium	High	Small	Short	None	None	Low
Redwood	Very Thick	Medium	Small	Short	Root Crown, Stump Sprouts	Sapling	Very High
Sitka spruce	Thin	Medium	Medium	Medium	None	None	Low
Tamarack	Medium	Medium	Small	Short	None	Mature	Medium
W. hemlock	Medium	High	Small	Short	None	None	Low
W. larch	Very Thick	Low	Small	Medium	None	Pole	High
W. redcedar	Thin	High	Small	Short	None	Mature	Medium
White spruce	Medium	High	Small	Short	None	Mature	Medium
Hardwoods							
Oaks							
Black oak	Thin/Med	—	—	—	Root crown, Stump Sprouts	Mature	Low/Med
Blackjack oak	Thin/Med	—	—	—	Root crown, Stump Sprouts	Mature	Low/Med
Blue oak	Thin	—	—	—	Root crown, Stump Sprouts	Mature	Low/Med
Bur oak	Medium	—	—	—	Root crown, Stump Sprouts	Mature	Medium

(con.)

Table 2-1—Con.

Species	Basal bark thickness, mature trees	Branch density	Size of buds	Length of needles	Ability to regenerate vegetatively after fire	Size[b] when fire resistance is gained[c]	Fire resistance at maturity
California black oak	Thin/Med	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Canyon live oak	Medium	—	—	—	Root crown, Stump sprouts	Mature	Medium
Gambel oak	Thin	—	—	—	Root crown, rhizomes	None	Low
Northern red oak	Medium	—	—	—	Root crown, Stump sprouts	Mature	Medium
Oregon white oak	Thin/Med	—	—	—	Root crown, Stump sprouts	Pole	Medium
Post oak	Thin/Med	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Southern red oak	Thin	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Turkey oak	Medium	—	—	—	Root crown, Stump sprouts	Mature	Medium
White oak	Thin/Med	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Other hardwoods							
American beech	Thin	—	—	—	Root crown, Stump sprouts Roots	None	Low
American elm	Thin/Med	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Aspen	Medium	—	—	—	Roots, Root collar	Mature	Low/Med
Basswood	Thin	—	—	—	Root crown, Stump sprouts	None	Low
Bigleaf maple	Thin	—	—	—	Root crown, Stump sprouts	None	Low
Bitternut hickory	Thin	—	—	—	Root crown, Stump sprouts	None	Low
Black cottonwood	Medium	—	—	—	Stump sprouts, Root crown	Mature	Low/Med
Blackgum	Thin	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Eastern cottonwood	Medium	—	—	—	Root crown, Stump sprouts	Mature	Low/Med

(con.)

Table 2-1—Con.

Species	Basal bark thickness, mature trees	Branch density	Size of buds	Length of needles	Ability to regenerate vegetatively after fire	Size[b] when fire resistance is gained[c]	Fire resistance at maturity
Honey mesquite	Thin	—	—	—	Root crown, Roots	None	Very low
Mockernut hickory	Thin	—	—	—	Root crown, Stump sprouts	None	Low
Pacific madrone	Thin	—	—	—	Root crown	None	Low
Paper birch	Medium	—	—	—	Root Collar	None	Low
Persimmon	Thin/Med	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Pignut hickory	Thin/Med	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Red alder	Thin	—	—	—	Stump sprouts	Mature	Low/Med
Red maple	Thin/Med	—	—	—	Root crown, Stump sprouts	Mature	Low/Med
Southern magnolia	Thin/Med	—	—	—	Root crown, Stump sprouts	Mature	Med/High
Sugar maple	Thin	—	—	—	Root crown (rarely)	None	Low
Sweetgum	Thin	—	—	—	Root crown, Stump sprouts	None	Low
Tanoak	Medium	—	—	—	Root crown, Stump sprouts	Pole	Medium
White ash	Thin	—	—	—	Root crown, Stump sprouts	None	Low
Yellow-poplar	Thin/Med	—	—	—	Root crown, Stump sprouts	Pole	Med/High

[a] The ratings of physical attributes are relative among the range of conditions observed for all tree species based on reviews of literature.
[b] Sizes are defined as follows: seedlings, <1 inch dbh; saplings, 1 to 4 inch dbh; poles, 5 to 10 inch dbh; mature, >11 inch dbh.
[c] Size when medium or high fire resistance is gained.
[d] For seedlings (loblolly, longleaf, and shortleaf pines) and saplings (loblolly and shortleaf pines). Shortleaf pine is a fairly strong sprouter; loblolly is weaker, and longleaf is the weakest of the three species (Wade 2000).

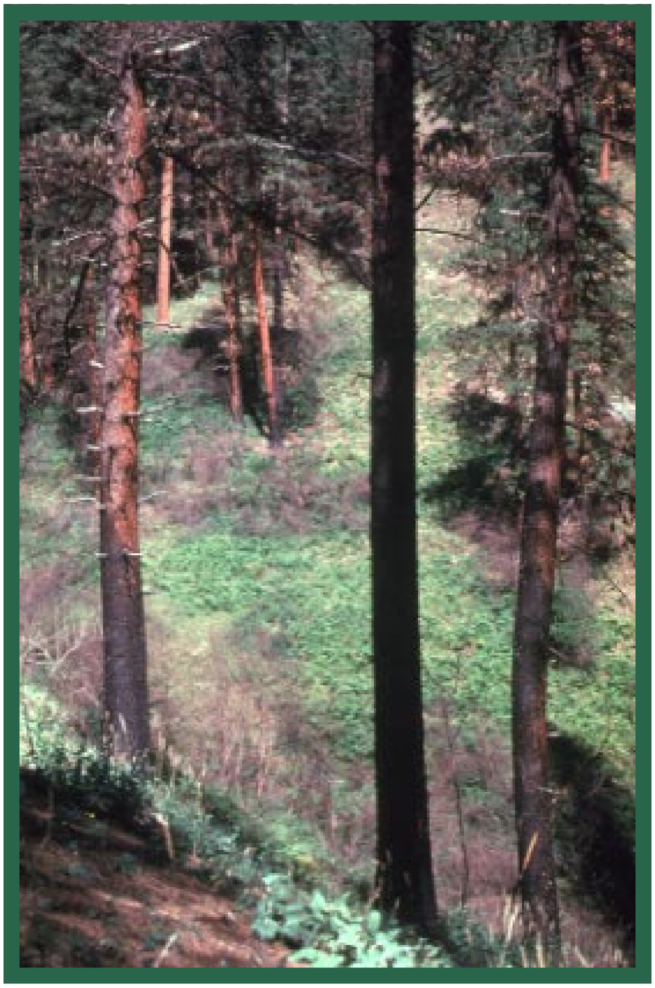

Figure 2-1—Scorched boles on surviving ponderosa pine, Selway-Bitterroot Wilderness, Idaho.

intensity. A summary of tree bark characteristics related to fire survival is in table 2-1.

Cambium that grows beneath thick bark layers typically found on mature Douglas-fir, western larch, and ponderosa (fig. 2-1), Jeffrey, longleaf, slash, and loblolly pines is insulated from heat released by the flaming front. However, the cambium can be killed by long-duration heating, such as from burnout of logs and smoldering combustion in deep litter and duff layers (fig. 2-2). Complete basal girdling is generally only caused by smoldering ground fires because the amount and distribution of dead woody fuels is rarely adequate to lethally heat the entire circumference of a thick-barked tree (Ryan and Reinhardt 1988). The deeper the basal mound of dry duff that is consumed, the more likely that tree cambium is killed (Harrington and Sackett 1992; Ryan and Frandsen 1991). In thick-barked trees, crown injury is more often the cause of mortality than bole damage (Ryan and Reinhardt 1988).

Fire scars occur where the cambium is killed and often are not evident until the dead bark sloughs from the tree (Smith and Sutherland 1999). Because charring doesn't happen unless the bark actually burns, charring often doesn't occur until a subsequent fire burns the exposed surface. Once tree cambium is injured by fire or mechanical damage, it is often more susceptible to additional fire scarring, both because the bark is thinner near the scar, and because of pitch that is often associated with wounds. Fire scars can become infected by wood-inhabiting microorganisms including decay fungi. The survival of chestnut and

Figure 2-2—Smoldering and glowing combustion in duff can lethally heat tree boles and roots such as in this Douglas-fir/western larch stand, Lubrecht Experimental Forest, Montana.

black oaks after surface fires in Eastern hardwood forests has been attributed to their ability to rapidly and effectively compartmentalize the wound, forming a boundary around the injured and decayed tissue that reduces the spread of infection (Smith and Sutherland 1999).

Many large hardwoods survive fire but have charred bark on the lee side, which in thin-barked species is a telltale sign that the underlying cambium has been killed. Even though the bark often remains intact for 1 or 2 years, the damaged sapwood begins to decay, reaching the heartwood in several years and then progressing upward at a more rapid rate. Height of decay is directly correlated to age of wound (Kaufert 1933). On fast-growing bottomland hardwoods, wounds less than 2 inches (5 cm) wide usually heal over before rot enters, but larger wounds are nearly always infected, ruining the butt log (Toole 1959). Decayed sapwood disintegrates rather quickly, creating the hollow found on many old growth hardwoods in the South. Most hollow trees also develop an enlarged buttress. Toole (1959) found that bottomland hardwoods that initially survive fire suffer considerable mortality over the next several years from breakage of decay weakened stems. Loomis (1973) presented methodology for predicting basal wound size and mortality to surviving trees in oak-hickory stands.

Root Mortality

Structural support roots growing laterally near the surface are more susceptible to fire damage than those growing farther beneath the surface. Roots found in organic layers are more likely to be consumed or lethally heated than those located in mineral soil layers. The locations of structural roots are summarized for important tree species in table 2-1.

Feeder roots collect most of a tree's water and nutrients, are small in diameter, and are usually distributed near the surface. Feeder roots located in organic soil layers are more subject to lethal heating and consumption than those located in mineral soil. Loss of feeder roots may be a more significant cause of tree mortality than structural root damage (Wade 1993). Feeder root death may not always kill the tree, but it can place the tree under significant stress. Increased amounts of root damage can result from fires that smolder in accumulations of litter beneath trees (Herman 1954; Sweezy and Agee 1991; Wade and Johansen 1986). This can be a critically important factor if most of the feeder roots are located in thick duff layers, caused by the exclusion of fire or a regime of dormant season prescribed burning that consumed hardly any surface organic matter. There may be enough root injury or death to kill trees and shrubs, even though little or no damage is apparent to their aerial crowns (Geiszler and others 1984). While tree crown mortality can be related to fireline intensity, mortality of buried plant parts depends much more on the duration of all phases of combustion that regulates the downward heat pulse, than on the duration of the flaming front (Wade 1986).

Fire Resistance

Tree resistance to fire generally increases with age. Crowns become larger and for some species, the height to the base of the live crown increases, either from self pruning or removal of basal branches by surface fires. Bark thickness and stem diameter increase. A suppressed tree may develop fire resistance characteristics at a much slower rate than a vigorous tree of the same age and species resulting, for example, in a much thinner bark in suppressed loblolly pine (Wade 1993). The growth stage at which important species of trees become fire resistant and the degree of resistance of mature trees are summarized in table 2-1.

Vegetative Regeneration

Sprouting is a means by which many plants recover after fire. Shoots can originate from dormant buds located on plant parts above the ground surface or from various levels within the litter, duff, and mineral soil layers (fig. 2-3). The type of plant parts that support dormant buds and where they are located in or above the soil are species-specific characteristics (Flinn and Wein 1977). The plant structures that give rise to regenerating shoots are summarized for different life forms of North American native plants in table 2-2.

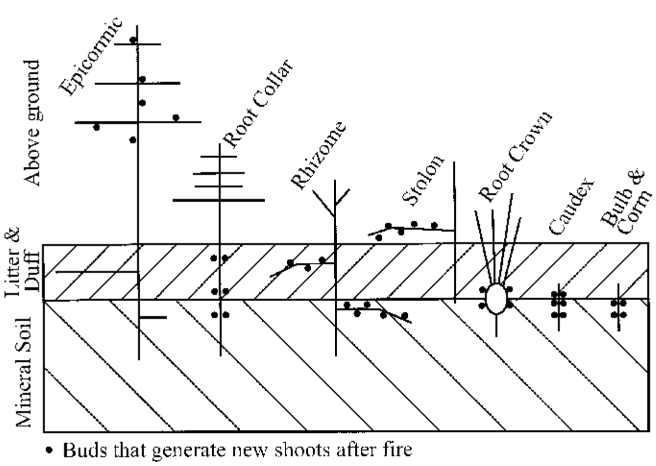

Figure 2-3—Various plant parts that regenerate new shoots and their location in and above the soil.

Table 2-2—Type and location of buds that regenerate new shoots after fire by broad plant groups (X = common, u = uncommon).

		Trees				Herbs	
Bud type	**Location**	**Conifer**	**Broad-leaf evergreen**	**Deciduous**	**Shrubs**	**Perennial forbs**	**Grasses**
Epicormic	Aerial	u	u				
Stolon	Above soil & duff					X	X
Root collar	In & above soil	u	X	X	X		
Root crown	In & above soil	u	X	X	X		X
Caudex	In soil or duff[a]					X	
Root	In soil or duff[a]		X	X	X		X
Rhizome	In soil or duff[a]				X	X	X
Bulb, corm	In soil					X	

[a] Budding organs may grow into duff after it accumulates to suitable depth.

Sprouting of Woody Plants

Dormant Bud Locations—Many woody plant species have dormant buds located in the tissue of stems, above or below the surface of the ground. These plants sometimes sprout from the root collar, the point where roots spread out from the base of the stem. Such species include antelope bitterbrush, bigleaf maple, rabbitbrush, mountain mahogany, turkey oak, northern red oak, and paper birch (table 2-1, fig. 2-4). Epicormic sprouts develop in species such as eucalyptus, pond pine, and pitch pine from buds buried in woody tissue of tree stems, or from bud masses present in branch axils. Lignotubers, burls, and root crowns are names for masses of woody tissue from which roots and stems originate, and that are often covered with dormant buds (James 1984). These buds may be deeply buried in wood, and may be located far below the surface if the tissue mass is large. Plants with this commonly occurring structure include white sage (Keeley 1998), chamise, willow, serviceberry, alder, and tanoak.

An unusual trait shared by pitch, pond, and shortleaf pines is the formation of a basal crook that enhances the ability of these species to produce basal sprouts when the stems are topkilled by fire. When seedlings are small, they fall over (presumably from their own weight), grow prostrate, and then resume vertical growth, which results in a basal crook at the soil surface (Little and Somes 1956). Primary needles with their axillary buds form just above the hypocotyl and just below the second bend of the crook (Stone and Stone 1954; Walker and Wiant 1966). Rootlets also form from the uppermost root tissue close to the bud cluster anchoring the stem in place. Buds on the lower side of the crook are thus well protected from fire. If fire topkills a seedling or sapling, these dormant buds sprout and the same growing process is repeated. Because sprouts originating after the second or third

Figure 2-4—Sprouts of paper birch that developed from root collar, Frenchman Lake, Alberta.

fire have a well-developed root system, their height growth is more rapid than that of the original seedling.

Dormant buds are often located on laterally growing stems or roots of woody plants. Some woody species, such as aspen and horsebrush, have dormant buds or bud primordia located along roots from which new shoots can originate. Rhizomes are the horizontal underground stems that have a regular network of dormant buds that can produce new shoots and adventitious roots (Welsh and others 1987; Zasada and others 1994). Woody rhizomatous species include blue huckleberry, bog blueberry, thimbleberry, white spirea, Gambel oak, creeping barberry, chokecherry, and Labrador tea (fig. 2-5).

Sprouting Process—Postfire sprouting in woody plants is a process that is regulated by the same factors that control vegetative regeneration after other types of disturbances. Consider the physiological interactions that produce new aspen shoots, a model summarized by Schier and others (1985) that likely applies to other plants with buried regenerating structures. The growth of most dormant buds or bud primordia is controlled by a phenomenon called apical dominance. Growth hormones, particularly auxin, a plant hormone manufactured in actively growing stem tips and adjacent young leaves, are translocated to dormant buds, which prevent them from developing into new shoots. If stem tips and leaves are removed, the source of growth hormones is eliminated. The balance of plant hormones within the buds changes. Growth substances in roots, particularly cytokinins, are translocated upward to the buds and can cause the dormant buds to sprout, or stimulate bud primordia to differentiate into shoots. Cytokinins may already be present in buds, and a decrease in the ratio of auxins to cytokinins provides the stimulus for bud outgrowth.

Fire initiates regeneration from buds by killing surface plant parts that inhibited their growth. The buds that become shoots are usually those nearest to the part of the plant killed by the fire. If dormant buds are destroyed, new buds may differentiate from wound tissue, called callus, and subsequently produce shoots (Blaisdell and Mueggler 1956). Once new shoots are actively growing, they produce growth hormones that are translocated to other dormant buds that are farther away from the point of damage, suppressing their growth (Schier 1972) (fig. 2-6).

The reduced understory cover and thickness of organic layers following fire can increase light near the surface, and in turn promote an increase in sprouting because light can cause rhizome tips to turn upward and develop leafy shoots once they reach the surface (Barker and Collins 1963; Trevett 1956). This possibility suggests that some postfire shoots may develop from rhizome tips, not dormant buds. Schier (1983) found that decapitating a rhizomatous plant caused laterally growing rhizomes to turn upward and become shoots. Additional rhizomes often form in response to vigorous aerial plant growth (Kender 1967), and may subsequently produce aboveground shoots (fig. 2-7). Sprouts from new rhizomes may recolonize areas where old rhizomes and other reproductive plant parts were killed by a fire. Plants may sprout soon after a fire, or not until the following spring if the fire occurs after the plants have become dormant (Miller 1978; Trevett 1962). Warmer soil temperatures following fire may enhance the amount of sprouting that

Figure 2-5—New shoot growth from a rhizome of Labrador tea, Seward Peninsula, Alaska.

Figure 2-6—Suppression of bud outgrowth farther down the stem by actively growing new shoot of blue huckleberry, Lubrecht Experimental Forest, Montana.

occurs (Zasada and Schier 1973). The initial energy required to support growth until the sprout is photosynthetically self-sufficient comes from carbohydrates and nutrients stored in the regenerating structures or in adjacent roots (James 1984).

Postfire sprouting ability can vary with plant age. Young plants that have developed from seed may not be able to sprout until they reach a certain age, which varies by species (Smith and others 1975; Tappeiner and others 1984). Older plants of some species may be able to produce few, if any, sprouts that survive. Older plants (80+ years) of other species such as pitch pine (Little and Somes 1956) can produce stump sprouts prolifically. Minimal postfire root sprouting such as documented in deteriorating aspen stands may be caused by a combination of root system dieback and continued inhibition of sprouting by residual stems (Schier 1975). The ability of selected tree species to vegetatively regenerate after a fire, and the structure from which the sprouts develop are summarized in table 2-1.

Sprouting and Burn Severity—Burn severity (also called depth of burn and ground char, see glossary) is a measure of the amount of fuel consumption and associated heating at and below the ground surface (also see fire severity in glossary). It is a function of the duration of the fire, and relates closely to the amount of surface fuel, litter and duff consumption, and their moisture content. Severity classes have been defined by Viereck and others (1979) and Ryan and Noste (1985). A strong relationship exists between subsurface heating and postfire sprouting in forested areas (Dyrness and Norum 1983; Miller 1977; Morgan and Neuenschwander 1988; Ryan and Noste 1985), and in rangeland shrubs (Zschaechner 1985), which can be related to the distribution of buried buds (Gill 1995). Figure 2-8 depicts the relationship between the

Figure 2-7—New rhizomes formed on postfire aster sprout that may colonize adjacent areas, East Kootenay Mountains, British Columbia.

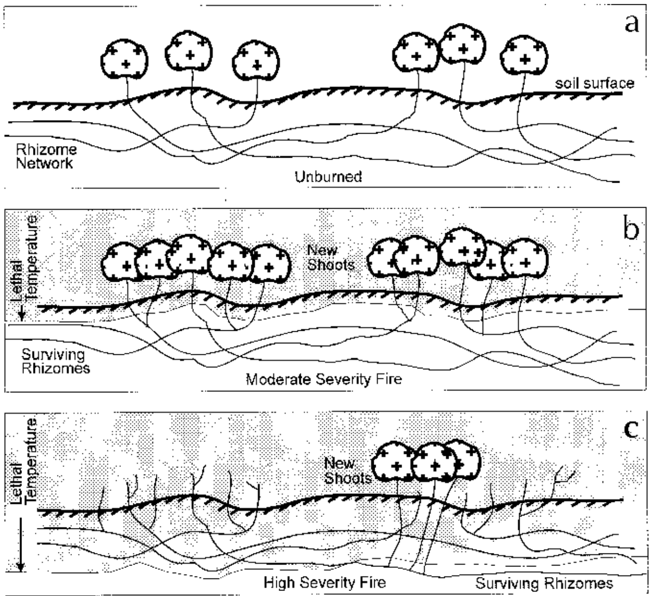

Figure 2-8—Effects of subsurface heating on postfire sprouting of a rhizomatous shrub: (a) Network of unburned rhizomes. (b) Moderate severity fire lethally heated the rhizomes near the surface, but many postfire sprouts grew from more deeply buried rhizomes. (c) High severity fire killed most of the rhizomes, but a few sprouts grew from a deep, surviving rhizome.

amount of postfire sprouting of a rhizomatous shrub and the depth of lethal heat penetration.

A low severity fire (lightly burned, short duration, low ground char) that only consumes some of the surface fuels may kill laterally growing rhizomes or roots near the surface, or stem buds that are not well protected. It has little effect on most buried plant parts and can stimulate significant amounts of postfire sprouting.

A moderate severity fire (moderately burned, moderate duration, moderate ground char) consumes the litter layer, and partially consumes both large woody debris and the duff layer. It incinerates plant structures in litter and the upper duff layer, such as shallow rhizomes, and may kill buds on portions of upright stems that are beneath the surface, and buds on the upper part of root crowns (fig. 2-9). Sprouting occurs from buds in deeper duff or soil layers. Johnston and Woodard (1985) found that mortality of rhizomes of beaked hazel and red raspberry only occurred in areas of relatively high surface fuel loading, but sprouting still occurred from more deeply buried rhizomes that survived the fire. Moderate severity fires frequently cause the greatest increase in stem numbers (fig. 2-8) of root sprouters such as aspen (Brown and Simmerman 1986) and of rhizomatous shrubs (Miller 1976). When heat prunes rhizomes below the surface where rhizome density is high (fig. 2-8b), shoots develop from buds along the rhizome, and appear as separate plants above the surface. Several shoots can replace what was previously one shoot. Other new shoots may develop from rhizome tips that are stimulated to turn to the surface and become shoots.

A high severity fire (heavily burned, long duration, deep ground char) removes the duff layer and most of the large woody debris, particularly rotten material. It can eliminate species with regenerative structures in the duff layer, or at the duff-mineral soil interface, and may lethally heat some plant parts in upper soil layers, particularly where concentrations of heavy fuels or thick duff layers are consumed. Any resprouting that does occur on heavily burned microsites can only occur from stolons and rhizomes that recolonize from adjacent areas or from deeply buried plant parts (fig. 2-8c). Miller (1977) observed that sprouting was delayed on a severely burned microsite, with a single huckleberry sprout not emerging until the third growing season from a rhizome about 9 inches below the surface. Abundant vegetative regeneration can still develop from species with deep roots such as aspen, or deep rhizomes such as Gambel oak.

Sprouting of Forbs

In North America north of the tropics, perennial forbs are broad leaved species that completely regrow their leaves and stems each year, after dying back during winter cold or summer drought. Some of these plants can persist in closed canopy environments that develop in the years after fire, producing a few leaves at the beginning of each growing season, which are often eaten by animals, or die back as the soils dry out (Christensen and Muller 1975).

Forbs have regenerative structures that are similar to those in woody plants, but also some that are unique. Stolons are stems of herbaceous species that grow on or near the surface of the ground, producing plants with roots at the node apex such as a series of strawberry plants (Benson 1957; Welsh and others 1987), and twinflower (McLean 1969). Dormant buds of fireweed and bracken fern are located on roots. Western yarrow, heartleaf arnica, showy aster, wild sasparilla, and star-flowered Solomon's seal are rhizomatous forb species.

A caudex is a largely underground, often woody stem base that persists from year to year and produces leaves and flowering stems (Benson 1957; Welsh and others 1987). This structure is found in species such as Indian paintbrush, lupine, wild columbine, and arrowleaf balsamroot. Other buried reproductive structures include bulbs (buried buds covered with thick fleshy leaves) found in common camas and deathcamas and corms (bulb-like, short thickened stems) of glacier lily and gayfeather species.

Figure 2-9—Sprouts originating from root crown of serviceberry, after a moderate severity fire killed buds on uppermost exposed surfaces, Piceance Basin, northwestern Colorado.

Leaves and stems suppress outgrowth of subsurface dormant buds. If fire occurs during the growing season, death of the apical meristems removes inhibition of subsurface buds and new shoots form. If a fire occurs when herbaceous plants are seasonally dormant, fire does not remove the source of inhibition because aboveground leaves and stems are cured and sometimes already decomposed. Dormant structures will grow new leaves after the occurrence of appropriate seasonal cues of temperature and moisture. Whether herbaceous plants recover after fire depends largely on whether their regenerative structures are exposed to lethal temperatures. Similar to woody plants, their survival depends on depth below the surface, whether they are located in combustible material, and the subsurface moisture regime at the time of the fire. While the stolons of strawberries make them susceptible to even low severity fire, the deep rhizomes of showy aster allow survival of some plants after fairly severe fires. Lupine and timber milkvetch can regenerate even when the entire plant crown is consumed (McLean 1969). Fireweed and bracken fern can produce significant numbers of sprouts after high severity fires because many buds far below the surface can survive even severe fire treatments (Frye 1934; Moss 1936). In California, 57 of 58 herbaceous perennial species resprouted after wildfire in chaparral (Keeley 1998).

Sprouting of Grasses

New leaf tissue of grasses forms at the meristems during the active growing period, and resumes after summer quiescence or winter dormancy. New growth also may occur by "tillering," branching from dormant axillary buds in the plant crown or on rhizomes. Grass plants are killed when all meristems and buds are lethally heated. Cool-season grasses that green up early in the growing season can be killed by the burning litter of associated warm-season grasses that are still dormant and more heat resistant (Anderson 1973). Perennial grasses also may be killed if fire burns in the cured litter of annual grasses while perennials are still actively growing (Wright and Klemmedson 1965).

Stoloniferous and Rhizomatous Grasses—Stoloniferous grasses are frequently killed by fire because most stolons are at or near the surface. Whether rhizomatous grasses are stimulated or killed by fire depends on rhizome depth below the surface, whether rhizomes are located in mineral or organic soil layers, the moisture content of these layers, and the amount and duration of heat generated by the surface fire. Rhizomatous grasses such as western wheatgrass often respond positively to rangeland fires because meristems and buds are usually protected by soil, and a long duration source of surface heat over large, contiguous areas is rarely present (Wright and Bailey 1982). In forested areas, grass rhizomes are more likely to be located in litter or duff layers or in association with dead woody fuels. On sites where fire consumes the duff layer, grass rhizomes located in the duff, such as pinegrass may be killed (S. A. Snyder 1991). However, some rhizomes often survive in deep mineral soil layers and can rapidly re-colonize severely burned areas.

Bunchgrasses—Meristems and dormant buds of different bunchgrass species can be located within the bunch above the level of the soil, or at various depths below the soil surface. Buds and meristems can be readily exposed to lethal temperatures, or be fairly well protected if deeply buried in unburned organic materials or in soil. For example, buds in the fairly compact root crown of Idaho fescue lie at or above the surface of the ground and are easily killed (Conrad and Poulton 1966), while basal meristems of bottlebrush squirreltail, Thurber's needlegrass (Wright and Klemmedson 1965), and wiregrass lie about 1.5 inches (4 cm) below the mineral soil surface and are more fire resistant (Uchytil 1992) (figs. 2-10, 2-11).

The moisture content of bunchgrass plants and adjacent fuels affect the amount of heat that the meristems receive. If plants are actively growing, their foliar moisture content can be too high to allow fire to enter the stand of plants. If bunchgrass crowns are moist, it is unlikely that they will ignite and burn. If the dead center of a bunchgrass plant is dry, it can ignite, smolder, and burn, killing most or all growing points. Heat from burning shrubs can dry and preheat adjacent bunchgrass clumps to ignition temperature, causing higher bunchgrass mortality than on a similar site with few shrubs that burned under the same conditions (Zschaechner 1985).

Dry bunchgrass crowns that are ignited are not always consumed. Midsummer fires in northwest Colorado burning under high windspeeds charred only the tops of the crowns of bluebunch wheatgrass and Indian ricegrass plants that were 4 to 5 inches (10 to 13 cm) in diameter. In this case, despite dry conditions, fire may have moved through the grass litter too quickly to ignite the root crowns, resulting in little plant mortality (Petersburg 1989).

Wright (1971) discussed the relationship between stem coarseness and the rate at which a bunchgrass clump burns. Fire tends to pass fairly quickly through coarse-stemmed bunchgrasses, which do not have much fuel concentrated at their base near reproductive structures. Fine-stemmed grasses with a dense clumping of basal stems can burn slowly and generate a fair amount of heat that can be transferred to meristems and buds. Fires tend to burn more rapidly through small-diameter bunches compared to large-diameter bunches. Larger bunches usually have more dead fuel and are thus more likely to produce enough heat to kill growing points (Wright and Klemmedson 1965). These relationships support Petersburg's (1989) observation that high rate of fire spread may have contributed to the survival of moderate diameter bunches of fine textured grass in northwestern Colorado that otherwise may have suffered fairly high mortality.

A rangeland bunchgrass grows where little surface fine fuel surrounds the plants, other than the dead grass blades immediately associated with the grass crown. The amount of postfire sprouting in this environment can be related to the amount of growing point mortality (Conrad and Poulton 1966). For those species having meristems above the mineral soil surface, the highest postfire sprouting potential usually is found in those plants with only some surface litter removed. Sprouting decreases as the amount of basal litter consumption increases, with new shoots tending to appear only from the outside edge of the bunch when little unburned stubble remains. Mortality is most likely to occur if all plant material above the root crowns is consumed.

Seedling Establishment

Seedling establishment is affected by the amount of seed present and conditions required to induce germination and provide a favorable environment for initial seedling growth. The interaction between the seed and its environment determines whether it successfully germinates and establishes. Requirements for successful germination and establishment can differ significantly among species.

Seed Supply and Dispersal

The supply of seeds of a given species is greatly influenced by annual seed production, which can vary significantly (Zasada and others 1978). Conifer regeneration may be limited if cone crops are poor during the time when exposed mineral soil seedbed is present. Surviving plants on or near the burned area may be too young (Barney and Frischknecht 1974; Zasada 1971)

Figure 2-10—Sand dropseed producing postfire growth from root crown meristems at the soil surface, Guadalupe Mountains National Park, western Texas

Figure 2-11—Basin wildrye growing after a fire from meristems below the soil surface, Bighorn Mountains, Wyoming.

or too old to produce much viable seed. Species of pines occurring in high fire frequency habitats generally begin producing cones earlier than other pine species (Keeley and Zedler 1998).

The timing of seed dispersal is a species-specific characteristic that varies with elevation and latitude (Zasada 1986). The occurrence of fire with respect to the dispersal of seeds can determine the rapidity of regeneration. Heat from fire may kill seeds that have recently fallen to the ground, preventing establishment of that species until after the next year's seedfall.

Seeds that are available to recolonize a burned site may have originated on-site or been dispersed from off-site after the fire. On-site seeds may come from surviving trees, from plants that grow after the fire, or from seed stored in the soil before the fire. The amount of off-site seed dispersal from unburned areas depends on the amount of available seed, the distance of the seed source from the burned area, the prevailing wind direction, and the type of seed. Seed dispersal mechanisms vary. Light seeds may be carried aloft while heavier seeds may skid across the surface of the snow. Some seeds have wing-like structures that enhance their movement through the air. Seeds with barbs or hooks may be carried in fur or feathers. Hard-coated seeds ingested along with their fruit pass through the bird or animal, sometimes with an enhanced likelihood of germination. Mature capsules of some species explosively release their seed (Parker and Kelly 1989). Animals and birds can disperse seeds at great distances from the parent plant, with Clark's nutcrackers (*Nucifraga columbiana*) observed to carry pinyon pine seed up to about 9 miles (Chambers and others 1999).

After dispersal, many seeds remain on or near the surface, although gravity, freezing and thawing, litterfall, and foraging activities of mammals, birds, and insects can deeply bury seeds (West 1968; Tomback 1986). Clark's nutcrackers, pinyon jays (*Gymnorhinus cyanocephalus*), squirrels, and mice cache a significant proportion of seed of certain species below the surface. There, seeds may exist within a matrix of soil, organic material, or a mixture of both.

Seedbank

The seedbank, the supply of seeds present on a site, is composed of transient and persistent seeds, which may be in litter and soil layers and in the tree canopy (table 2-3). There may be an enormous reserve of seed in the seedbank. Seed supply of various species and inherent seed longevity both affect the numbers of viable seeds. Some plants produce seeds that are a transient part of the seedbank, such as willow, which may remain viable for only a few weeks. There is little or no annual carryover of pine seed in soil seedbanks (Pratt and others 1984), and few conifer seeds are present in the forest floor of a mature forest (Archibold 1989; Ingersoll and Wilson 1990; Kramer and Johnson 1987). Many seeds, particularly large ones, are lost from the seedbank by predation.

Soil-Stored Seed—In a ponderosa pine community, viable seeds of most grass and annual forb species were found mostly in the litter layer, indicating short longevity or recent dispersal, while seeds of perennial forb species were found mostly in mineral soil, and probably were fairly long-lived (Pratt and others 1984). Seeds of some species persist in the soil for years after dispersal. Seeds of pincherry can survive in the seedbank for up to 100 years after the parent trees have died out of the overstory (Whittle and others 1997), while snowbrush ceanothus seeds remain viable for 200 to 300 years or more (Noste and Bushey 1987).

Species present in the seedbank of mature deciduous (Pickett and McDonnell 1989) and coniferous forests (Archibold 1989) are often shade-intolerant, early seral species, which may not be present in the overstory or understory. Few large seeded or shade tolerant species reside for long in the deciduous forest seedbank (Pickett and McDonnell 1989). In many grassland communities, there is a distinct difference between the species growing on the site and those present in the seedbank (Rice 1989). Seedbanks tend to contain more annual than perennial species, more forbs than grasses, many leguminous species, and more weedy species that colonize disturbed sites. In contrast, the chaparral seedbank generally reflects the composition of the standing vegetation with large, persistent seed banks of many species of dominant shrubs, although significant numbers of "fire-following" annuals may also be present (Parker and Kelly 1989). The life-forms of plants likely to have soil and canopy stored seed are shown in table 2-3.

Canopy-Stored Seed—Serotinous cones of species such as lodgepole, jack, pitch, Table Mountain, and pond pines retain some of their seeds because of the presence of a resin bond between scales on some of their cones. Serotinous cones slowly open and release their seeds after they are heated to at least 113 to 122 °F (45 to 50 °C), a temperature that melts the resin bond (Lotan 1976). Cones protect a significant portion of pitch pine seeds from the high temperatures reached during fire (Fraver 1992). Lodgepole pine seeds survived in cones heated in flames for a length of time typical of crown fires (Despain and others 1996). Numerous viable lodgepole pine seeds are dispersed even after a long duration crown fire. There is considerable variation in the amount of lodgepole cone serotiny, both on individual trees (fig. 2-12), and geographically. A high degree of cone serotiny is likely to occur where there are large, stand-replacement fires; relatively short, fire-free intervals; and fire sizes large

Table 2-3—Occurrence of transient seeds and persistent seeds consisting of soil and canopy stored and fire stimulated seed germination for broad plant species groups in the United States and Canada.

		Persistent seed		
Species group	Transient seed	Soil stored	Canopy stored	Fire stimulated
Conifer trees	X		X	
Broad-leaf evergreen trees	X			
Deciduous trees	X			
Shrubs	X	X		X
Annual forbs	X	X		X
Perennial forbs	X	X		X
Grasses	X			

enough to limit seed dispersal from unburned areas (Muir and Lotan 1985; Parker and Kelly 1989).

The semiserotinous cones of black spruce open and release their seeds over a period of years (Zasada 1986). Cones are usually bunched near the top of the tree, which shields some cones from heating and provides a postfire seed source. An additional on-site seed source from canopies may be immature cones that survive a fire and continue to ripen. This has been observed in ponderosa pines with scorched foliage (Rietveld 1976), in white spruce with boles completely girdled by fire (Zasada 1985), and in scorched cones of western larch and Douglas-fir where the tips of the seed wing had been singed (Stickney 1999).

Seed Environment

The seed environment describes the microsite in which a seed rests after it has been dispersed from the parent plant, including seedbed, temperature, humidity, shade, and potential competition from other plants. Moss, litter, and duff are poor seedbeds in many

Figure 2-12—Jack pine branch with serotinous and nonserotinous cones. Jack pine is ecologically similar to and hybridizes with lodgepole pine, Acadia National Park, Maine.

climates because they frequently dry out in the summer, resulting in seedling death if roots have not yet reached mineral soil. Organic seedbeds, even rotting logs, may be able to successfully support seedling establishment and survival if water is not limiting during the growing season (Zasada 1971). Other attributes of organic seedbeds such as the presence of allelopathic chemicals may inhibit seedling establishment. Oak litter has been observed to be a mechanical barrier to Table Mountain pine regeneration, although it enhances germination and survival of oak seedlings (Williams and Johnson 1992).

Fire creates significant changes in site conditions, which can vary substantially within the burned area depending on the severity and pattern of the fire. Consumption of fuel, especially the forest floor, is an important determinant of postfire conditions, because it controls the amount and distribution of good seedbed conditions. Where bare mineral soil seedbeds are created, any allelopathic chemicals are volatilized (McPherson and Muller 1969; Everett 1987b). Nutrients may be more readily available in ash, and the mineral soil does not dry out as readily as organic material. Moisture was more readily available at 12 inch depths beneath exposed mineral soil than below organic layers in late summer, allowing better growth of seedlings that can develop taproots such as ponderosa pine (Harrington 1992). The blackened surface causes warmer soil temperatures that enhance nutrient cycling and can favor growth, particularly in cold limited environments such as the boreal forest (Viereck and Schandelmeier 1980). After a severe fire, there is less competition from sprouting plants, seedlings, and trees if feeder roots and seeds stored in the duff and soil were killed. There may be little shade in the first few postfire years because of plant mortality. The length of time that a seed environment retains these characteristics after fire determines the number of postfire years that establishment of certain species from seed can take place (Shearer and Stickney 1991).

The physiological requirements of individual species determine whether postfire conditions are favorable for seedling establishment. For most species that develop from seeds dispersed after fire, the best seedbeds are microsites where most or all of the organic layer has been removed by fire because they provide the greatest chance for seedling survival. For some shade intolerant species, this is the only time that seedlings can establish, but these conditions can result in abundant regeneration, notably of western larch and many species of pine. Some perennial forbs resprout after fire, flower, and produce abundant seeds that establish in the second and subsequent postfire years (Keeley 1998). Wiregrass produces copious amounts of viable seed only after late spring and early summer burns.

If some residual organic matter remains, species with rapidly elongating roots may be favored over species that grow more slowly. Small seeded species are more likely to establish where little organic matter remains. Because seedlings originating from small seeds may be quite limited in their ability to grow through organic layers to mineral soil (Grime 1979), species with large seeds may be favored over small-seeded species where duff layers still exist.

In an Alaskan black spruce/feathermoss (Schreber's and mountain fern mosses) stand, germination and first year survival of black spruce and seven species of deciduous trees and shrubs occurred on both moderately and severely burned seedbeds. However, by the third year, seedlings survived almost exclusively on severely burned surfaces with no residual organic matter (Zasada and others 1983) (table 2-4).

Some species that establish from seed may be temporarily eliminated from a burned area because the postfire environment does not favor their establishment. In chaparral communities, species such as Nuttall's scrub oak, hollyleaf cherry, and toyon recover by sprouting after fire and thus remain on the site. However, seedlings of these species do not establish until the canopy closes and a deep litter layer forms (Keeley 1992).

Fire Stimulated Germination

Dormant seeds will not germinate when exposed to appropriate temperature and moisture conditions (Keeley 1995). Dormancy is maintained by environmental

Table 2-4—Seedling establishment on moderate and heavily burned seedbeds in an Interior Alaska black spruce forest.

	Number of germinants		Number of year 3 survivors	
Species	Moderately burned	Heavily burned	Moderately burned	Heavily burned
Alder	33	160	0	65
Paper birch	72	875	6	527
Balsam poplar	17	71	0	39
Bebb willow	105	144	0	46

conditions such as high and low temperature, low moisture, and inadequate amounts or quality of light (Baskin and Baskin 1989); or it can be imposed by an impermeable seed coat (Stone and Juhren 1953). Some species, such as chamise and hoaryleaf ceanothus, produce a proportion of seeds that remain dormant, while other seeds from the same plant will germinate under any suitable moisture and temperature conditions (Christensen and Muller 1975).

Fire can induce germination of dormant seeds of some species, resulting in an abundance of seedlings of these species in the first postfire year. Because essentially no seed germination occurs in subsequent years, annual plants flower, set seed, and are gone after the first year. Perennial seedlings that mature will flourish, depending on their inherent longevity, for as long as the site meets their specific environmental requirements. Eventually, they may persist only as seeds.

Germination of hard seeds can occur only after fire ruptures seed coat fissures or causes cracks to form in the seed coat, allowing water to enter (Keeley 1987; Rasmussen and Wright 1988). Requirements for optimum germination may be specific. Redstem ceanothus seed has the highest percentage of germination after exposure to moist heat at 176 °F (80 °C) (Gratkowski 1973), followed by stratification through a period of exposure to cold, wet conditions (Quick 1959). Fire stimulated germination has been documented for other hard-seeded genera including *Cassia*, showy partridgepea (Martin and others 1975; Tesky 1992); *Iliamna*, particularly wild hollyhock (Brown and DeByle 1989); *Lotus* or trefoil species (Keeley 1991); *Rubus* including blackberries and raspberries (Morgan and Neuenschwander 1988; Rowe 1983; Stickney 1986); *Ribes* such as gooseberry and currant (Lyon and Stickney 1976); and *Prunus* (Morgan and Neuenschwander 1988). Plant life-forms with fire-stimulated seed germination are shown in table 2-3.

Dormancy of species without hard seed coats can be broken by exposure to smoke and to chemicals leached from charred materials, although some species will germinate only in association with additional stimuli, such as cold stratification or burial (Keeley and Fotheringham 1998). This phenomenon has been studied most intensely in Mediterranean ecosystems, such as chaparral, and in Australia. Germination of chamise and many herbaceous species of the California chaparral can be induced by these treatments (Keeley 1991). Smoke exposure requirements varied significantly among species, with the duration of exposure that is optimum for germination for some species being lethal to others (Keeley and Fotheringham 1998). This suggests that seed germination in chaparral, both pattern and species, may be relative to different types of fire behavior and levels of fuel consumption, because these can result in significant variation in the amount and duration of smoke. Fire behavior may also relate to establishment of lodgepole pine. The greatest proportion of germination of lodgepole pine seeds from serotinous cones was enhanced by exposure of cones to a duration of flaming that most commonly occurs in crown fires (Despain and others 1996).

Seed germination for some chaparral species is adapted to wildfires that normally occur during fairly hot, dry late summer or fall conditions. Some seeds require dry heat to induce germination, but are killed by lower temperatures if they have imbibed moisture. Other seeds require higher temperatures for a longer duration to induce germination than generally occur under spring burning conditions (Parker 1989). If chaparral sites are burned under moist spring conditions, germination of both of these types of seeds is often much reduced. This is a particular concern for maintaining seedbanks of fire-following annuals, shrubs, and perennial forbs that can only reproduce from stored seed (Parker 1987a).

Burn Severity and Seed Regeneration

Variation in burn severity including its pattern and associated effects on seed mortality, seed stimulation, and seedbed quality can cause considerable variation in seedling numbers and species after a fire. The influence of burn severity on regeneration depends partly on where seeds are located. Viable seeds are characteristically present at different depths within the duff and soil profile. Seed produced by short-lived species that are stimulated to germinate by the heat of the fire, or that establish only on bare mineral soil, are usually found at the base of the forest floor layer, on top or near the surface of the mineral soil (Stickney 1991). Seeds of longer lived early seral species will be present near the duff mineral soil interface but will also have some vertical distribution within the duff layer. On sites without disturbance, these plants may have died out, and their seeds may not be present in the uppermost layers. Transient seeds are only present on and near the surface of the litter layer.

While fire kills most seeds within the surface litter layer, the temperature and duration of subsurface heating controls the amount of mortality and heat-stimulation of buried seed (Morgan and Neuenschwander 1988; Weatherspoon 1988). The pattern of severity relates to the pattern of fuel consumption and can cause variable mortality or stimulation of seeds around a burned site. Where little soil heating occurs, few heat-requiring seeds may germinate. Redstem ceanothus seedlings tended to occur on severely burned microsites within a matrix of less severely burned sites (Morgan and Neuenschwander 1988). If the lethal temperature isotherm penetrates below the level at which most duff and soil stored seeds occur, much less postfire plant establishment from the soil seedbank will occur than after a fire of more moderate severity (Weatherspoon 1988). However, there can still be a significant amount of regeneration from seedbank species where seeds in lower layers were heat stimulated but not lethally heated. Generally, the dryer the fuels, the more severe the fire, and the more seedbank mortality will occur. Fires of high burn severity also create more bare mineral soil seedbed, opening the site to colonization by seed dispersed from on-site or off-site after the fire. For those obligate, early seral species that only establish on bare mineral soil seedbeds, fires of high burn severity favor their regeneration.

Seasonal Influences

Carbohydrates

Carbohydrates, primarily starches and sugars, are manufactured by plants and provide energy for metabolism, and structural compounds for growth (Trlica 1977). Energy and material needed for initial plant growth following fire are provided by carbohydrates stored in undamaged plant parts, usually belowground structures. The timing of a fire, and its relationship to a plant's carbohydrate balance, can be a factor in postfire recovery because the rate and amount of regrowth is related to carbohydrate reserves (Trlica 1977).

The importance of carbohydrate reserves to plant regrowth after fire depends on survival of photosynthetically capable material, such as leaf blades and sheath leaves on grass stubble (Richards and Caldwell 1985). If some photosynthetic tissue remains or new tillers rapidly regenerate, newly grown leaf material soon manufactures all the carbohydrates that the plant needs for growth and respiration (Caldwell and others 1981). Evidence from clipping and grazing studies has shown that the recovery of grass plants is more related to the removal of growing points than to the carbohydrate level at the time of defoliation (Caldwell and others 1981; Richards and Caldwell 1985). However, fire may have a greater impact on grass plants than severe defoliation because it kills all photosynthetic material and elevated meristems. New growth must be supported by stored reserves.

There is a seasonal cycle of depletion and restoration of total nonstructural carbohydrates related to the growth cycle of the plant. The most rapid depletion usually occurs during periods of rapid growth, but carbohydrates may also be used for flower and fruit development, cold-acclimation ("hardening off" for winter), respiration and cellular maintenance during winter dormancy, and warm weather quiescence (Trlica 1977). Restoration of carbohydrates occurs when production by photosynthesis exceeds demands for growth and respiration. The timing of fluctuations in the annual cycle of total nonstructural carbohydrates (TNC) differs among species because of variability in plant growth cycles and growing season weather (Zasada and others 1994).

The limited survival of chamise sprouts after spring prescribed fires has been attributed to low winter and spring carbohydrate reserves because of high spring demand for growth, flowering, and fruiting (Parker 1987b). Number and dry weight of shoots of salmonberry were lowest on rhizome segments collected from May through July, which was also the seasonally lowest level of stored TNC (Zasada and others 1994). Salmonberry is most susceptible to physical disturbance during this time (Zasada and others 1994). For other species, the effects are most negative if the plant is burned late in the growing season because the plant uses a considerable amount of stored carbohydrates to sprout, but does not have enough time to restore reserves before winter dormancy (Mueggler 1983; Trlica 1977).

Severely burned chamise root crowns produced fewer sprouts than plants that experienced less heating, probably because more dormant buds were killed. Subsequent death of plants and limited sprouting may occur because insufficient carbohydrates are produced to sustain the root mass (Moreno and Oechel 1991). Root system dieback after excessive defoliation (Moser 1977) is considered to be a significant cause of plant mortality in grasses.

Repeated burning during the low point of a plant's carbohydrate cycle can increase any negative effects of treatment. Reduced density, canopy cover, and frequency of Gambel oak in southwestern Colorado, after two summer burns 2 years apart, were attributed to an inability to restore spent carbohydrate reserves for the 9 months after top-killing and resprouting (Harrington 1989). In the Southeast, annual summer burning nearly eliminated understory hardwood vegetation in a loblolly pine stand (Waldrop and Lloyd 1991). Burning when carbohydrates were low eventually killed or weakened root systems. Annual winter burning resulted in significant increases in numbers of small diameter sprouts on these same plots, because burning occurred when reserves were fairly high and sprouts had a full growing season to restore reserves before the next treatment.

If burning occurs in close association with heavy use of the plant community by livestock or wildlife, either before or after the burn, plant recovery may be delayed or prevented because of the excessive demand on stored reserves. Heavy postfire grazing or browsing of perennial plants in the first growing season after a fire is likely to cause the most harm, particularly in arid and semiarid range communities (Trlica 1977).

Flowering

Burning has long been used as a tool to enhance flower and fruit production of blueberry. Flowering of grasses such as pinegrass and wiregrass has been noted to increase significantly after burning (Brown and DeByle 1989; Uchytil 1992) (fig. 2-13). Burning during the growing season of April to mid-August causes profuse flowering of wiregrass in Florida, a marked contrast to a paucity of flowering that follows dormant season burning (Myers 1990b). Warmer soil temperature resulting from litter removal in these months may be the flowering stimulus (Robbins and Myers 1992). Increased light resulting from removal of the chaparral canopy stimulates flowering in golden brodiaea, a perennial forb that produces only vegetative growth in the shade (Stone 1951). This has also been observed in the Northern Rocky Mountains, in heartleaf and broadleaf arnicas, showy aster, and pinegrass. Increased availability of soil moisture and soluble nutrients also stimulates increased flowering.

In response to late spring fires, Henderson and others (1983) observed significantly greater flowering in big bluestem, little bluestem, sideoats grama, and Indian grass, all Wisconsin warm-season grasses. Increased flowering was attributed to higher levels of carbohydrate production caused by improved growing conditions, such as mulch removal. Grass flowering and seed production draw heavily upon carbohydrate reserves. Higher net photosynthate production was observed in big bluestem after spring burning. In contrast, cool-season grasses that were actively growing during late spring experimental fires showed a marked reduction in flowering, possibly because growth initiated after the fires further depleted carbohydrate reserves already drawn down by early growth. There also may have been more damage to meristematic tissue because plants were actively growing at the time of burning.

Fires enhanced flowering of dominant forb and shrub species in longleaf pine forests on the Florida panhandle (Platt and others 1988), with the most significant effects resulting from growing season fires. These fires increased the number of flowering stems, decreased the average flowering duration per species, and synchronized the period of peak flowering of

Figure 2-13—Abundant flowering of Thurber's needlegrass the first growing season after an October prescribed fire, near Carey, Idaho.

herbaceous plants, particularly fall flowering composites with a clonal growth form. Fire killed the elevated apical meristems, which no longer suppressed dormant buds on rhizomes, roots, and stolons. Multiple stems were initiated from these buds at times of the year when photoperiod strongly induced flowering. Dormant season fires had little effect on flowering periods because apical meristems were located at or below the ground surface, were little affected by fire, and continued to suppress secondary meristems the next growing season.

These mass flowering events are a means by which plants that regenerate after the fire redistribute themselves within the stand (fig. 2-14). For plants that germinate from soil stored seed, their profuse flowering in the first few years after fire resupplies the seedbank and ensures their presence after the next fire (Stickney 1990) (fig. 2-15).

Phenology

Plant growth stage at the time of a fire can result in different plant responses. Fire effects can vary substantially during a specific season, such as spring, because several phenological stages can occur in that 3 months. Phenology and the accompanying variation in plant condition, not season, leads to observed differences in plant response to fire. Phenological differences that affect plant responses to fire include varying levels of stored plant carbohydrate, presence of elevated herbaceous meristems that are more susceptible to fire because of their location, and presence of actively growing tissues that are more sensitive to high temperatures than when they are dormant or quiescent. The seasonal growth pattern that is characteristic of each species can be significantly modified by temperature and moisture in a specific year.

As an example of seasonal influences, ponderosa pine trees scorched in late October survived higher percentages of crown damage than trees scorched in early June and mid-August. The increased survival of fall burned trees was attributed to reduced physiological activity, lower bud tissue moisture contents, bud protection by fully developed bud scales and needles, and replenished carbohydrate stores, allowing adequate reserves to support spring shoot and root growth (Harrington 1987a, 1993).

Phenology also affects flammability. Moisture content of 1 year and older foliage of Western conifers is lower in the early part of the growing season than later in the summer (Chrosciewicz 1986; Jameson 1966; Philpot and Mutch 1971), and may contribute to higher spring crowning potential (Norum 1975). Seasonal differences in the moisture content of surface vegetation can determine whether the vegetation is a heat sink or is dry enough to be a heat source and thus

Figure 2-14—Postfire seed production by arnica, Bob Marshall Wilderness, Montana.

Figure 2-15—Flowering of wild hollyhock, developed from seeds stimulated to germinate by fire, Caribou National Forest, Idaho.

contribute to fire spread. Seasonal curing of herbaceous vegetation changes it from live to dead fuel.

Burning Conditions

Fuel and soil moisture conditions have a major influence on upward and downward heat flows that affect plant responses. Seasonal fluctuations in temperature and precipitation cause a progression of moisture content in dead woody fuels, litter, duff, soil organic layers, and soil. For a given vegetation and fuel type, burning conditions vary seasonally according to a general pattern; and the response of individual plant species to fires occurring under typical seasonal fuel and soil moisture conditions are fairly predictable based on their life-form.

Yearly variations in weather and associated departures from average moisture conditions can cause substantial variation in fire behavior and fire effects. For example, a winter and spring of above average precipitation results in wet woody fuels and duff in higher elevation forests that limit fire spread and fuel consumption. Below average precipitation in winter and spring can create dry enough conditions in forests to create potential for fires with high fuel consumption, and significant amounts of heat release both above and below the surface. Dry large fuels, duff, and mineral soil increase the potential for significant amounts of surface and subsurface heating, with concomitant mortality of roots, buried regenerative structures and seeds, and tree cambium.

Differences in seasonal weather in shrub/grass types can result in a large range in grass production, particularly annual species, which creates different fire behavior potentials. A wet year in the Great Basin leads to much more herbaceous biomass in sagebrush/grass communities, a greater likelihood of ignition, and larger sizes of fires that do occur. However, whether these higher fuel loadings relate to greater consumption of basal fuels and higher mortality of bunchgrasses and sprouting shrubs has not been documented.

The pattern of fire effects across the landscape varies with burning conditions. Areas of tree crown consumption, crown scorch, and little crown damage can be intermixed (fig. 2-16). Heavily burned areas of the forest floor where significant amounts of fuel were consumed and most buried plant parts were killed can be adjacent to areas where prefire fuel loading was low, and little subsurface heating occurred (fig. 2-17). On rangelands, the pattern can vary between areas of significant heat release associated with consumption of shrubs and accumulated litter and other areas where little heat was generated due to sparse fine fuels (fig. 2-18).

During a dry season, especially in a drought, a much higher percentage of forest canopy is apt to be scorched

or consumed. Lethal temperatures may be driven to greater depths because fuel and duff consumption is fairly complete. During a wet year or early in the year before significant drying has occurred, less canopy will be killed and consumed and few buried plant parts will be killed.

Discussion

Plant response to fire is a result of the interaction between severity of the fire and characteristics of the plants in the fire, both their inherent resistance to injury and ability to recover. Fuel quantity and arrangement, fuel moisture content, topography, windspeed, and structure of the plant community itself cause the lethal heat zone created by fire to vary significantly in time and space. Fire can cause dramatic and immediate changes in vegetation, eliminating some species or causing others to appear where they were not present before the fire. However, in burned areas with a high component of surviving trees and resprouting understory vegetation, within a few

Figure 2-16—Fire mosaic in a forest canopy, Selway-Bitterroot Wilderness, Idaho.

Figure 2-17—Burn pattern in a forest floor, with different plant species present on moderate and severely burned areas, Washington Creek, interior Alaska.

Figure 2-18—Burn pattern on rangeland, north of Boise, Idaho.

years it can be difficult to determine that a fire recently occurred.

For the vascular plant groups discussed in this chapter, the recovering plant community, in the first few years after a fire, comprises individuals from the following categories:

- Plants that survived the fire with their form intact
- Sprouts or suckers that grew from the base or buried parts of top-killed plants
- Plants that established from seed

Seedlings can be further described as:

- Plants that re-established from seed dispersed from surviving plants, usually trees
- Plants that re-established from seed dispersed from off of the burned site
- Plants that re-established from fire stimulated seed within the seedbank
- Plants that re-established from seed that developed on plants that resprouted after the fire

Certain species can only recover after fire by a single means. Some will only be present after fire if regenerative structures survive and produce sprouts, because their seedlings are unlikely to survive in postfire environments. Species of plants that cannot resprout after top-killing must establish from seed. However, some species can successfully recover from fire both by resprouting and by seedling establishment. Severity of the fire largely determines whether new plants are sprouts or seedlings.

Where fire top-kills plants that can vegetatively regenerate, sprouting will be a significant source of postfire vegetation. Where lethal temperature penetrates deeply enough to kill many regenerative structures, sprouting may be limited, but some buried seed may receive the proper stimulus to germinate and produce significant numbers of seedlings. A microsite that sustained lethal heat deeply enough to kill all stored seed probably had enough fuel and duff consumed to prepare areas of bare mineral soil seedbed. Reproduction on these sites occurs from seeds dispersed onto these burned surfaces. The availability of canopy-stored seeds depends on the height to which lethal temperatures reached into the tree crowns, while dispersal from wind-carried offsite seeds depends on distance and direction of prevailing winds. Surviving trees and resprouting plants may produce seeds that can establish within the next few growing seasons while suitable seedbed exists.

The immediate response of plants can differ within the same fire because of variations in the pattern of burn severity. For example, chamise and redstem ceanothus can sprout after a low to moderate severity fire treatment. High severity fires kill existing plants but they are replaced by new plants that develop from fire-stimulated seeds. The postfire community may contain both sprouts and seedlings of these species with the proportion related to the severity of the fire.

Postfire species composition is usually an assemblage of many of the species that were growing on the site and represented in the seedbank at the time of the fire. Vegetative regeneration is common to many species and can make a major contribution to the postdisturbance community (Ingersoll and Wilson 1990), contrary to the commonly held notion that seed reproduction is dominant. Resprouts from rhizomes, root crowns, or protected meristems can account for a substantial proportion of postfire recruitment (Lyon and Stickney 1976). Many of the seedlings present in the first few postfire years may have grown from seeds formed on resprouting species such as fireweed and heartleaf arnica. The only locations in which new species are likely to be added to the plant community are microsites that are severely burned and receptive to germination and establishment of seeds from species dispersed from off of the site (Stickney 1999).

How This Applies to Management

Knowledge of plant response to fire can be critical to successful application of prescribed fire. Designing fire prescriptions requires knowledge of fire behavior, fire severity, species survival mechanisms, and methods of postfire vegetation recovery. Fire prescriptions should describe a set of weather and fuel moisture conditions that will control the rate and amount of fuel consumption by different size classes. Properly conducted, the prescribed fire will result in the desired amount of mortality, injury, resprouting, and seedling establishment from target species. To ensure that a desirable range of plant species establishes after prescribed fire, it is helpful to acquire predictive means of assessing how different prescriptions can produce different fire and plant responses (Whittle and others 1997). A fire prescription that takes both the surface and subsurface heat regime into account, thereby regulating its severity, is most likely to achieve desired fire effects.

How can you learn about the effects of fire and postfire response for individual plant species? The Fire Effects Information System is an Internet database that contains specific information about 900 plants from the United States and Canada (see chapter 9).

The best way to learn about fire effects in a specific location is to visit areas recently burned under different fire prescriptions and fuel and soil moisture conditions. Answers to the following check-list questions can help you understand the type of fire treatment that will create the effects you desire:

Manager's Checklist

- Which species are present?
- How much injury did surviving plants sustain?
- Are there new plants of different species sprouts or seedlings?
- From what depths and structures did sprouting plants originate?
- Are seedlings rooted in organic or mineral substrate?
- Is there a pattern of sprouts and seedlings that is related to fuel and duff consumption?
- Is the species composition of seedlings related to canopy mortality or proximity to unburned areas?

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Chapter 3: Fire in Northern Ecosystems

Mixed Fire Regime

Major Vegetation Types

Major forest types include those where aspen, eastern white and red pine stands, and jack pine stands are found either as fire-maintained seral types or exceptionally as climax stands (see table 3-1 for FRES, Kuchler, and SAF cover type designations). This includes extensive areas from Newfoundland across to Alaska and the Great Lakes region. Fire regime characteristics varied extensively for these major vegetation types, reflecting local and regional topography, fuel type and climate. Fire regime characteristics (summarized in table 3-1) are discussed by first describing distribution and site features of the major vegetation types followed by discussion of the nature of fire.

Fire Regime Characteristics

Aspen—With its continental distribution, aspen is the most widely distributed forest type in North America extending from Newfoundland to Alaska, then southward through the western mountains of Canada and United States to Mexico (Eyre 1980; Farrar 1995). Aspen-dominated ecosystems are generally found as seral and more rarely as climax ecosystems.

Aspen occurs mostly as a pioneer type (fig. 3-1, 3-2) on burns or clear-cut areas on a wide variety of soil types excluding only the driest sands and the wettest swamps (Eyre 1980). In eastern Canada, aspen is found in association with sugar maple, balsam fir, speckled alder, eastern white pine, and paper birch (Eyre 1980). The transition area between the northern boreal forest and the central North American grasslands is dominated by aspen in central Canada where most even-aged stands have a fire origin (Jones and DeByle 1985). This forest type, frequent in Manitoba south of Lake Winnipeg and in North Dakota and northwestern Minnesota, occurs as well stocked stands in the northern portion of its range, and dwindles into patches surrounding moist depressions and streams farther south. Aspen is found in association with bur oak and in wetter locations with balsam poplar. On alluvial soils of eastern Saskatchewan and Manitoba, aspen is associated with white elm, green ash, Manitoba maple, and eastern cottonwood. Farther west, pure stands of aspen are found in association with the chernozen soil zones of Saskatchewan and Alberta (Corns and Anna 1986). Within the southern part of the grasslands, patches of aspen parkland are found in moist depressions and on bluffs and hills. In its western range, aspen is found most frequently as pure stands or in association with various conifers such as Engelmann spruce, lodgepole pine, ponderosa pine, and Douglas-fir.

Table 3-1—Occurrence and frequency of presettlement fire regime types by Forest and Range Environmental Study (FRES) ecosystems, Kuchler potential natural vegetation classes (1975 map codes), and Society of American Foresters (SAF) cover types. Occurrence is an approximation of the proportion of a vegetation class represented by a fire regime type. Frequency is shown as fire interval classes defined by Hardy and others (1998) followed by a range in fire intervals where data are sufficient. The range is based on study data with extreme values disregarded. The vegetation classifications are aligned to show equivalents; however, some corresponding Kuchler and SAF types may not be shown.

			Fire regime types						
			Understory		Mixed		Stand-replacement		
FRES	**Kuchler**	**SAF**	**Occur[a]**	**Freq[b]**	**Occur**	**Freq**	**Occur**	**Freq**	**Nonfire**
Aspen-birch 19	Aspen Parklands [c]	Aspen 16			M	2	M	2	
		Paper birch 252, 18					M	2	
W. aspen [d]	W. spruce-fir K015	Aspen 217			m	1,2	M	1,2	
White-red-jack pine 10	Great Lakes pine K095	Red pine 15			M	2			
		White pine 21			M	2			
		White pine-hemlock			M	2			
		White pine-red oak-red maple 20					M	1,2	
		Jack pine 1			m	1	M	2	
Spruce-fir 11	Great Lakes Spruce-fir K093	Balsam fir 5					M	2,3	
	Northeastern spruce-fir K096	White spruce 107					M	2	
		Red spruce 32					M	3	
		Red spruce-balsam fir 33					M	2	
		Paper birch-red spruce-balsam fir 35					M	2	
—	Black spruce [c]	Black spruce 12					M	2	
—	Conifer bog K094	Black spruce-tamarack 13					M	2	
		Tamarack 38					M	2	
—	Tundra [c]						M	2	

[a]M: major, occupies >25% of vegetation class; m: minor, occupies <25% of vegetation class.
[b]Classes are 1: <35 year, 2: 35 to 200 years, 3: >200 years.
[c]This type occurs primarily in Canada and was not defined by Kuchler.
[d]Added subdivision of FRES.

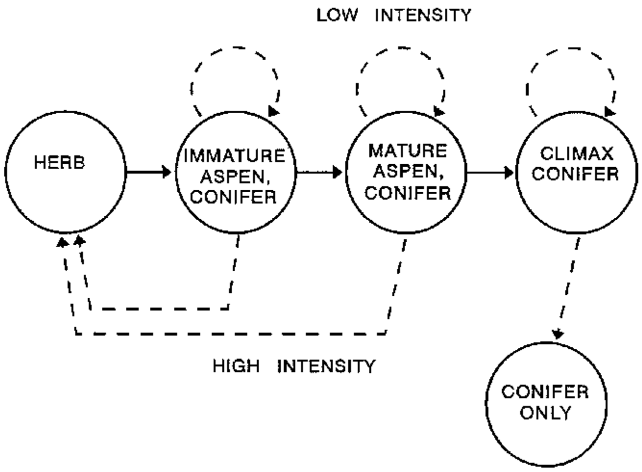

Figure 3-1—Successional pathways involving seral aspen, fire, and conifers (Brown 1985).

Pure aspen stands are particularly susceptible to mortality of aboveground stems from fire of low intensities, an unusual characteristic for a forest type so well adapted to regeneration after fire (Jones and DeByle 1985; Mutch 1970). Occasionally, fires of intensities greater than 30 Btu/ft/s (100 kW/m) have been shown not to damage mature trees (Quintillio and others 1989).

Paradoxically, aspen stands do not ignite easily and specific site and climatic conditions are necessary before fire can ignite and spread (Jones and DeByle 1985; Peterson and Peterson 1992; Wright and Bailey 1982). Generally, fires in young aspen stands are low intensity surface fires unless there is a great deal of fuel on the forest floor. In older stands, particularly those that are breaking up, abundant fuel can lead to higher intensity fires (Peterson and Peterson 1992).

White and Red Pine—Eastern white and red pine associations were generally fire-maintained seral types and existed occasionally as self-perpetuating climax under mixed fire regimes (fig. 3-3). The former include an extensive area associated with the Great-Lakes and St-Lawrence River ranging from Newfoundland to eastern Manitoba in Canada and New England, New York, Minnesota, Wisconsin, and Michigan in the United States (Eyre 1980; Farrar 1995; Roberts and Mallik 1994). Red and white pine were found in pure stands, or either species comprised the majority of stocking with jack pine, red oak, red maple, aspen, pin cherry, white spruce, and balsam fir. Pure red pine stands are mostly associated with dry, coarse soils whereas white pine tends to dominate in lighter structured soils (Eyre 1980).

Before fire protection, white and red pine stands were subjected to a mixture of lethal and nonlethal fires (Burgess and Methven 1977; Cwynar 1977, 1978;

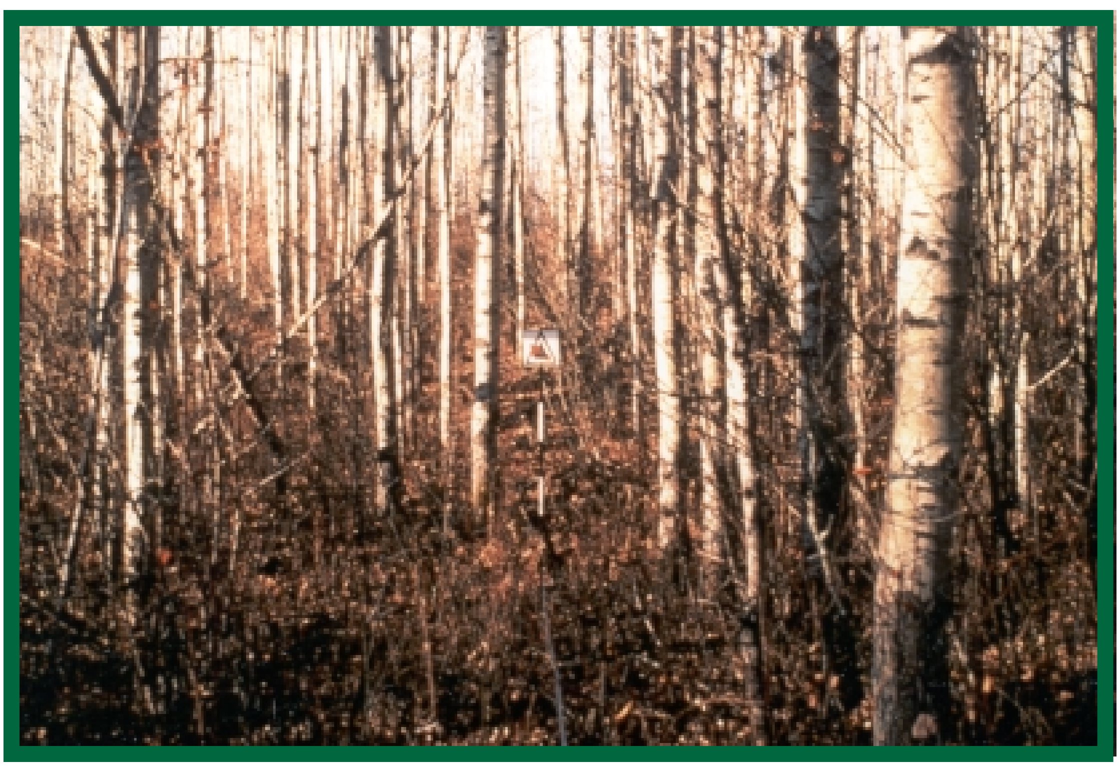

Figure 3-2—Seral aspen stand in Ontario (Canadian Fire Danger Rating Group photo point).

Figure 3-3—White and red pine stand in Ontario (Canadian Fire Danger Rating Group photo point).

Frissell 1973; Heinselman 1981; Maissurow 1935, 1941; Methven 1973; Methven and Murray 1974; Van Wagner 1970; Wendel and Smith 1990). Pine stands on dry sites underwent a cycle of light to moderate fires every 20 to 40 years, then high intensity fires every 100 to 200 years. On mesic sites, white pine experienced mostly high intensity fires. Postfire survival depended on fire behavior and tree age. This burning pattern for white and red pine stands led to a negative exponential age-class distribution with approximately 55 percent of forest stands older than 125 years, and the pattern applies to landscapes at least 960,000 acres (400,000 ha) (Baker 1989), suggesting a large proportion of so-called old growth forests.

On dry sites, fire frequency varied from 20 to 300 years with an average return interval of approximately 100 years for the entire Great Lakes and St. Lawrence forest region (Bergeron and Brisson 1990; Burgess and Methven 1977; Cwynar 1977, 1978; Duchesne and Gauthier; Frissell 1973; Heinselman 1981; Maissurow 1935, 1941; Methven and Murray 1974; Rowe 1972; Van Wagner 1970) and from 15 to 30 years in Newfoundland (Roberts and Mallik 1994). Mesic sites colonized mainly by white pine were characterized by longer fire intervals, probably 200 to 300 years or more (Heinselman 1983).

Jack Pine—Jack pine forests (fig. 3-4) are a widespread ecosystem type extending across Canada from Newfoundland to the Mackenzie river valley in the Northwest Territories (Eyre 1980; Farrar 1995), where they are found as fire-dominated pioneer stands. In the United States, jack pine stands are mainly found around the Great Lakes States, and less often in northern New England and New York. Typically, jack pine constitutes the majority of the stocking with several possible associate species. Associate tree species in the boreal forest are aspen, paper birch, balsam fir, and black spruce whereas, in the Great Lakes, they are northern pin oak, red pine, aspen, paper birch, balsam fir, and white pine (Eyre 1980). In general, jack pine stands are found on eskers, sand dunes, rock outcrops, clays, and organic soils. On the better sites, they exist primarily as pioneer stands but they tend to persist on xeric sites that are periodically visited by fire. Eyre (1980) recognized six variant types of jack pine:

1. Jack pine-balsam fir-black spruce in the mixed wood and hardwood forest zones of the boreal forest.
2. Jack pine-feather moss on clay and sands of the boreal forest.
3. Jack pine-sheep laurel on coarse sand and rocky outcrops of the mixed wood and coniferous forest zones.
4. Jack pine-Sphagnum on poorly drained organic deposits of the mixed wood and coniferous forest zones.
5. Jack pine-bog labrador tea on well-drained deposits of a different nature in the coniferous forest zone.

Figure 3-4—Mature jack pine stand (Canadian Fire Danger Rating Group photo point).

6. Jack pine-lichen in the taiga on various kinds of deposits.

Fires in jack pine forests, as on most of the boreal forest, were dominated by crown fires or high intensity surface fires that cover broad areas (Van Wagner 1983) because of the climate, topography, and continuity of fuel. Fires were frequent on dry sites with average return intervals as low as 15 to 35 years in eastern Canada (Bergeron 1991; Heinselman 1983) and the Great Lakes. Fire intervals were greater in the West (Heinselman 1983). Exceptionally, jack pine is submitted to nonlethal understory fire regimes in insular situations (Bergeron 1991).

Fuels

Aspen—Aspen stands are only flammable in the spring, late summer, and fall when they are leafless due to the drying effect of sun and wind on the leaf litter. Furthermore, in the fall the herbaceous plant and shrub component of the understory is dead and dried out, forming a continuous layer of loosely organized fine fuel. In general flammability depends largely on the amount of herbaceous and shrub fuels present in the stand. Herbaceous fuels can vary from as little as 0.04 tons/acre to 0.75 tons/acre (0.08 t/ha to 1.60 t/ha). Shrub fuels range from 0 to over 2.25 tons/acre (0 to 4.80 t/ha) (Brown and Simmerman 1986). Forest floor litter layer typically ranges from 0.38 to 2.9 tons/acre (0.80 to 6.60 t/ha) (Brown and Simmerman 1986; Quintilio and others. 1989).

White and Red Pine—White and red pine stands are fire-prone. Van Wagner (1970) reports that high density red pine stands are the most flammable fuel type of all northern cover types. The organic layer of white and red pine stands is generally 4 to 6 inches (10 to 15 cm) deep (Forestry Canada Fire Danger Group 1992). Light to moderate fires consumed the litter and shrubs along with invading shade tolerant conifers, but only scarred trees older than 30 to 75 years (Heinselman 1983), whereas high intensity fires killed most trees. In addition to age, white and red pine survival to fires of all intensity varied according to rooting, fuel loading, density, and fire behavior. Trees with more than 75 percent crown damage were more likely to die within the first postfire year (Van Wagner 1970). Fuel loadings in white pine stands include approximately 7.1 and 11.1 tons/acre (15.9 and 24.8 t/ha) of woody surface fuel and forest floor material, respectively, and 5.3 and 10.5 tons/acre (11.8 and 23.6 t/ha) of woody surface fuel and forest floor material, respectively, for red pine (Stocks and others 1990).

Jack Pine—Immature jack pine stands are stocked with 4,000 to 12,000 stems/acre (10,000 to 30,000 stems/ha) and heavy thinning mortality results in a large quantity of standing dead stems and down woody fuel, creating both horizontal and vertical continuity in the fuel. Fully stocked mature stands 400 to 800 stems/acre (1,000 to 2,000 stems/ha) have reached canopy closure and the base of the crown is separated from the ground. Surface woody fuel loadings vary from 2.9 to 17.2 tons/acre (6.5 to 38.6 t/ha) and forest floor loadings vary from 7.0 to 51.4 tons/acre (15.7 to 115 t/ha) (Quintilio and others 1977; Stocks 1989; Stocks and others 1990).

Postfire Plant Communities

Aspen

Pre-1900 Succession—Before settlement by Euro-Americans, large expanses of western aspen and aspen parkland existed in both the Canadian and American West. These stands were often perpetuated as a shrublike cover by light to moderate intensity fires that swept across the prairie grasslands and ignited the aspen stands on a regular 3 to 15 year basis. Aspen regenerated well after fire. Settlement of the West in the late 1800s and early 1900s increased fire frequency because of land clearing fires, slash burning, and railway traffic (Murphy 1985). In more recent times, following the implementation of rigorous fire protection programs, lack of fire has threatened the continued existence of aspen in the West (Brown and DeByle 1987, 1989; Peterson and Peterson 1992).

In the Rocky Mountains, low intensity fires caused thinning and encouraged all-aged stands whereas high intensity fires resulted in new even-aged stands. In early postfire communities aspen may be dominant but replacement of seral aspen by conifers is gradual and may take 200 to 400 years or more (Bartos and others 1983), depending on the potential for establishment and growth of conifers. Forage production declines two- to four-fold during succession.

Post 1900 Succession—Fire suppression since the 19th century has altered dynamics in aspen stands in the mountainous Western United States, changing fire frequencies from as little as 10 years (Meinecke 1929 in DeByle and others 1987) to approximately 12,000 years (DeByle and others 1987). Without the occurrence of disturbance, aspen clones mature in about 80 to 100 years (Schier 1975) and regeneration for this species is threatened. The dying back of those stands is now favoring shade-tolerant conifers or in some case grasses, forbs, or shrubs, depending on the availability of seed sources (Beetle 1974; Bergeron and Danserau 1993; DeByle 1976; DeByle and others 1987; Krebill 1972; Schier 1975).

Aspen stands may be replaced by conifers if subjected only to low intensity fires that do not kill conifer regeneration. Such conversion may require the absence of moderate to high intensity fire for as little as one aspen generation to as long as 200 to 1,000 years in the southern boreal forest or in the Rocky Mountains (Bergeron and Dubuc 1989; J. K. Brown 1985; Brown and Simmerman 1986; Perala 1990).

Management Considerations—In those areas where the perpetuation of pure aspen stands is desired, prescribed burning can be an economical and ecologically sensitive silvicultural tool since it closely mimics the natural disturbance and regeneration patterns (Brown and DeByle 1987, 1989). Wildland fire use in wilderness areas and prescribed fire are the most acceptable management tools for maintaining aspen stands in the American West (DeByle and others 1987) and can be useful in aspen stands in the East as well (Perala 1974; Weber 1990). The timing and resulting intensity of the fires is critical because high intensity burns can reduce site productivity (Weber 1990). But also, low intensity fires may not spread adequately and with enough heat to kill most aspen resulting in a poor response of aspen suckers (Brown and DeByle 1987). The prescription window for successfully burning aspen tends to be narrow due to inherently low flammability. However, flammability varies substantially among aspen stands. Appraising flammability to recognize good burning opportunities can increase chances of success (Brown and Simmerman 1986). Thus, prescribed burners must be ready to act quickly when opportunities arise. Fuel enhancement by cutting some trees especially conifers to increase surface fuel loading and continuity can be helpful in some situations. In the Western United States, grazing by domestic stock and sometimes wild ungulates must be controlled to assure successful regeneration of aspen (Bartos and others 1994).

White and Red Pine

Pre-1900 Succession—Full succession of plant communities was not possible because of the fire regime associated with white and red pine stands. Recurrent nonlethal fires eliminated shade-tolerant competing vegetation and prepared seedbeds for trees. Red pine regeneration was probably more closely associated with high intensity fires than white pine regeneration because of red pine's greater intolerance to shade and humus. However, shading is required for optimal regeneration of both white and red pine. The resulting forest is an even-aged stand for both white and red pine. The within stand age-class distribution is greater for white pine than red pine based on repeated observations that white pine can regenerate well under a nurse crop if the seedbed is suitable (Eyre 1980).

Post-1900 Succession—As white and red pine stands mature, their regeneration fails to establish under cover, and tolerant species slowly invade. Without fire, stands on dry sites evolve into mixed wood forests dominated by balsam fir and black spruce, or on mesic sites, white birch and aspen. White pine frequently pioneers on abandoned agricultural land, particularly in New England. However, in stands where fire is delayed for excessive periods, invasion by shade-tolerant species is a widespread phenomenon (Day and Carter 1990) leading to other forest types dominated by shade-tolerant species. After a high intensity 1995 wildfire in mature white and red pine stands in Quetico Provincial Park, red pine seedlings were present in low numbers but white pine seedlings were rare (Lynham and Curran 1998).

Management Considerations—Traditionally white and red pine stands were clear-cut (Mayall 1941) or clear-cut with residual seed trees at one to eight seed trees/acre (one to 20 seed trees/ha). However, competing vegetation, particularly beaked hazel and balsam fir on mesic sites, invade easily and prevent white and red pine from regenerating properly. This has led to a normal-shaped age-class distribution, contrasting strongly with a natural negative exponential distribution. Moreover, average age of white and red pine forests is now 72 years whereas pre-settlement average age was approximately 230 years. Shelterwood cutting is now the preferred silvicultural method for white pine management because it favors regeneration of white pine and to a lesser extent red pine, provided that mineral soil is bared.

Contrary to southern pine associations, fuel in white and red pine associations stabilizes after several years and there is no need for fuel control to prevent catastrophic fires. Furthermore, understory invasion by shade tolerant species has a negative impact on fire behavior by creating a layer of less flammable material (Van Wagner 1970).

Because white and red pine ecosystems are fire-prone and so dependent on fire for their regeneration, prescribed fire is an ideal silvicultural tool for regeneration and removal of competing vegetation (McRae and others 1998). Parameters of the Canadian Forest Fire Weather Index System (FWI) (Van Wagner 1987) that provide optimal results from prescribed burning in white and red pine stands have been determined to be (1) fine fuel moisture code 90 to 95, (2) initial spread index 8 to 16, (3) build up index up to 52, and (4) fire weather index 12 to 24 (Van Wagner and Methven 1977). Restoration forestry on rich sites may require two successive burns to eradicate resilient understory competition, particularly beaked hazel, and to prepare seed beds.

In the United States, low-intensity prescribed fire is the only practical treatment currently available to control white pine cone beetle that can ruin close to 100 percent of the cone crop in white pine seed orchards if not controlled. Wade and others (1990) developed guidelines for using prescribed fire in seed orchards throughout the Eastern United States.

Jack Pine

Pre-1990 Succession—Owing to its serotinous cones, jack pine regenerated well following fire and its shade-tolerant understory associates were eliminated. Jack pine is a shade-intolerant short-lived, pioneer species that almost always grew in even-aged stands (Eyre 1980; Farrar and others 1954; Heinselman 1981). Seeds germinate best on mineral soil or highly reduced organic soil. Postfire growth was found to be favored on thin organic soils in Ontario (Weber and others 1987). However, in northern Quebec, postfire organic horizon thickness did not affect seedling growth (St-Pierre and others 1991). Without fire, jack pine forests are succeeded by balsam fir stands on xeric to mesic morainic surface deposits in the southern part of the Quebec boreal forest (Bergeron and Dubuc 1989). Alternatively, jack pine forests may be replaced by white spruce or black spruce in western Canada (Rowe and Scotter 1973; Wein 1983) and by balsam fir, aspen, paper birch, and eastern white cedar in eastern Canada (Bergeron and Dubuc 1989).

Post-1900 Succession—Fire suppression has a dramatic effect on the long-term productivity of jack pine associations. In the Northwest Territories, Wein (1983) described the possible impact of the absence of fire in boreal jack pine stands as follows. As soil organic matter builds up, jack pine becomes less vigourous and black spruce tends to move into the jack pine habitat until finally the organic matter becomes too deep or nutrients become too limiting for black spruce regeneration and black spruce is replaced by willow. In eastern Canada, old-aged stands of the boreal forest suffer from high levels of dieback and are transitory to associations dominated by black spruce or balsam fir (Cogbill 1985). In Michigan, the loss of extensive areas of dense, fire regenerated jack pine stands that existed prior to 1900 is the primary reason for the decline of the Kirtland's warbler, a species whose breeding habitat requires large tracts of 8 to 20 year-old jack pine associations (Radke and others 1989).

Management Considerations—Clear-cutting is the preferred harvesting method for jack pine. However, competing vegetation (particularly on the richer sites), insufficient seed rain, and thick humus can prevent development of proper stocking (Burns and Honkala 1990; Chrosciewicz 1990). In Quebec, and probably throughout the entire jack pine range, only 4 percent of jack pine stands have sufficient pre-established growth before clear-cutting (Doucet 1988). On the other hand, jack pine can regenerate well in clearcuts if 40 to 50 percent of treated areas display bare mineral soils and if branches are scattered evenly over the terrain (Ball 1975; Chrosciewicz 1990). If planting is necessary, jack pine must be planted on site-prepared terrain (Beland and Bergeron 1993; Sutton and Weldon 1993).

Slash burning, together with seed tree systems, featuring approximately eight seed trees/acre (20 trees/ha) is the best method of regenerating jack pine after cutting (Chrosciewicz 1988, 1990). In addition, direct seeding has been shown to be successful on sites that have received prescribed burning (Cayford and McRae 1983). In practice, it was found that maximum regeneration can be achieved when burning occurs under the following conditions of the Canadian Forest Fire Danger Rating System: fire weather index of 4, duff moisture code of 37, fine fuel moisture code of 80, build up index of 58, and initial spread index of 1.2 (Chrosciewicz 1988). Regeneration of jack pine is best achieved by high intensity fires (>434 Btu/ft/s or >1500 kW/m) that consume most of the humus layer (Weber and others 1987).

Stand-Replacement Fire Regimes

Major Vegetation Types

Major vegetation types experiencing stand-replacing fire regimes include stands dominated by black spruce, white spruce, red spruce, and balsam fir along with conifer bogs and tundra (table 3-1). All these types are characterized by stand-replacement fire regimes having different fire cycles that vary according to climate and topography. Stand-replacement fires are the most common type of fires in northern forests, sometimes becoming as large as 2.8 million acres (1.4 million ha) (Murphy and Tymstra 1986).

Fire Regime Characteristics

Black Spruce—Black spruce associations are widespread throughout the northern biome from Alaska to Newfoundland and from the northern limits of the tree line south into the northeastern and North-Central United States. This species is found on a wide variety of sites ranging from moderately dry (fig. 3-5) to very wet. At the southern portion of its range, black spruce is found primarily on wet organic soils, but farther north its abundance on uplands increases. In the Lakes States and in New England, black spruce is mostly found in peat bog and swamps.

In southern areas of the black spruce range, it is commonly associated with a variety of other tree species including balsam fir, paper birch, tamarack, and aspen (Hare 1954; Rowe 1972). The stocking and associated vegetation on organic peatland soils depends largely on water sources and movement (Eyre

Figure 3-5—Black spruce stand (Canadian Fire Danger Rating Group photo point).

1980). On moist sites, well stocked to dense stands of black spruce grow above a well-developed mat of feather mosses such as Schreber's moss, mountain fern moss, and knight's plume moss) (Eyre 1980; Meades and Moores 1989). On very wet mineral or organic soils, the moss carpet is dominated by Sphagnum mosses. Typically, moist to very dry nutrient-poor sites are characterized by open stands of black spruce with the ground covered by species of Cladina lichens. The shrub layer in these forests consists mainly of ericaceous plants such as sheep laurel and bog Labrador tea. The organic layer may exceed a depth of 8 to 12 inches (20 to 30 cm) and comprises branches, other woody debris, and a variety of slowly decomposing plant material (less than 2 percent of organic matter decays per year) (Van Cleve and others 1979). In interior and south central Alaska, black spruce is extremely common on cold, poorly drained sites (Viereck and Dyrness 1980).

Fires in boreal black spruce ecosystems are large and frequent owing to the flammable nature of the forest, continuity of fuel sources and continental climate conditions (Heinselman 1981; Viereck 1983). Most fires in black spruce associations are either severe ground fires or crown fires of sufficient intensity to damage aboveground vegetation including the black spruce overstory and all of the associated understory shrubs. Some of the organic soil component usually remains, but high intensity summer fires, during dry, windy climatic conditions may achieve sufficient intensity to burn off this entire layer and completely expose the mineral soil. When drought conditions are extreme the entire forest structure becomes highly flammable and burns readily. In less extreme conditions, the lichen-dominated black spruce forest burns while the moister and older feather moss-dominated stands or deciduous mixed wood areas remain unburned (Foster 1983).

In general, the fire rotation of black spruce forests tended to be relatively short in the West while it was longer eastward, in response to a moister climate. Estimates of fire rotations range from 50 to 100 years in northwestern Canada and Alaska (Heinselman 1983; Rowe and others 1974; Viereck 1983) and for eastern Canada, 500 years in southeastern Labrador (Foster 1983; Viereck 1983), and 480 years in western Newfoundland (Wilton and Evans 1974). The longer rotation in eastern areas of the Canadian boreal forest are probably the result of higher levels of precipitation, and the occurrence of natural firebreaks such as lakes and extensive areas of moist peatlands.

White Spruce—White spruce associations have a transcontinental range from Newfoundland and Labrador, west across Canada along the northern tree line to Hudson Bay, Northwest Territories, Yukon, and Alaska. From British Columbia the range extends east through Alberta and Manitoba and south and east through northern Minnesota and Wisconsin, central Michigan, northeastern New York, and Maine. In the eastern forest, white spruce grows from sea level to elevations of about 5,000 feet (1,520 m) in pure stands or in mixed stands where it is the major component. Associated species include black spruce, paper birch, aspen, red spruce, and balsam fir. In eastern Canada and northern New England white spruce is a climax species. Closed pure white spruce stands and spruce-balsam fir occur sporadically throughout the boreal forest on moist and fresh soils, particularly on intermediate and low slopes (Burns and Honkala 1990; Eyre 1980).

Because mature white spruce forests accumulate large amounts of organic matter consisting of feather mosses, woody fuels, flaky barks, and shrubs, they are highly susceptible to fire. Also the crown and canopy structure with tree crowns extending nearly to the ground is ideal for ignition and propagation of crown fires (Rowe and Scotter 1973; Van Wagner 1983). In the boreal forest, the fire regime was characterized by crown fires or severe surface fires with a return interval averaging 50 to 150 years. Lightning caused most fires, especially in the months of July and August (Heinselman 1981). Fires could cover areas of 3 to 250 acres (1 to 100 ha), but the most ecologically significant fires were large, often thousands of acres. One of the longest fire cycles, 300 years, characterized the flood plain white spruce forests. In the East cycles may average 150 to 300 years. In British Columbia, Hawkes and others (1997a) reported fire cycles ranging from 794 to 2,083 years for the wet, cool sites of white/englemann spruce–subalpine fir forests in the northern Rocky Mountains. The dry, cool sites of white spruce/aspen forests (Kluane Section, B.26d, of the boreal region; Rowe 1972) in Kluane National Park, Yukon, had mean fire return intervals ranging from 113 to 238 years while individual stand intervals ranged from 9 to 403 years. Francis (1996) reported that white spruce dominated forests of the Shakwak Trench, Yukon, were characterized by large-scale, infrequent fire disturbances on north slopes while the south-slopes had small to medium-scale, frequent disturbances.

Balsam Fir—Balsam fir associations are found throughout eastern and central Canada as a band stretching from Newfoundland and central Labrador south to New York in the east and to central and northern Alberta in the west (Hosie 1973). Balsam fir is found in pure stands, and also in association with many species common to moist and wet sites where the ground-covering feather mosses build layers from less than an inch (a few centimeters) to more than 1 foot (30 cm) in thickness depending on stand maturity.

Balsam fir is a shade tolerant and late successional species. In the Eastern forests important associates were red spruce (mainly in New Brunswick and Maine) and paper birch. In the boreal forest region, it was more commonly found with black spruce, white spruce, jack pine, paper birch and trembling aspen. Balsam fir occurred on a wide range of soils including heavy clays, loams, and sandy glacial till. It was a climax association on upper slopes and tops of mountains (Burns and Honkala 1990; Eyre 1980).

In the high precipitation areas of eastern Canada and the Northeastern States, fire cycles were between 150 to 300 years (Heinselman 1981; Wein and Moore 1977, 1979). Stand-killing crown fires of high intensity or severe surface fires occur often after long droughts (Heinselman 1981; Rowe 1983; Van Wagner 1983). Unique spatial arrangements and dry conditions are responsible for several smaller, nonstand replacing fires of lower intensities, every 20 to 30 years in northwestern Quebec (Dansereau and Bergeron 1993). The periodic outbreak of the spruce budworm (Choristoneura fumiferana) causes heavy tree mortality (fig. 3-6), leading to a large fire potential peaking 5 to 8 years after tree death. However, the flammability of such forests decreases gradually after that period as balsam fir fuel starts to decompose and understory vegetation proliferates (Stocks 1987).

Red Spruce—Red spruce was one of the more important conifers in the Northeastern United States and adjacent Canadian provinces. Red spruce was most abundant in the Maritime Provinces, neighboring portions of Quebec, south-central Ontario and in parts of Eastern North America but especially in Maine. It grows best in a cool, moist climate, in either pure stands or as a major component of the growing stock. In the northern part of its range, red spruce grows at elevations from near sea level to about 4,600 feet (1,400 m). In mixed stands other common associates were red maple, eastern hemlock, eastern white pine, white spruce, eastern white cedar, and black spruce (on wet sites). Red spruce stands are found over a range of sites including moderately well drained to poorly drained flats and the thin-soiled upper slopes. On acidic tills it is considered climax. It is present on fresh and moist acidic outwash and on well drained slopes and varying acidic soils in abandoned fields and pastures where it is usually subclimax (Eyre 1980) being replaced by shade-tolerant hardwoods such as sugar maple and beech.

The ground cover in dense red spruce stands consisted mostly of bryophytes, lichens, tree litter, and patches of young conifer germinants that rarely survive over 2 or 3 years. As stands opened up and light conditions improved additional arboreal species,

Figure 3-6—Aerial photograph of a Balsam fir stand after infestation by spruce budworm (Canadian Fire Danger Rating Group photo point).

shrubs, and herbs developed. The three main associations—Hylocomium/Oxalis, Oxalis/Cornus, and Viburnum/Oxalis—indicate increasing site productivity and increasing hardwood competition. The Oxalis/Cornus association is considered the best for growing conditions (Eyre 1980). Presettlement fire rotations in red spruce associations averaged 100 to 150 years and higher (Heinselman 1981; Wein and Moore 1977, 1979).

Conifer Bogs—Conifer bogs are found over a wide range of soil and climatic conditions in Canada, and the North-Central United States and Alaska. Tree height, stand density, and especially floristic composition can differ considerably. Stand density varies from open to well-stocked stands. Composition can range from pure black spruce, tamarack, or eastern white cedar stands to black spruce in association with tamarack, balsam fir, eastern white cedar, paper birch, speckled alder, black ash, poplar, and willow (Armson 1982; Bergeron and Dubuc 1989; Eyre 1980; Johnston 1975, 1976).

Black spruce-dominated bogs, the most abundant bog type, are widely distributed throughout the boreal forest from Newfoundland to Alaska and south into the Great Lakes-St. Lawrence Region. Tamarack bogs, or wetlands stocked mainly with tamarack, are found from Quebec across the boreal forest to northwestern Alberta, south into the Lake States, and large portions of Minnesota, New York and New England. Pure eastern white cedar or eastern white cedar dominated bogs are limited to southern Ontario, Quebec and New Brunswick, the Lake States, and Northeastern States (Eyre 1980).

In general, conifer bogs are not as fire-prone as other forest stand types because of their wetter nature (Heinselman 1973, 1981; Payette and others 1989; Rowe and Scotter 1973). The high water table in the spring, a green dense understory, and a more humid environment render conifer bogs unsusceptible to forest fire except in severe drought years. These conditions, favoring low decomposition rates, enable the formation of an organic layer that can become greater than 3 feet (1 m) thick (Heinselman 1981). In turn, the thick organic layer of conifer bogs maintains a high moisture content that further contributes to the longer fire cycle of bog sites (Heinselman 1981; Rowe and Scotter 1973).

A second characteristic of bogs that tends to exclude fire from them is their tendency to occupy depressions and lowland areas. Because conifer bogs are mainly located in convex depressions, they are avoided by fire, which seeks convex landforms such as hilltops and slopes (Payette and others 1989; Rowe and Scotter 1973). Consequently, bog sites are spared from even large, high intensity forest fires, leaving unburned pockets of forests that become important seed sources to accompanying uplands (Bergeron and Dubuc 1989; Payette and others 1989).

Fire in conifer bogs occurs mostly in July, August, or September during severe drought years when a low water table allows the forest floor to become thoroughly desiccated. Under these circumstances with sufficient wind, spruce, tamarack, and eastern white cedar trees of conifer bogs can sustain major crown fires. Heinselman (1981) estimated the fire rotation to be 100 to 150 years for large forested spruce bogs in Minnesota. He found that the average fire rotation of spruce bogs is longer than those of nearby upland forests.

Tundra—Tundra ecosystems can be separated into three main types: (1) arctic tundra occurring at high altitudes and low elevations, (2) alpine tundra occurring at higher elevations farther south, and (3) the sedge tussock-mixed shrub in Alaska, throughout the Seward Peninsula and interior. Whereas vegetation is similar in the first two types of tundra, they differ mostly in climatological factors such as maximum summer temperatures, extremes of day length, and intensity of solar radiation (Barbour and others 1980). In the three tundra types, the upper to 6 to 24 inches (15 to 60 cm) of soil is subjected to a seasonal freeze and thaw cycle. This layer, called the active layer, lies above the permafrost layer (Barbour and others 1980). Vegetation is short, often only 6 inches (15 cm) tall, and is dominated by perennial forbs, grasses, sedges, dwarf shrubs, mosses, and lichens (Barbour and others 1980; Brown 1983). Herbaceous perennials make up 95 to 99 percent of the tundra flora and 60 percent of these are hemicryptophytes (perennials with perennating buds just at the soil surface).

In general, distribution of vegetation over the tundra biome is characterized by a patchy occurrence of dense vegetation, sparse vegetation, and bare ground offering an interrupted fuel bed (Payette and others 1989). However, in the sedge tussock-mixed shrub tundra, the fuel layer, made up mostly of sheathed cottonsedge, is dense and continuous leading to large, fast spreading conflagrations (Racine and others 1987).

The transition zone between the upper limit of the boreal forest and the tundra has been distinguished as forest tundra and is characterized by a fragmented cover of scattered forest stands of lichen-spruce and lichen-heath communities on well-drained sites. The forest tundra is further delineated into the forest subzone and shrub subzone (Payette and others 1989; Sirois and Payette 1991). In both the forest tundra and tundra, the ground vegetation is the primary carrier of fire. Lichens resemble dead tissue more than live tissue in their susceptibility to fire and often serve as the initial point of ignition. The low shrub component can provide a high percentage of fine, dry fuel with a low temperature of ignition (Auclair 1983).

Burning patterns of tundra ecosystems generally are characterized by moderate intensity surface fires that may kill all aboveground plant parts but seldom destroy underground parts (Bliss and Wein 1972; Van Wagner 1983; Viereck and Schandelmeier 1980). The fire cycle may be as short as 100 years, but it is usually much longer (Viereck and Schandelmeier 1980). According to Sirois and Payette (1991) and Payette and others (1989), the modern fire rotation period increases from 100 years in the upper boreal forest to 180 to 1,460 years in the forest subzone and to 9,320 years in the shrub subzone of the forest tundra of northern Quebec. Tundra west of Hudson Bay is more influenced by a continental climate and is more likely to burn and therefore have a shorter fire rotation (Payette and others 1989). In addition, the tussock-sedge tundra of Alaska may be a fire-dominated ecosystem, although the fire interval has yet to be determined (Racine and others 1987).

The long fire rotations in the tundra are probably related to the prevalence of cold, humid summers, saturated peat profiles, and the absence of continuous vegetation cover. These features serve to restrict fire spread over a large area (Payette and others 1989). In northern Quebec, small sized fires (<120 acres or <50 ha) occurred more frequently in the tundra than in the forest tundra; and forest tundra fires were smaller, on average, than boreal forest fires. Some large fires overlapped the limit between the biomes of the northern boreal forest and the forest tundra, but there was no overlapping between the forest tundra and the shrub tundra (Payette and others 1989).

Fuels

Black Spruce—The Canadian Forest Service's Fire Behaviour Prediction system recognizes two fuel types associated with spruce forests (Forestry Canada Fire Danger Group 1992). In spruce-lichen woodlands (fuel type C1), the lichens that dominate the ground cover are exposed to sunlight and, with no vascular tissue, dry out easily (Auclair 1983). These dry lichens form a mat 0.75 to 1.5 inches (2 to 4 cm) thick and combine with a variety of woody debris and shrubs to form a highly flammable ground fuel (Johnson 1992). Furthermore, spruce-lichen woodlands without a closed canopy permit the passage of wind, thus enhancing the rate of fuel drying; and once a burn is initiated, wind passage carries fire and augments radiant heat transfer (Foster 1983). Alexander and others (1991) reported that wind tilts the flame easily in the lichen layer but a high-intensity flame radiation surface fire does not develop. The combination of the lichen layer and scattered black spruce tree clumps produce high-intensity fires with spread rates from 2.0 to 168 ft/min (0.6 to 51 m/min) and fire intensities of nearly 9,540 btu/ft/s (33,000 kW/m).

The boreal spruce fuel type (C2), the second black spruce fuel type, consists of a continuous mat of feather moss, lichens, moderate amounts of woody debris and a well developed ericaceous shrub layer. This fuel type can become highly flammable when it is dry. However, when forming a wet peatland, it is not readily flammable and can act as a firebreak (Foster 1983). This fuel type typically occurs in old, closed canopy black spruce stands where the mineral soil may remain frozen for most of the year (Foster 1983) because it is covered by a thick (4 to 8 inches; 10 to 20 cm) moss layer lying over a substantial accumulation of humus (6 to 12 inches; 15 to 30 cm). These sites are less prone to drying than the spruce-lichen woodlands because the ground fuel receives less radiant energy and less wind (Foster 1983; Viereck 1983). The tendency of black spruce to hold dead branches along the entire length of their stems provides a fuel ladder for fire to spread from the flammable shrub layer up into the crown in both fuel types (Viereck 1983).

In Alaska, open canopy black spruce with feather moss, lichens, low shrubs, and little woody debris is a common forest type (Viereck and Dyrness 1980). In such associations, the shrub layer is not an important part of the fuel ladder since fire bridges from the moss-lichen layer into the spruce crowns. In fuel studies surface woody fuels varied from 0.67 to 20.0 tons/acre (1.50 to 44.9 t/ha) and forest floor varies from 5.4 to 23.8 tons/acre (12.2 to 53.4 t/ha) (Dyrness and Norum 1983; Kiil 1975; Stocks and others 1990).

White Spruce—Mature northern white spruce stands have well developed moss layers, mostly mountain fern moss, Schreber's moss, knights plume moss, dicranum moss, and sphagnum. In northern Ontario, surface woody fuels varied from 10.9 to 13.1 tons/acre (25.3 to 32.4 t/ha) and forest floor varied from 19.6 to 111.1 tons/acre (45.6 to 53.8 t/ha) (Stocks and others 1990). In the British Columbia Rocky Mountains, surface woody fuels varied from 17.1 to 69.1 tons/acre (38.3 to 155.5 t/ha) and forest floor varied from 23.3 to 61.5 tons/acre (52.3 to 137.7 t/ha) (Hawkes 1979; Hawkes and others 1997b).

Balsam Fir—Fire behavior in budworm-killed stands differs greatly between spring and summer. To illustrate, spring fires resulted in sustained crowning and spotting with extremely fast spread rates whereas the summer fires were unable to sustain themselves and did not spread. The major reason for this difference seems to be a proliferation of lush green understory vegetation in the summer due to crown openings (Stocks 1987; Stocks and Bradshaw 1981).

Spring fires in balsam fir stands within the boreal and Great Lakes-St. Lawrence forest regions fires were common. They behaved explosively with continuous crowning, high spread rates, and severe downwind spot fires that killed and regenerated entire stands. But they seldom burned much of the organic layer because it is still cold and wet (Heinselman 1981; Rowe and Scotter 1973). Summer fires were much more likely to consume the organic layer. Surface woody fuels varied from 6.53 to 17.4 tons/acre (14.5 to 38.7 t/ha) and forest floor loading varies from 6.53 to 32.7 tons/acre (18.7 to 52.8 t/ha) (Stocks and others 1990).

Red Spruce—Under drought and extreme fire weather conditions, fires of high intensity covering large areas or severe surface fires were possible. In old stands where red spruce was associated with balsam fir the periodic outbreak of the spruce budworm caused heavy tree mortality. This made these stands more susceptible to wildfires due to crown breakage and the proliferation of highly flammable fine fuels such as needles, dry twigs, and bark (insect-wildfire hypothesis) (Furyaev and others 1983). Fire potential is greatest 5 to 8 years after tree mortality. During this period, fires of great intensities tend to spread quickly due to evenly distributed fuel. To our knowledge, fuel loadings have not been reported in the literature for red spruce ecosystems. Presumably they are comparable to those of black spruce and white spruce ecosystems.

Conifer Bogs and Tundra—Fuel loadings in conifer bogs are highly variable because of the multiple species combinations found in this forest type. Fuel loadings for tundra are extremely variable, ranging from nearly nothing in tundra barrens to a maximum of 375 lb/acre (400 kg/ha) in Alaskan sheathed cottonsedge communities (Ranice and others 1987).

Postfire Plant Communities

Black Spruce

Pre-1900 Succession—The semiserotinous cones of black spruce, located near the main stem, act as a potential seed bank from which seeds are dispersed after fire (Johnson 1992). Once open the cones release seed for 2 to 3 years following fire. Black spruce is relatively slow-growing and is best suited to mineral soils. However, the seedling and sapling stages are relatively shade tolerant, allowing black spruce to establish well even on organic soil.

Revegetation following fire usually follows one of two basic paths. On moderate to moist relatively fertile soils, black spruce feather moss succession dominates (Black and Bliss 1978; Viereck 1983; Wein 1975). After high severity fires that expose mineral soil, bryophytes and herbs dominate while black spruce seedlings establish. On the less severely burned sites, herbs and shrubs develop from rhizomes that were not killed by fire. This initial re-establishment lasts for up to 4 years, and although black spruce becomes established at this time, the shrub layer may dominate the site for the first 25 postfire years. Eventually, the shade-tolerant black spruce, often growing at densities up to 12,000 stems/acre (28,800 stems/ha) (Viereck 1983) outcompete the shrub layer and forms the canopy.

The second regeneration path begins with the establishment of black spruce with mosses, lichens, and in some cases grasses. After 10 to 60 years, the ground cover on such sites is dominated by fruticose lichens along with low ericaceous shrubs (Black and Bliss 1978; Viereck 1983). During this phase, the black spruce component eventually forms an open canopy until approximately 100 years after fire, a typical black spruce lichen association is developed (Viereck 1983).

Post-1900 Succession—In much of the Canadian and Alaskan boreal forest, fire regimes and postfire communities remain similar to the presettlement period due to remoteness and lack of access. Most fire disturbed stands eventually regenerate back into black spruce ecosystems just as they would have before effective fire suppression. In areas undergoing commercial harvest, slash material and improved access (and resulting human activities) may combine to cause more frequent and more severe fires than in the period preceding fire suppression (Heinselman 1981). Fire protection has lengthened fire rotation from 120 to 200 years in spruce lichen forest and from 100 to 150 years in closed black spruce stands of Northwestern Canada and Alaska (Viereck 1973). In Ontario and Quebec, fire protection has increased fire rotations from 50 to 100 years to approximately 90 to 150 years (Heinselman 1981).

Management Considerations—Large-scale commercial harvest of black spruce and its associated species in Canada is one of the single most important industrial activities in the boreal forest. The resultant emphasis on protection of black spruce stands from insect and fire damage has generally increased the fire rotation in black spruce, particularly in protected forest reserves (Heinselman 1981; Viereck 1983). Harvest scheduling of high risk stands (high fir component; older or senescent stands) is a strategy that has also been used to reduce fire risks. In Alaska, black spruce stands occur mostly on permafrost with low stand productivity and are a low priority for fire protection.

Where fire exclusion is practiced, the flammable nature of black spruce ecosystems combined with their remoteness present major problems to fire managers (Viereck 1983). In spruce-lichen woodlands, drying occurs rapidly (Auclair 1983; Johnson 1992) causing high fire hazard throughout the spring, summer, and fall. In spruce-feather moss ecosystems, particularly in the East, higher moisture levels reduce fire hazards. However, periodic droughts allow large tracts of this forest type to burn (Heinselman 1981).

Prescribed fire in black spruce cannot be used in areas with shallow soil profiles as soil conservation is a major concern in such sites. In low-lying Sphagnum-dominated sites, fire is usually not required for seedbed preparation as black spruce seeds easily germinate on the Sphagnum bed itself (Archibald and Baker 1989). In lowland spruce-feather moss stands, moderate to high severity fires are required to remove the feather moss layer, slash, and litter to prepare the seedbed. When heavy competition from alder or other boreal hardwoods is present, high severity burns are required to kill off the subterranean rhizomes of the unwanted species, although if herbicides are applied before the burn, low to moderate severity fires may suffice (Archibald and Baker 1989).

The weather indices used for prescribed burns vary with silvicultural goals. In northwestern Ontario, low severity fires designed to burn fine fuels without removing large amounts of duff require a duff moisture code of 4 to 10 and a buildup index of 15 to 19. Moderate severity burns for the purpose of removing fine fuels, some heavy fuels and patches of duff require a duff moisture code of 8 to 16 and build up index of 18 to 28. Severe fires removing both fine and heavy fuels as well as 0.5 to 1.5 inch (1 to 4 cm) of duff require a duff moisture code of 13 to 25 and a build up index of 24 to 45 (Archibald and Baker 1989). Quantitative slash and forest floor consumption can be predicted using tables developed for lowland and upland black spruce forest by McRae (1980).

White Spruce

Pre-1900 Succession—Depending on seed rain, white spruce stocking may either decrease or increase in postfire communities. Seed production varies greatly from year to year with good seed crops every third year or so. Cones mature in late August or September and the majority of the seeds fall soon thereafter. So postfire regeneration of white spruce is increased when fire occurs in late summer of a good seed year. Otherwise, the only available seeds come from unburned patches or the edge of the burn (Dix and Swan 1971).

Feather mosses and lichens are eliminated wherever the moss layer has burned but reappear 10 to 15 years after fire and often reach high ground coverage within 50 years. If only white spruce is present in the prefire community, and seed trees are eliminated over large areas, an initial establishment of aspen seedlings may occur, thus slowing the reestablishment of white spruce (Van Cleve and Viereck 1981).

Post-1900 Succession—Past fires, logging, and land-clearing in the Northeastern forests caused white spruce associations to be replaced with aspen. The successional trend for such stands is generally toward spruce-fir and, where soil and climatic conditions permit, toward northern hardwoods. Fire suppression and control in Alberta allow stands to mature and succession favors shade-tolerant species such as balsam fir and tolerant hardwoods (Weetman 1983).

Management Considerations—Clearcutting is the usual harvesting method for white spruce ecosystems. Clearcutting followed by broadcast slash burning results in the most marked changes in vegetation and soil temperature. Experiments in Alaska showed that many of the species present before disturbance disappeared afterwards (Dyrness and others 1988). Soil temperature increased on clearcut and shelterwood sites, but not as much as on clearcut and burned areas. Thinning had the least impact on vegetation and soil temperature (Dyrness and others 1988). Prescribed fire is effective in reducing heavy slash, seedbed preparation, controlling unwanted vegetation, and promoting vigor in desired species. Because white spruce is particularly sensitive to environmental conditions during germination and early growth, it is among the most difficult conifer species to regenerate naturally. However, summer logging or scarification that exposes mineral soil seems to improve regeneration (Brand and Janas 1988; Eis 1965, 1967; Endean and Johnstone 1974).

Natural white spruce stands can respond well to cultural practices, especially releasing. White spruce stands should be maintained at basal areas from 100 to 140 ft^2/acre (23.0 to 32.1 m^2/ha) to provide maximum volume growth and good individual tree development. Below these levels, individual tree increment and resistance to some pests are greatly increased, whereas total volume production is reduced (Burns and Honkala 1990).

Balsam Fir

Pre-1900 Succession—Owing to the high sensitivity of this species to fire, balsam fir survives only extremely low intensity fires or in patches of unburned forest. Even if trees are only damaged, fungal diseases and insect attacks will quickly destroy the stand structure. Following canopy opening by a first fire, a second fire can cause even greater changes in species composition (Little 1974).

Long fire cycles allow establishment of old-growth stands dominated by shade-tolerant balsam fir regeneration, a condition that is most abundant about 100 years after fire (Heinselman 1973). Because balsam fir has little fire tolerance, fire in balsam fir dominated forests tends to eliminate most of the existing stems and favors conversion to other tree species. Accordingly, fire in black spruce-balsam fir stands tends to favor the black spruce component (Candy 1951; Damman 1964; MacLean 1960). In the absence of conifer seed trees, postfire communities are dominated first by aspen or paper birch seedlings that originate from wind-borne seeds. Subsequently, white or red spruce seedlings invade the burns, developing as an understory to the aspen-birch complex and eventually replace it. On some of the better sites, northern hardwoods such as sugar maple and beech eventually replace white spruce. Alternatively, balsam fir dominates (Day 1972).

Post-1900 Succession—Fire protection, coupled with clear-cut logging have favored the establishment of pure balsam fir associations. The shade-tolerant seedlings of balsam fir establish well in the shade of balsam fir or spruce stands. Thus, following logging, pre-established balsam fir generally dominates in the regenerating stand. However, in some situations, bracken fern, raspberry, and hardwood suckers may represent intense competition to the balsam fir regeneration for the first 10 to 25 years after disturbance (Burns and Honkala 1990). Repeated burns or cutting and burning in Newfoundland can lead to the development of Kalmia barrens or heathland conditions (Damman 1964), which once established are difficult to reforest. Black spruce invades such sites due to poor seed supplies of balsam fir and soil deterioration (Damman 1964).

Management Considerations—Clearcut logging has been the most used harvesting method for this forest type. However, site characteristics, the time of season, logging practices and size of the cut area are important factors to prevent damage for the pre-established regeneration of most spruce and fir stands, especially in the boreal region. A study in northwestern Quebec has shown that there is a deterioration in stand quality and a severe softwood mortality (92 percent) after harvesting and a shift in species composition from softwood-dominated advance regeneration to mixed or hardwood and shrub-dominated secondary forests. Often, it will be necessary to treat the area if the softwood crop is desired. Silvicultural treatments include scarification, direct seeding, planting, brush control, and subsequent thinning.

Overmature balsam fir forests (about 100 years of age) are susceptible to spruce budworm infestations (Heinselman 1973). Silvicultural options include harvest or protection against spruce budworm outbreaks, which involves the use of chemical or biological insecticides (Dimond and others 1984). Budworm outbreaks have serious economical and social impacts on forest industry. Accordingly, there is interest in converting fir forests in eastern Canada that are susceptible to spruce budworm to less susceptible species, particularly black spruce. Species conversion requires intensive site preparation owing to the presence of understory fir regeneration that provides strong competition to planted spruce seedlings on many sites. A cost-effective method of site preparation is prescribed burning, which eliminates fir regeneration completely and favorably prepares the site for spruce seedlings. Prescribed burning in balsam fir forests could also be used for removal of slash prior to planting, or for reduction of fire hazard (Furyaev and others 1983). Prescribed burning in budworm-killed forests has been applied in Ontario (Stocks 1987) and in Quebec (Robitaille 1994).

Red Spruce

Pre-1900 Succession—Red spruce was a late successional species forming old-growth stands in areas with long fire cycles (Burns and Honkala 1990). Its shallow root system, thin bark, and flammable needles make trees of all ages susceptible to fire damage. Natural reproduction of red spruce depends on seedling survival since seedlings have an exceptionally slow-growing, fibrous, and shallow root system. A critical factor for their survival and establishment is the depth of the organic layer. Although a shade-tolerant species, the relative tolerance of red spruce to shade varies with soil fertility and climate. At first, seed germination and initial seedling establishment proceed best under cover. However, older seedlings require 50 percent or more sunlight for optimum growth (Burns and Honkala 1990).

Red spruce's chief competition comes from balsam fir and hardwoods such as beech and maple. Red spruce seems to respond to release even after many years of suppression (up to 100 years). Nevertheless, many of its associated tree species such as balsam fir and hemlock may outgrow red spruce after release (Burns and Honkala 1990).

Post-1900 Succession—Clearcutting had a serious impact on red spruce ecosystems. Many former spruce sites are now occupied by inferior tree species, blackberries, and ferns. Dense growth of bracken, raspberry, and hardwood sprouts are the chief competition for seedlings on cutover lands (Burns and Honkala 1990). Logging and fire in the Maritimes has left large areas of red spruce stands in poor conditions in terms nutrient status and species composition, leading to poor growth and conversion to heathlands. In turn, this has caused shortages of saw timber and pulpwood (Weetman 1983).

Management Considerations—Red spruce may be grown successfully using even-aged silvicultural prescriptions. Because red spruce is shallow-rooted, it is highly sensitive to windthrow. To minimize the danger of wind damage, it is recommended that no more than one-fourth to one-half of the basal area, depending on site, be removed in the partial harvest of a spruce-fir stand (Frank and Blum 1978).

Conifer Bogs

Pre-1900 Succession—There are two different views on the effect of fire on cold northern bogs. Rowe and Scotter (1973) describe how fire can contribute to site paludification. Fire removes the transpiring crowns, the water table rises, light levels increase on the ground floor, which promotes the establishment of Sphagnum mosses. Then moss layers build up, insulating the soil and lowering soil temperature. All of these fire effects create poor seedbed and poor rooting conditions for trees, thus excluding tree establishment and leading to a tundra type community. Conversely, Heinselman (1981) found that prolonged fire-free intervals cause a rise in permafrost tables, impede soil drainage and general site degradation. The result is a depauperate black spruce muskeg (Heinselman 1981). Strang (1973) found the same successional pattern in northern open black spruce-lichen associations.

Spruce-Sphagnum bogs that have been destroyed by fire are replaced by birch and other pioneer species, then evolve to a stage where both birch and spruce are present and, finally, to the spruce stage at which paludification begins (Larsen 1980). However, recurrence of fire in conifer bogs prevents most stands from reaching their climax stage (Bergeron and Dubuc 1989; Heinselman 1973, 1981).

Lightning was, and still is, a major cause of fire in the boreal forest. Lightning strikes during rainless storms during prolonged summer droughts ignite major fires to which conifer bogs are most susceptible. Such fires expose the mineral soil and create seedbed conditions conducive to the regeneration of conifer bog species (Heinselman 1981).

Spring and fall fires become more common in the boreal forest with increased human activity. Such fires have relatively little effect on reducing the duff layer and subsequently produce poor seedbed conditions for conifer bog species (Heinselman 1981; Viereck and Johnston 1990). Lightning fires are probably still responsible for most of the annual area burned.

Before 1900, fire prevented high concentrations of dwarf-mistletoe on black spruce because this parasite is temporarily eliminated when fire removes its host. Therefore fire was able to eradicate centers of infestation, thus preventing this disease from spreading. Current fire exclusion policies favor vast expansions of mistletoe (Heinselman 1973).

Post-1900 Successions—Human influence on conifer bog sites has been somewhat limited because most bogs have poor timber productivity. Most anthropogenic influence arises from the harvesting of surrounding stands and the side effects of fire protection on those same higher value sites. The policy of fire protection has lengthened the fire cycle for the boreal forest and, therefore, the fire cycle of nearby bog sites (Heinselman 1981).

Management Considerations—Management practices in conifer bogs depend greatly on site productivity. Poorly drained, wet sites have such low wood productivity that little silvicultural management is done in these conifer bogs (Viereck and Johnston 1990). For the more productive peatland black spruce sites, clear-cutting in strips or patches, combined with broadcast burning of slash, followed by direct or natural seeding, is the best silvicultural treatment (Johnston 1975; Viereck and Johnston 1990). This same system works well for northern white cedar, but shelterwood management is preferred as it supplies partial shade for establishing seedlings (Johnston 1990b).

The shade-intolerance of tamarack dictates the use of even-aged management, with some adaptation of clearcutting or seed tree cutting generally considered the best silvicultural system (Johnston 1990a). Uneven or all-aged management is best applied to poor black spruce sites where stands are wind firm and have abundant layering (Viereck and Johnston 1990). Prescribed burning improves seedbed conditions and kills back competing brush (Armson 1982; Johnston 1973, 1975, 1976). However, tamarack slash is difficult to burn (Johnston 1973, 1975). Seedling survival is substantially greater on prescribed burn sites.

Tundra

Pre-1900 Succession—Fire effects in tundra ecosystems vary greatly according to the composition of prefire plant communities (Bliss and Wein 1971). Generally, fire favors rapidly growing species, particularly graminoids, and there is a decreased abundance of slow growing species such as evergreen shrubs immediately following fire (Bliss and Wein 1971, 1972). The recovery of mosses and lichens is slow as opposed to that of sedges and grasses, and recovery of shrubs is intermediate (Bliss and Wein 1972). Postfire regeneration is reliable because all shrubs and deciduous tree species common to the tundra are capable of reproducing vegetatively after fire (Auclair 1983; Viereck and Schandelmeier 1980). Establishment of pioneer species is mainly by wind-borne seeds from adjacent plant communites (Auclair 1983). Most lichens establish by small thallus fragments within the first several years following a burn, but their slow growth limits their abundance for the first 25 to 30 years (Auclair 1983).

The forest tundra zone is transitional, depending on fire cycle and fire severity (Viereck and Schandelmeier 1980). Sirois and Payette (1991) suggested that recurring fire progressively decreases the regenerative potential of trees and that sustained depletion of tree populations following several fires is a key process in the development of the forest tundra. In the shrub subzone of the forest tundra where tree regeneration is tenuous, a single fire event can eradicate the presence of trees and creates treeless tundra. The boreal forest-forest tundra interface may be prone to sudden fragmentation caused by fire and climate interactions (Sirois and Payette 1991). Severe fires causing depletions of trees resulting in shifts from forest to forest-tundra communities have been reported from subalpine sites in Western North America and subarctic sites in northern Sweden and northern North America (Sirois and Payette 1991).

Post-1900 Succession—There has been relatively little human activity in the vast expanse of the arctic tundra (Heinselman 1981). Most activity has been since the late 1960s with oil and gas exploration and the construction of pipelines. Disturbance to tundra communities has been mostly in the form of cut-lines (or seismic lines), winter roads, air strips, and fire (Bliss and Wein 1971). Continuing investigation into the effect of these anthropogenic disturbances has resulted in the use of technologies that have a less deleterious impact on plant communities of the tundra.

Management Considerations—Efforts to contain or stop the spread of fire in the tundra produce more drastic long-term effects than the fire itself (Brown 1971; Viereck and Schandelmeier 1980). Construction of firelines with bulldozers strips away all insulating moss and peat layers and exposes bare mineral soil. This allows the summer heat to penetrate directly into the frozen ground, which in turn increases the depth of the active layer under the fireguards compared to under burned areas. This causes a more rapid and greater degree of subsidence under the firelines than under the burned areas due to the melting of ground ice (Brown 1971), erosion and gully formation (Brown 1983).

Notes

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Chapter 4: Fire in Eastern Ecosystems

Prior to Euro-American settlement, fire was a ubiquitous force across most of the Eastern United States. Fire regimes spanned a time-scale from chronic to centuries. Fire severity varied from benign to extreme (fig. 1-2). Today, fire is still a major force on the landscape. In some ecosystems fire stabilizes succession at a particular sere, while in others, succession is set back to pioneer species. The wide range in fire regimes coupled with elevation and moisture gradients produce a myriad of plant communities that continually change over time in both stature and composition, although it is not uncommon for the major species to remain dominant. Discussion is primarily about major vegetation types, for example, oak-hickory. However, some minor types such as spruce-fir and Table Mountain pine are also covered. Vegetation types are discussed under the most representative fire regime type, recognizing that some vegetation types overlap two fire regime types (table 4-1).

Understory Fire Regimes

Major Vegetation Types

The presettlement understory fire regime (overstory survival typically exceeds 80 percent) primarily applies to southern pine and oak-hickory forests comprised of pine and pine-oak associations such as Kuchler's southern mixed forest, oak-hickory-pine, Northeast oak-pine types, and oak-hickory associations. Kuchler potential natural vegetation classes are listed by fire regime types in table 4-1 and cross-referenced to FRES ecosystems and Society of American Foresters cover types.

The extent of the various forest types at the time of Euro-American colonization is difficult to reconstruct. Considering old survey corner trees, Plummer (1975) concluded that pine and post oak predominated in the Georgia Piedmont. Based on soils, Nelson (1957) concluded that 40 percent of the Piedmont was in hardwoods, 45 percent was hardwood with varying degrees of pine, and 15 percent was predominantly pine. Nomenclature used by early naturalists was often ambiguous including different names for the same species, or use of a species name to group several species together such as the use of the term short-leaf pine to differentiate all short-leafed pines from longleaf pine. Furthermore, earlier works estimated that 80 percent of this region had been cleared at some point in time (Nelson 1957). Hamel and Buckner (1998) described the "original southern forest" at three periods and concluded that no specific period represents the "true" original condition of the southern forest because it has been shaped by humans for millennia.

Table 4-1—Occurrence and frequency of presettlement fire regime types by Forest and Range Environmental Study (FRES) ecosystems, Kuchler potential natural vegetation classes (1975 map codes), and Society of American Foresters (SAF) cover types. Occurrence is an approximation of the proportion of a vegetation class represented by a fire regime type. Frequency is shown as fire interval classes defined by Hardy and others (1998) followed by a range in fire intervals where data are sufficient. The range is based on study data with extreme values disregarded. The vegetation classifications are aligned to show equivalents; however, some corresponding Kuchler and SAF types may not be shown.

			Fire regime types						
			Understory		Mixed		Stand-replacement		
FRES	Kuchler	SAF	Occur[a]	Freq[b]	Occur	Freq	Occur	Freq	Nonfire
Longleaf-slash pine 12	Southern mixed forest K112	Longleaf pine 70	M	1a:1-4					
		Longleaf-slash pine 83[c]	M	1a:1-4					
		Slash pine 84	M	1a:3-8	M	1a:5-10			
Loblolly-shortleaf pine 13	NE oak-pine K110	Virginia pine 79			M	1b			
		Pitch pine 45	M	1a	M	1b			
	Oak-hickory-pine K111	Shortleaf pine 75	M	1a:2-15					
		Loblolly-shortleaf pine 80	M	1a:3-10					
		Loblolly pine 81	M	1a:3-8	m	1a:5-10			
	Pocosin K114	Pond pine 98	m	1a:3-8	M	1:6-25			
	Sand pine-scrub K115	Sand pine 69					M	1b:25-45	
Oak-pine 14	Oak-hickory-pine K111	Shortleaf pine-oak 76	M	1a					
		Virginia pine-oak 78			M	1b			
		Table mountain pine					M	1,2	
	Southern mixed forest K112	Loblolly pine-hardwood 82[c]	M	1					
		Slash pine-hardwood 85	m	1					
		Longleaf pine-scrub oak 71	M	1a:6-10					
Spruce-fir 11	S.E. spruce-fir K097	Red spruce-Frazer fir 34					M	2,3	M
Oak-hickory 15	Oak-hickory K100	Northern pinoak 14	M	1					
	Mosaic of above and bluestem prairie K082	Bur oak 44	M	1a					
	Appalachian oak K104	Northern red oak 55	M	1b					
		Bear oak 43	M	1					
		Chestnut oak 44	M	1a:3-8					
		Black oak 110	M	1					
		White oak-black oak-n.red oak 52	M	1					
		Southern scrub oak 72	M	1	M	1			
	Blackbelt K089	Yellow popular 57	M	1	M	1			
	Cross timbers K084	Post oak-blackjack oak 40	M	1a					
	Oak-savanna K081		M	1a:2-14					
Elm-ash-cottonwood 17	Northern floodplain forest K098	Black ash 39			M	1,2	M	1,2	
	Elm-ash forest K101	Silver maple-American elm 62			M	1,2	M	1,2	
		Cottonwood 63			M	1,2	M	1,2	
		Sugarberry-Am. elm-green ash 93			M	1,2	M	1,2	
		Sycamore-sweetgum-Am. elm 94			M	1,2	M	1,2	

(con.)

Table 4-1—Con.

			Fire regime types						
			Understory		Mixed		Stand-replacement		
FRES	Kuchler	SAF	Occur[a]	Freq[b]	Occur	Freq	Occur	Freq	Nonfire
Oak-Gum-Cypress 16	Southern floodplain forest K113	Alantic white-cedar 97					M	2,3	
Maple-beech-birch 18	Maple-basswood K099	Sugar maple-basswood 26			M	3:>1000			m
	Beech-maple K102	Beech-sugar maple 60			M	3:>1000			m
		Black cherry-sugar maple 28			M	3:>1000			m
	N. hardwoods K106	Sugar maple-beech-yellow birch 25			M	3:>1000			m
		Sugar maple 27			M	3:>1000			m
	N. hardwoods-fir K107	Hemlock-yellow birch 24					M	3	
	N. hardwoods-spruce K108						M	3:300-500	
Mixed mesophytic forest[d]	Mixed mesophytic forest K103				M	2,3			
Prairie 39	Bluestem prairie K074						M	1a	
	Nebraska Sand Hills K075						M	1a	
	Blackland prairie K076						M	1a	
	Bluestem-sacahuista prairie K077						M	1a	
	Cedar glades K083						M	1a:3-7	
	Fayette prairie K088						M	1a	
Wet grasslands 41	Tule marshes K049						M	1	
	N. cordgrass prairie K073						M	1a:1-3	
	S. cordgrass prairie K078						M	1a:1-3	

[a]M: major, occupies >25% of vegetation class; m: minor, occupies <25% of vegetation class.
[b]Classes in years are 1: <35, 1a: <10, 1b: 10 to <35, 2: 35 to 200, 2a: 35 to <100, 2b: 100 to 200, 3: >200.
[c]This type developed after European settlement.
[d]This type is a subdivision of FRES ecosystem types.

The Southern mixed hardwood forest occupies the Atlantic and Gulf Coastal Plains and is often classified as deciduous broadleaved forest even though it includes pine-hardwood and broadleaved evergreen types. Southern pines are the predominant overstory species. The potential hardwood climax, however, was probably only reached on sites with a geomorphologic position or hydrologic condition that protects such sites from the chronic fire regime that characterizes this region. Southern mixed hardwood forests were confined to narrow strips between floodplain forests and uphill longleaf pine forests before Euro-American colonization (Batista and Platt 1997). Some authors placed the former longleaf pine forests in this forest type, but this seems illogical where time since disturbance, primarily by hurricanes and fire, and the elevation gradient (measured in inches) determine plant community composition. Embedded within this overall forest type are numerous other ecosystems such as nonalluvial depressional wetlands, including Carolina bays, limesinks, cypress ponds and savannas, gum ponds, bay swamps, pitcherplant bogs, shrub bogs, and spring seeps. Water stands in these depressions during at least part of the growing season, but they also dry allowing fire to enter from adjacent upland plant communities (Kirkman and others 1998; Wharton 1977). These fires regulate numerous processes needed to maintain these communities (Christensen 1977; Wade and others 1980; Wright and Heinselman 1973). Periodic fire also knocks back the wall of hardwoods that continually invades adjacent upland sites (Barrett and Downs 1943; Chaiken 1949; Chapman 1947; Harcombe and others 1993; Heyward 1939; Long 1888; Oosting 1942; Platt and Schwartz 1990; Wahlenberg 1949; Wells 1928). As the time span between fires increases, the stature of the ever-present hardwoods also increases, shading out the herbaceous groundcover, thereby breaking up its continuity and ability to support spread of fire; this allows the hardwoods to further expand until they eventually dominate the site (fig. 4-1). Tansley (1935) recognized this situation and classified Southeastern pine forest as a disturbance-dependent pyrogenic stable type. Wilhelm (1991) extended this thinking to the Central Hardwoods Region and questions the whole concept of succession and climax vegetation in the Midwest.

Over three dozen oaks and almost two dozen hickories form a myriad of potential overstory plant communities that comprise the oak-hickory forest (fig. 4-2). We define it as all stands in which oaks and hickories, singly or in combination, compose at least 50 percent of the dominant trees (J. A. Barrett 1994; Braun 1950). This forest type includes the oak-hickory-pine forest of the mid-Atlantic States, the Appalachian oak forest, the Northeast oak-hickory-pine forest, and stands within the western mesophytic and mixed mesophytic forest regions where oaks and hickories comprise a majority of the dominant trees. Such stands occur primarily on average to dry upland sites, although they can be found on moist upland sites as well, depending upon past disturbance history. Where other hardwood species dominate within the western and mixed mesophytic forest regions, they are discussed as mixed fire regime types.

Fire Regime Characteristics

Southern Pine Forests—The regional climate is characterized by long, hot growing seasons, abundant rain punctuated by occasional multiyear droughts, and the most frequent wind and lightning (Komarek 1964) storms in North America. Natural disturbance events such as microbursts, tornadoes, and hurricanes can significantly impact forested lands at several scales and often set the stage for more intense fires (Myers and Van Lear 1997). Lightning becomes increasingly common as one moves from north to south. Although the number of lightning fires peaks in June and July, the vast majority of acreage now burns in May and June in Florida (Robbins and Myers 1992) and southern Georgia, before the summer thunderstorm pattern becomes fully entrenched; this probably was the historic pattern as well.

The few remaining old-growth southern pine forest relics are too small to experience natural fire regimes, and neither dendrochronology nor palynology can be used to determine fire history in this region (Landers 1991). Thus, historical southern pine fire regimes must be inferred from interpretation of old records, field observations, experimental studies, and species traits. Frost's (1998) generalized map of pre-Euro-American settlement fire frequency for the Southern United States summarizes current thinking and shows that all forest types mentioned above had a fire-return interval of less than 13 years except pond pine pocosins in North Carolina.

Before Euro-American settlement, some fires were undoubtedly far ranging because they were associated with dry frontal passages. Growing season fires during severe drought years can burn for weeks or months, particularly in organic soils that underlie the deeper depressional wetlands, before being extinguished by rain and a rising water table (Cypert 1961, 1973). Such fires make frequent forays into adjacent upland communities. Most fires were, however, probably limited in extent, at least once the summer convective weather pattern set in. Nighttime humidities near 100 percent are the norm during the summer; and such humid, calm conditions tend to extinguish fires in light fuels. The pattern of occasional high-intensity, wind-driven fires and severe drought fires was superimposed on the chronic lightning and Native American fire regime

Figure 4-1—Fifty years of fire exclusion in an old-growth longleaf pine stand, Flomaton, Alabama. Photo by John Kush, 1994.

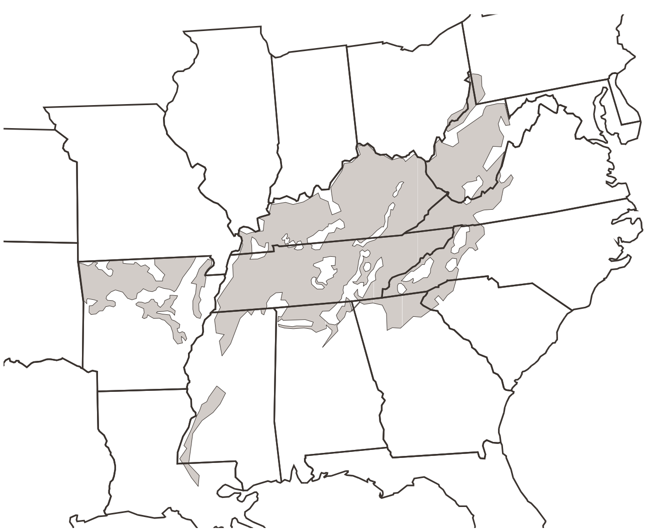

Figure 4-2—The combined distribution of the Oak-Hickory Forest Region, Mixed Mesophytic Forest Region, and Appalachian Oak Forest Region (Kuchler Types 100, 103, and 104) as depicted by Martin and others (1993).

creating the open woodlands referred to by early Euro-American explorers (Barden 1997; Landers and others 1990; Olson 1996) (fig. 4-3). The journals of many of these explorers also mention numerous smoke columns and extensive smoke and haze often lasting for days.

Native Americans used fire extensively to shape the vegetative mosaic for thousands of years, and for the past 400 or more years Euro-Americans have subjected these same lands to varying degrees of fire use and abuse, including the exclusion of fire. Whether human ignitions substantially increased the total acres treated by fire is uncertain, but based on the sheer number of artifacts left behind, accounts by early white explorers, and studies of depopulation ratios following disease (Dobyns 1966; Jacobs 1974), Native American populations must have been much larger than is generally recognized. According to Kennedy (1994), the population of Cahokia in central Illinois was greater than that of London in the 13th century. Native American-induced changes include the fact that they ignited fires throughout the year, but before their influence, many sites burned more frequently by fires confined to the spring and summer thunderstorm season (Martin and Sapsis 1992). A slight shift toward reduced fire intensity and severity took place. Growing-season fires tend to be patchier than dormant-season fires because of different weather patterns and

Figure 4-3—Pine flatwoods near Tallahassee, Florida, around 1900. Note open understory. Florida Department of Environmental Protection archives.

greater variation in fine-fuel moisture once greenup occurs.

Komarek (1982) pointed out that it is difficult to understand the practice of prescribed fire in the South without an appreciation of Southern history. It is also a prerequisite to realizing the tremendous impact humans have had in shaping the present vegetative mosaic. For example, prior to the Civil War, over 75 percent of the white population were pastoral herdsmen (Owsley 1945) who came from the British Isles, Spain, and France where fire was an integral part of their livelihood. They brought this practice with them, blended their knowledge with that of the Native Americans they displaced, and aggressively expanded the use and frequency of fire throughout the South. A circa 1731 North Carolina law required the annual burning of all pastures and rangelands every March (Carrier and Hardison 1976). Eldredge (1911) described the turn-of-the-century fire situation in north Florida as follows:

> …the turpentine operator burns his woods and all other neighboring woods during the winter months, generally in December, January, or February. The cattleman sets fire during March, April, and May to such areas as the turpentine operator has left unburned. During the summer there are almost daily severe thunderstorms, and many forest fires are started by lightning. In the dry fall months hunters set fire to such "rough" places as may harbor game. It is only by chance that any area of unenclosed land escapes burning at least once in two years.

There is little doubt that this pattern was typical throughout the region, although not everyone was in agreement with this ubiquitous use of fire. The debate regarding the benefits of intentional burning versus fire exclusion has been ongoing for more than 100 years as attested to by the following quote from the 1875 Atlanta Constitution (Blocker 1875).

> I have seen an article in the Early County News of June 12th, copied from the Columbus Sun condemning the burning of woods, but the writer is very badly mistaken in his views on the subject. I think there has been a great deal more injury from not burning the woods at a proper time, and when in proper condition, than has ever been done by burning; for fire will accidentally get in some time, and often when it is very dry, and do a great deal of injury.

Although some early Federal foresters thought fire was universally bad (Schwarz 1907), others disagreed. Chiefs of the United States Forest Service Gifford Pinchot (1899) and Henry Graves (1910), Supervisor of the Florida National Forests I.F. Eldredge (1911), as well as university researchers Gifford (1908), Bryant (1909), and Chapman (1912) recognized the invariable result of attempted fire exclusion and advocated the use of prescribed burning in Southern pines as a hazard reduction measure. However, with passage of the Clarke-McNary Act in 1924, which offered the States Federal funding for fire suppression, thinking within the Forest Service reversed itself, the use of fire was condemned (Demmon 1929), and the Dixie Crusaders crisscrossed the Deep South preaching the evils of woods burning. An excellent account of the methods used to force compliance with this new fire exclusion policy is given by Schiff (1962).

With attempted fire exclusion, dead fuels accumulated on the forest floor and a needle-draped understory of highly flammable shrubs such as saw palmetto, gallberry, and wax myrtle developed within 5 or 6 years on all but xeric sites. Given this receptive fuel bed and human and natural ignition sources, wildfire occurrence remained high. At first, these fires were easy to extinguish, but as fuels accumulated on unburned areas, fires became increasingly difficult to suppress, and the probability of catastrophic, high-intensity fire increased. Often, the postfire outcome was atypical successional pathways with concomitant declines in flora and fauna (fig. 4-4).

Many of these ecosystems can still be restored with the judicious reintroduction of fire (fig. 4-5), sometimes in combination with other chemical and mechanical methods, because the long association between fire and Southern vegetation has evolved species traits (table 2-1) that favor them in fire-prone ecosystems (Christensen 1977; Landers 1991). Providing a certain threshold limit has not been reached, the natural resiliency within these systems allows them to recover (Vogl 1976). However, once this threshold limit has been exceeded, nature can no longer rectify the situation. Thus, many components of the original ecosystem cannot survive long periods without fire (Garren 1943).

Most students of fire history agree that typical longleaf pine sites burned every 1 to 4 years prior to the arrival of Europeans, and then every 1 to 3 years until aggressive fire suppression activities began in the 1920s and 1930s (Landers 1991; Landers and others 1990). As typical upland sites grade toward mesic (wet) or xeric (dry and thus low rates of fuel accumulation), fire frequency decreases. Loblolly, slash, and pond pines were historically confined to wetter sites (Monk 1968) where a 3 to 4 year fire-free interval allowed saplings to become large enough to withstand low intensity fire. In developing a definition for old-growth wet-pine forests, Harms (1996) emphasized the fact that fire is necessary to establish and maintain these ecosystems. The historic shortleaf pine fire return interval is thought to have ranged from about 2 to 6 years on fertile, lower elevation sites to 6 to 15 years on drier, nutrient-poor sites where fuels take longer to accumulate. Annual burning was extensively practiced throughout the shortleaf pine region (Matoon 1915).

Fires in the depressional wetlands embedded in the understory fire regime types are typically stand-replacement. Wharton (1977) gives fire cycles for these types as follows: 3 to 9 years in herb bogs and high-diversity shrub bogs (including pocosins); 20 to 30 years in many cypress ponds, gum ponds and bog swamp; 20 to 50 years in low diversity (titi) shrub bogs; and 50 to 150 years in some cypress ponds and bay swamps.

Perturbations to these chronic low-intensity Southern pine fire regimes occurred in the form of episodic (often catastrophic) events such as blowdowns (fig. 4-6) and drought, which were precursors to fires of much higher intensity and severity (Myers and Van Lear

Figure 4-4—The forest floor accumulated from decades of fire exclusion completely consumed after an extreme drought-year fire on Myakka River State Park, Florida. Fetterbush (center), which typically resprouts prolifically after fire, was killed. Note weak resprouting of palmetto. Photo by Robert Dye, 1985.

Figure 4-5—Longleaf pine sandhill site following three prescribed fires during 6 years, Wekiwa Springs State Park, Florida. Note wiregrass in flower 1-year post burn. Photo by Jim Stevenson, 1983.

Figure 4-6—Growing-season burn in a 3 year rough on Francis Marion National Forest, 8 months after Hurricane Hugo. Photo by Ken Forbus, 1990.

1997). Explosive increases in southern pine beetle (*Ips* spp. and *Dendroctonus frontalis*) populations and subsequent pine mortality often either preceded or followed these fires. In nature, such infrequent events are often controlling factors that have profound effects limiting vegetative development past a certain point so that succession is a cyclic phenomenon (Vogl 1976).

Intentional use of fire to manage vegetation again became commonplace after World War II. Besides rough reduction, vast acreages were burned to: (1) improve native range (Halls and others 1952,1964; Hilmon and Hughes 1965a; Lewis 1964; Shepherd and others 1951), (2) control the relentless hardwood invasion into pine stands (Brender and Cooper 1968; Chaiken 1952; Ferguson 1957; Harrington and Stephenson 1955; Lotti 1955), and (3) manipulate wildlife habitat to favor herbs and forbs and replace unpalatable out-of-reach woody understory crowns with succulent, nutritious sprouts (Brennan and others 1998; Harlow and Van Lear 1981, 1987; Harris 1978; Wood 1982).

During the 1980s, an estimated 4 million acres (10 million ha) of forest land and 4 million acres of range and agricultural land were treated with prescribed fire each year in the Southern United States (Wade and Lunsford 1989). Prescribed fire is used by Southern resource managers to meet widely varying objectives: restoration and maintenance of ecosystems, reduction of hazardous fuels, reduction of wildfire size and suppression costs, reduction of firefighter risks, preparation of seedbeds and planting sites, facilitation of harvesting (fiber, plants, game animals, insects, and earthworms), enhancement of wildlife habitat, range improvement, control of insects and diseases, thinning overly dense stands, maintenance of scenic vistas, promotion of showy herbaceous species, manipulation of understory structure and composition, creating a safer environment for woods workers and equipment operators, favoring plant and animal species of special concern, eradication of vermin, control of exotic species, stimulation of fruit and fiber production, disposal of crop residues, and recycling of nutrients. The majority of treated acreage is for hazard reduction, wildlife habitat improvement, and range management. Nearly 2 million acres (0.8 million ha) of rangeland are burned annually in Florida alone. An increasing area is burned each year to maintain the function of fire in ecosystems, particularly on State and Federal lands. Today, prescribed fire is used to treat over 6 million acres annually, which is only a small fraction of the land that once supported the vast Southern pine forests.

Oak-Hickory Forests—The fire regime of the oak-hickory forest has varied spatially and temporally because of changing cultural influences before, during, and after Euro-American settlement. Before such influence, the fire regime for oak-hickory forests was dictated by the activity of Native Americans because they were the primary ignition source (Abrams 1992; Buckner 1983; Denevan 1992; Pyne 1997). Native Americans in the Central Hardwoods Region (Delcourt and Delcourt 1997, 1998; Olson 1996), the Appalachians and Piedmont (Van Lear and Johnson 1983), and the Northeast (Buell and others 1954; Day 1953) commonly used fire for numerous reasons throughout the year. Surface fires predominated and burned over large areas; only natural barriers or unfavorable weather stopped them. Hough (1877) thought the "oak openings," "barrens," and prairies east of the Mississippi resulted from Native American use of fire to promote grass growth and attract game. He stated, "Scarcely a year passes without the occurrence of fires of sufficient extent to attract public notice." Numerous authors (DeViro 1991; Patterson and Sassamen 1988; Stewart 1951, 1963; Van Lear and Waldrop 1989) have discussed the vast extent to which Native Americans used fire. One historian (Russell 1983) agreed that fire frequency was greater near camps and villages than would be expected by lightning, but found no strong evidence that Native Americans burned large areas in the Northeast.

Lightning fires are uncommon in many of these regions because thunderstorms occur primarily during the growing season, usually accompanied by rain (Barden and Woods 1974; Ruffner and Abrams 1998). However, lightning fires regularly occur in some areas such as the Piedmont of North Carolina (Barden 2000).

Presettlement fire frequencies are not known. Delcourt and Delcourt (1997, 1998) believe they varied considerably depending on closeness to Native American habitation. Other references point out that Native Americans maintained an extensive trail system throughout the East that was kept open with fire. Euro-American explorers reported many areas treated with annual and biennial fires (Barden 1997; Buckner 1983; Day 1953). Dendrochronological studies, which give conservative estimates, suggest fire return intervals of 7 to 14 years in the mid-Atlantic and Ozark regions (Buell and others 1954; Guyette and Day 1997). Cutter and Guyette (1994) reported a fire-return-interval of 2.8 years during 1740 to 1850 on a ridgetop in the Mark Twain National Forest. Presettlement fires in southern New England generally occurred during the spring and summer (Bromley 1935; Christianson 1969). Brown (1960) believes the prevalence of oak in Rhode Island is the result of a long history of fire. The fire regime was probably more pronounced in Southern areas than in Northern areas due to more favorable climatic conditions for ignition and spread, greater populations of Native Americans, and vegetation more conducive to burning.

The frequency and extent of Indian burning decreased substantially after white contact, which introduced new diseases and decimated their population by 90 percent or more over the next 100 to 150 years (Denevan 1992; Dobyns 1983; MacCleery 1993). As a result grasslands, savannas, and woodlands succeeded to closed forest (Buckner 1983; Denevan 1992; Dobyns 1983; MacCleery 1993, 1995; Pyne 1997) (fig. 4-7). Subsequent settlement of the oak-hickory forests by Euro-Americans, who used fire for many of the same reasons as the Native Americans, increased the frequency and extent of burning (Abrams 1992; Pyne 1997; Van Lear and Waldrop 1989). Fire return intervals were shortened to 2 to 10 years with many sites burning annually (Cutter and Guyette 1994; Guyette and Day 1997; Holmes 1911; Sutherland and others 1995; Sutherland 1997). For example, the barrens of Pennsylvania and Maryland were burned annually at least through 1731 (Tyndall 1992).

Presently, the fire regime of oak-hickory forests is characterized by infrequent, low-intensity surface fires that occur during the spring and fall. They are caused almost exclusively by humans, and burn small areas (Pyne and others 1996). Lightning is a minor ignition source (Barden and Woods 1974; Ruffner and Abrams 1998). Fire return intervals have lengthened from a few years to several millennia (Harmon 1982), the longest fire-free intervals in the history of the Central Hardwoods Region (Ladd 1991).

Fuels and Fire Behavior

Southern Pine Forests—Most studies show that the amount of dead understory and forest floor material less than 3 inches in diameter varies widely by site, overstory basal area, and time since last fire (table 4-2). Live and dead accumulated fuel loadings in slash pine and longleaf pine stands can increase five to ten fold from 1 year after a fire to 20 years later. In mature Southern pine forests on the Atlantic Coastal Plain, stand average forest floor fuel loadings ranged from about 1.5 tons/acre (3.4 t/ha) under an annual dormant-season fire regime to 13 tons/acre (29.1 t/ha) after 40 years (Boyer and Fahnestock 1966; Bruce 1951; Geiger 1967; McNab and others 1978; Southern Forest Fire Laboratory Staff 1976; Williston 1965). Live groundcover and understory fuels less than 1 inch in diameter ranged from about 0.75 tons/acre (1.68 t/ha) with annual burns, to over 11 tons/acre (25 t/ha) after 25 years (table 4-3). Halls (1955) reported that grass production varied from about 300 lb/acre (336 kg/ha) under dense longleaf-slash pine stands to 1,000 lb/acre (1,120 kg/ha) on open forest rangelands. Equations for predicting forest floor and understory fuel weights from stand factors and age of rough were developed by Williston (1965), Geiger (1967), and McNab and others (1978) that explain 70, 80, and 86 percent of the variation in fuel weight, respectively.

Figure 4-7—Oak-dominated stand with no recent history of fire on the Cumberland Plateau, eastern Tennessee. Photo by Tom Waldrop, 1982.

Table 4-2—Total litter loading (tons/acre ovendry basis) under slash pine and loblolly pine stands by stand basal area and age of rough (Southern Forest Fire Laboratory Staff 1976).

Basal area	Age of rough (years)								
(sq ft/acre)	1	2	3	4	5	7	10	15	20
Slash pine[a]									
30	1.5	2.5	3.4	4.2	4.8	5.9	7.0	8.1	8.4
50	1.6	2.8	3.8	4.7	5.4	6.6	7.9	9.0	9.4
70	1.8	3.2	4.3	5.2	6.1	7.4	8.8	10.1	10.5
90	2.1	3.5	4.8	5.9	6.8	8.3	9.9	11.3	11.7
110	2.3	4.0	5.4	6.6	7.6	9.3	11.1	12.7	13.2
130	2.6	4.4	6.0	7.3	8.5	10.4	12.4	14.2	14.7
150	2.9	5.0	6.7	8.2	9.5	11.6	13.9	15.9	16.5
175	3.3	5.7	7.7	9.5	11.0	13.4	16.0	18.3	19.0
200	3.8	6.6	8.9	10.9	12.6	15.4	18.4	21.1	21.9
Loblolly pine									
30	1.4	2.2	2.9	3.4	3.8	4.5	4.7	4.7	—
50	1.5	2.4	3.2	3.9	4.3	5.0	5.3	5.3	—
70	1.6	2.8	3.7	4.3	4.8	5.6	5.9	5.9	—
90	1.9	3.0	4.1	4.8	5.4	6.3	6.6	6.6	—
110	2.1	3.5	4.6	5.4	6.0	7.1	7.4	7.4	—
130	2.4	3.8	5.1	6.0	6.7	7.9	8.3	8.3	—
150	2.7	4.3	5.7	6.7	7.5	8.8	9.3	9.3	—
175	3.0	5.0	6.6	7.8	8.7	10.2	10.7	10.7	—
200	3.5	5.8	7.6	8.9	10.0	11.7	12.3	12.3	—

[a]Applies to stands with and without understory vegetation (McNabb and Edwards 1976).

Fuel loading trends are similar on Piedmont sites for shortleaf pine and mixed pine-hardwood (Metz 1954); shortleaf pine (Crosby 1961; Johansen and others 1981); loblolly pine (Southern Forest Fire Laboratory Staff 1976); loblolly, shortleaf, and Virginia pines (Metz and others 1970); loblolly pine and groundcover (Brender and Williams 1976); longleaf, loblolly, and shortleaf pines, mixed pine-hardwood, and wiregrass (Albrecht and Mattson 1977). Fuel loadings were highest in loblolly and longleaf pine stands, and appreciably lighter in shortleaf and Virginia pine stands. Fuel loadings in shortleaf and mixed shortleaf pine-hardwood stands in the mountains of North Carolina were substantially heavier than those on the Piedmont, partly because of a heavier understory component (Albrecht and Mattson 1977). Prediction equations that explain 80 percent of the variation in forest floor loadings and 90 percent for the groundcover were developed by Brender and Williams (1976).

Photo series publications are available showing fuels and estimated loadings for various Southern pine types (Lynch and Horton 1983; Ottmar and Vihnanek, in press; Sanders and Van Lear 1988; Scholl and Waldrop 1999; Wade and others 1993).

Resource managers usually prescribe conditions that limit fuel consumption to 1 to 3 tons/acre (2.2 to 6.7 t/ha)

Table 4-3—Understory vegetative loading (tons/acre dry weight) in the palmetto-gallberry type related to age of rough and understory height (Southern Forest Fire Laboratory Staff 1976).

Understory	Age of rough (years)							
height (ft)	1	2	3	5	7	10	15	20
1	0.4	0.4	0.5	0.6	0.9	1.4	2.6[a]	4.2[a]
3	2.6	2.6	2.7	2.8	3.1	3.5	4.7	6.4
4	4.5[a]	4.5	4.6	4.7	5.0	5.5	6.6	8.3
5	7.0[a]	7.0[a]	7.0	7.2	7.4	7.9	9.1	10.8
6	10.0[a]	10.0[a]	10.0[a]	10.2	10.4	10.9	12.1	13.8

[a]A situation not likely to be found in nature.

during passage of the flame front (fig. 4-8). Residual combustion can more than double these values, especially under drought conditions, or 5 to 6 years after a major disturbance when large down woody fuels have become punky. Percent litter reduction and energy release from backfires in palmetto-gallberry can be predicted given the weight and moisture content of the total litter layer (Hough 1968). Available fuel can be estimated knowing total fuel loading and moisture content (Hough 1978).

Annual and biennial prescribed headfires on the Atlantic and Gulf Coastal Plains typically move through the herbaceous groundcover with fireline intensities less than 500 Btu/ft/s (144 kW/m). Dead woody stems killed in the previous fire are often still standing and can substantially contribute to fire behavior. Where the canopy is open, midflame wind speeds often approach 4 to 5 mph. Short distance spotting up to 10 to 15 feet (3 to 4.6 m) is fairly common in herbaceous fuels such as wiregrass, broomsedge, and switchcane, especially as relative humidity drops below 35 percent. Rate of spread is always less than midflame windspeed on level ground (ignoring spotting) and typically falls between 200 and 1,500 feet/hour (60 to 460 m/hr) for heading fires and less than 150 feet/hour (46 m/hr) in backing fires. Short runs where headfire rate-of-spread is more than doubled are not infrequent under gusty winds. Headfire flame length can vary widely, from less than 0.5 feet (15 cm) to over 10 feet (3 m) in 2 year palmetto-gallberry roughs depending primarily upon windspeed, fine-fuel moisture content, fuelbed depth and porosity, and ignition pattern. Where the objective is to topkill hardwoods, backfires are often utilized because they concentrate released heat energy near the ground (Lindenmuth and Byram 1948).

In uniform 4 year old palmetto-gallberry roughs, headfire spread rates can exceed 0.5 mph (0.8 km/hr) with flame lengths in excess of 20 feet (6 m) and fireline intensities of 2,000 Btu/ft/s (578 kW/m) under "good" burning conditions and Lavdas Dispersion Index (Lavdas 1986) values above 70. Firebrands consisting primarily of dead palmetto fronds are common and often ignite spot fires 10 to 30 feet (3 to 9 m) ahead of the fire front as relative humidities drop below 35 percent. More detailed descriptions of fuels and fire behavior can be found elsewhere (Cheney and Gould 1997; Hough and Albini 1978; Johansen 1987; Wade 1995; Wade and Lunsford 1989; Wade and others 1993).

Oak-Hickory Forests—Leaf litter is the primary fuel that sustains fire. Loading and thickness vary depending on site, stand age, and time of year (Albrecht and Mattson 1977; Blow 1955; Crosby and Loomis 1974; Kucera 1952; Loomis 1975; Metz 1954). Hardwood leaves tend to cup and hold water after a rain, although the leaves of some species of oak tend to curl

Figure 4-8—Dormant-season prescribed fire backing through 12-year palmetto-gallberry rough in Collier County, Florida. Photo by Dale Wade, 1977.

and dry quickly in comparison to other hardwoods, allowing fire to run through oak litter when other hardwood fuel types are too wet to burn. Fuel loadings in like stands on comparable sites vary little longitudinally, but increase northward because decreasing mean temperatures slow decomposition. Litter loading and depth are greatest immediately after leaf fall in the autumn and decline until the following autumn.

Most stands have litter loadings ranging from 1 to 4 tons/acre (2.5 to 9.9 t/ha) and depths of 1 to 5 inches (2.5 to 12.7 cm), depending on time of year. For example, the forest floor under a 150 year old hickory stand (91 sq.feet basal area) on the Piedmont of South Carolina averaged 3.1 tons/acre (7.0 t/ha) and 4.1 tons/acre (9.2 t/ha) under a nearby 50 year old oak stand (62 sq.feet basal area) (Metz 1954). Annual litter production in fully stocked oak stands in Missouri averaged 2.1 tons/acre (4.7 t/ha) (Loomis 1975). The total forest floor loading (exclusive of material larger than 0.5 inch diameter) in the same stands averaged 8.3 tons/acre (18.6 t/ha), but 6.4 tons/acre (14.3 t/ha) in a younger stand (Crosby and Loomis 1974). In hardwood stands of several Northeastern States, the L, F, and H layers averaged 1.0, 3.3, and 5.9 tons/acre (2.2, 7.4, and 13.2 t/ha), respectively (Mader and others 1977). Generally duff contributes little to fire spread but does influence fire effects, especially during droughts.

Savannas typically contained grasses that were 3 to 6 feet (1 to 2 m) high according to early explorers (Barden 1997; Buckner 1983; Denevan 1992). Based on grass height, loadings may have ranged from 2 to 5 tons/acre (4 to 11 t/ha). After decades, fire exclusion allows the mid-story forest canopy to close, and herbaceous fuels are shaded out and become relatively unimportant as a fire carrier.

Small woody fuels are often abundant in young stands originating after a major disturbance. Woody fuels are less abundant in mid-successional and mature stands and increase in old-growth stands due to accumulation of large woody material. When present, ericaceous shrubs such as mountain laurel and rhododendron can burn with extreme fire behavior resulting in a mixed-severity or stand-replacement fire (Waldrop and Brose 1999). Many of the firefighter fatalities in hardwood forests have occurred because of the explosive nature of these fuels.

Postfire Plant Communities

Current and pre-Euro-American settlement vegetation types may have little in common because most of the original vegetation was either logged or cleared for agriculture prior to World War I. Many Eastern States currently have more acres in forest than they did 150 years ago.

Longleaf pine

Vegetation Dynamics—The longleaf pine ecosystem is distinguished by open pine forests, woodlands, and savannas. It is found on the Coastal Plain from Virginia to Texas, and in the Piedmont and Appalachian Highland (both Blue Ridge and Valley) physiographic provinces of Alabama and Georgia (Boyer 1990a; Wahlenberg 1946) (fig. 4-9). Longleaf pine tolerates a wide range of sites from wet, boggy flatwoods underlain with tight clays across xeric, deep-sand hills to thin stony soils on southerly-facing mountain slopes. Surface soils are typically acidic, tend to dry quickly after precipitation (especially the Quartzipsamments), and are characterized by a lack of organic matter and low fertility (Landers and Wade 1994). Vegetation that dominates these oligotrophic sites is resistant to biological decomposition because of its high C to N ratio but is amenable to thermal decomposition (Hon Tak Mak 1989). Key plants exhibit pronounced fire tolerance, are long lived, and are efficient at gathering nutrients and water, which reinforces their dominance and restricts the rate and spatial scale of vegetation change including species turnover. A feedback loop thus exists that includes climate, key plants, and vast expanses that are quick drying and topographically susceptible to disturbance; these factors all interact to maintain a chronic fire regime. Recurrent fire is

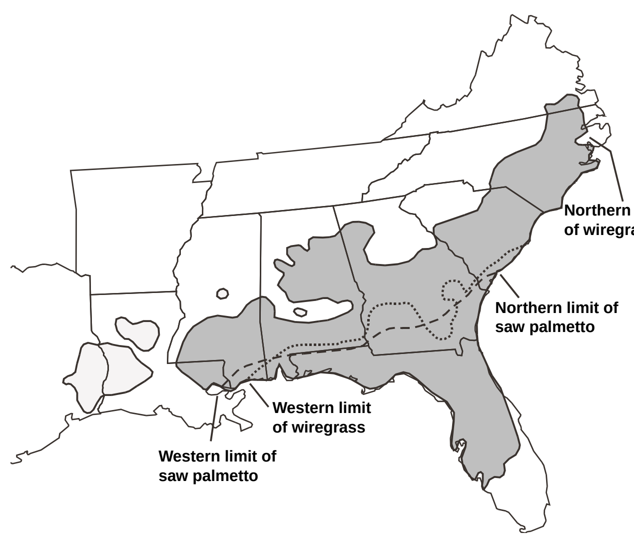

Figure 4-9—The longleaf pine forest region (after Stout and Marion 1993). The shaded area shows the distribution of longleaf pine (Critchfield and Little 1966). A dotted line indicates the landward extent of wiregrass and the western and northern limits of its distribution. A dashed line indicates the northern limit of saw palmetto.

crucial to perpetuation of longleaf pine ecosystems as noted by Andrews (1917) and many others since. These ecosystems persist and maintain their diversity because of, rather than in spite of, constant disturbance and infertile soils (Christensen 1993a; Landers and Wade 1994; Landers and others 1995; Wells and Shunk 1931).

The flora and fauna that dominate this ecosystem are those with well-developed adaptations to chronic fire. This fire regime maintained a two-tiered structure comprising an open longleaf pine overstory and an unusually diverse groundcover dominated by bunchgrasses (fig. 4-3). According to Peet and Allard (1993), this is one of the most species-rich ecosystems found outside the tropics. Overviews of the plant and animal communities that form this ecosystem can be found in Bridges and Orzell (1989), Harcombe and others (1993), Myers and Ewel (1990), Platt and Rathbun (1993), Skeen and others (1993), Stout and Marion (1993), and Ware and others (1993).

Longleaf pine has numerous traits that make it highly tolerant of fire (table 2-1). After an initial grass stage it puts on a growth spurt (called bolting) that quickly gets its terminal bud above potential flames. Bark thickens rapidly during the first year of height growth and protects stems from light surface fires. The large buds have a high heat capacity and are protected by a surrounding sheaf of long needles.

Fire results in small-scale temporary changes in light, water, nutrients, and space along the elevation-moisture gradient. These sites are further modified by frequent treefalls resulting from lightning, beetle activity, and faunal excavations primarily by the gopher tortoise (*Gopherus polyphemus*) and pocket gopher (*Geomys* spp.). Hermann (1993) described the importance of these microsites. Strong wind events and hydrologic extremes introduce ephemeral resource features over much broader areas. Thus a wide array of habitats is continuously available to vegetative change, which allows opportunistic plant species to coexist with more permanent ones. Without frequent fire, woody species overtop the herbs, strongly acidic pine needles accumulate forming a duff layer, nutrient dynamics change, and soil fertility increases, which all favor extrinsic species at the expense of endemic residents. About 191 taxa of vascular plants associated with the longleaf-bunchgrass system are threatened and endangered (Walker 1993), primarily because: (1) habitat has been paved over, (2) intensive site preparation associated with row crops took place prior to agricultural abandonment or when directly converting to other pine species, (3) frequent fire has been excluded, and (4) water tables are dropping. The fauna also contains many endemics that are either Federally or State listed as threatened or endangered.

Natural stands of longleaf pine may average from just one or two canopy trees per acre on extremely dry, deep sand ridges to over 150 per acre on good mesic sites (Schwarz 1907). Glitzenstein and others (1995) found that when burned on a 2 year cycle, longleaf pine canopy tree damage is primarily related to variation in fire intensity rather than month of application. Forest openings resulting from death of overstory trees are maintained by frequent fire until eventually colonized by shade-intolerant longleaf pine whose seed require a mineral soil seedbed. Germination takes place as soon after seed fall as moisture becomes adequate. The seedlings can withstand low-intensity fire the following growing season; fire is sometimes applied at this time to destroy seedlings of competing species to increase the longleaf pine component. If the longleaf pines are not yet fully established, they are often topkilled, but sometimes resprout (Grace and Platt 1995). With extended periods of fire exclusion, hardwoods dominate forest openings (Gilliam and Platt 1999).

Longleaf pine seedlings undergo a grass stage lasting several years where no height growth occurs, but a taproot develops and the rootcollar thickens. During the grass stage, seedlings are often infected by a fungal disease called brownspot needle blight (*Scirrhia acicola*), which weakens the plants so they eventually die. Fire controls this disease by consuming the infected needles and stimulating the seedlings to initiate height growth, perhaps due to the release of nutrients (Christensen 1977).

The bunchgrass and longleaf pine fuel mixture can burn within hours of a soaking rain so that an early afternoon lightning strike in a dead snag often smolders in the accumulation of dead bark, litter, and branches at its base resulting in a fire later that day or the next day. Thus the longleaf pine type transforms lightning strikes into fires, which then spread through the highly flammable herbaceous groundcover and needle litter throughout its ecosystem (Platt and others 1988b) and into neighboring ecosystems as well. It is not surprising that lightning is the leading cause of mortality of longleaf pine (Komarek 1968; Platt and others 1988b). Kirkman and others (1998) characterized the ecotone between upland longleaf pine and seasonally ponded wetlands. The hydric soil boundary was consistently upslope from the vegetation boundary, which led them to conclude that the abrupt changes in vegetation were likely related to fire periodicity.

Southern pines have been exploited since Euro-American colonization. During the first two decades of the 20th century, the remainder of the "original" Southern pine forest was heavily cutover, and then indiscriminately burned at least every spring to promote forage for free-ranging cattle (Stoddard 1962). These fires eliminated all pine regeneration except for grass-stage longleaf pine. However, reforestation of this species was prohibited by lack of an adequate longleaf pine seed source (its seed is too heavy to be effectively wind disseminated) and feral pigs that uprooted seedlings (Frost 1993). Fire exclusion was seen as the only choice that would allow reforestation of these lands. This policy allowed the aggressive, less fire-tolerant loblolly and slash pines to move from the more mesic sites where they had been confined under the previous chronic fire regime (Landers and others 1995). The result was a dramatic shift in the makeup of the emerging forest. Extensive type conversion by the pulp and paper industry to genetically groomed loblolly and slash pines with faster juvenile growth than longleaf pine further exacerbated these species composition shifts. The longleaf pine ecosystem, which once dominated about 75 million acres (30 million ha) (Chapman 1932), has been extirpated from over 95 percent of its historical area (Frost 1993; Outcalt and Sheffield 1996) and is listed as critically endangered by Noss and others (1995).

Associated Vegetation—Conifer associates of longleaf pine include loblolly pine, slash pine, and pond pine on wetter sites and shortleaf pine, Virginia pine, and sand pine on drier sites depending upon the fire return interval and fireline intensity, particularly during the juvenile growth stage.The groundcover comprises primarily bunchgrasses such as wiregrass along the Atlantic seaboard (Lemon 1949; Lewis and Hart 1972; Lewis and Harshbarger 1976) and little bluestem and slender bluestem from central Alabama westward (Grelen and Duvall 1966). Common woody understory species include saw palmetto, gallberry, and wax myrtle. The live foliage of all three shrubs is unexpectedly flammable because of the release of volatiles, which promote fire spread and higher fireline intensities. Numerous studies describe the relationship between fire and various longleaf pine-dominated plant assemblages (Harcombe and others 1993; Huffman and Werner 2000; Landers 1991; Lewis and Hart 1972; Peet and Allard 1993; Wells and Shunk 1931; Yahr and others 2000), between fire and indigenous fauna (Brennan and others 1998; Engstrom 1993; Folkerts and others 1993; Guyer and Bailey 1993; Jackson 1989), and between both flora and fauna (Stout and Marion 1993; Ware and others 1993; Weigl and others 1989).

Under natural fire regimes, dry-site oaks are generally the only hardwood associates to reach the midstory. Common species include turkey oak, bluejack oak, blackjack oak, and sand post oak. Based on current information, Rebertus and others (1993) hypothesized that both longleaf pine and these upland oaks are fire dependent and that longleaf pine becomes more important relative to the oaks as fires become more frequent and occur during the early growing season. Greenberg and Simons (1999) presented evidence showing oaks have been an integral midstory component of many high pine sites (Myers and Ewel 1990) in Florida for at least several centuries.

Disrupted fire regimes result in an invasion of other hardwoods such as sweetgum, oaks, hickories, common persimmon, and southern magnolia (Daubenmire 1990; Gilliam and Platt 1999) (fig. 4-10). These

Figure 4-10—Hardwoods are confined to the understory with annual dormant-season fire (on left) and form a midstory with fire exclusion (on right), Okefenokee National Wildlife Refuge. Photo by Ron Phernetton, 1992.

hardwoods form a midstory and prevent the shade-intolerant longleaf pine from reestablishing. Many of these hardwoods are somewhat resistant to low-intensity fires when mature (Blaisdell and others 1974). Understory trees are usually top-killed, but their rootstocks are able to withstand all but annual growing-season fires (Waldrop and others 1987). Where oaks form a midstory because of fire exclusion, Glitzenstein and others (1995) found that late spring and early summer burns increased oak topkill. Where variations in microsite result in patchy growing-season burns, hardwoods occasionally reach the overstory, but not in sufficient numbers to assert dominance. For example, Plummer (1975) determined from old records that the two most common hardwood associates of the 18th century longleaf pine forest in southwestern Georgia were oak and hickory, which comprised 1.0 and 0.5 percent of the stems, respectively, and longleaf pine 91 percent. The influx of exotics that are promoted by fire such as cogongrass (Lippincott 1997), Japanese climbing fern, and melaleuca (Wade 1981; Wade and others 1980) will be an increasingly serious problem.

Management Considerations—Fragmentation of the original landscape by conversion of sites to nonforest uses and an extensive road network, coupled with rapid, initial attack response times have dramatically changed fire patterns in this ecosystem. Lightning fires driven by shifting winds historically molded habitats that were much more structurally diverse than those managed today with rigidly scheduled line-fires set under predictable dormant-season weather conditions. The increasing use of variable growing-season fire regimes (currently about 15 to 25 percent of the total acreage), point-source ignitions (both aerial and ground), and "soft" firelines should help ameliorate this problem over time. Burn schedules that incorporate variability into season, frequency, and pattern of burn can be found in Robbins and Myers (1992). Streng and others (1993) and Olson and Platt (1995) suggested that many fires during the same season over time are necessary to produce changes in groundcover species composition.

Whenever the short fire-return interval is disrupted, pines that are more prolific seeders or have faster juvenile growth, outcompete the shade-intolerant longleaf pine. Burning the site while these competitors are still seedlings or saplings will selectively favor longleaf pine. However, once any of the multinodal Southern pines attain a basal diameter of about 2 inches (in 3 to 4 years), they become fairly immune to girdling by low-intensity fire (Wade 1993; Wade and Johansen 1987). Assuming no bud damage, longleaf, slash, and loblolly pines all usually survive complete crown scorch except during late summer and early fall when death often results, regardless of bud damage (Storey and Merkel 1960; Weise and others 1990). Any fire that kills buds during the growing season (crown consumption is a good visual indicator) greatly diminishes a tree's chances of survival (Wade 1985).

Desired fire-return intervals and timing of fire depend upon the objective. Early on, fire was advocated to facilitate regeneration (Long 1889), reduce hazardous fuel accumulations (Mattoon 1915), eliminate invading hardwoods (Ashe 1910), control brownspot (Siggers 1934), and manage rangelands (Wahlenberg and others 1939). Quail plantations along the Florida-Georgia border have burned annually to favor this game bird for close to100 years (Stoddard 1931). Showy plants such as orchids and pitcherplants (fig. 4-11) that frequent acidic depressions can be maintained by conducting annual late-spring burns (Komarek 1982). Long-term repeated prescribed fire studies include the 18 year effects of fire to control hardwoods in southern Alabama (Boyer 1993), understory plant community changes in a longleaf pine-wiregrass ecosystem in southern Georgia resulting from several decades of dormant-season burns at 1, 2, and 3 year intervals (Brockway and Lewis 1997), and understory plant response to 30 years of prescribed burning at various frequencies and seasons on the Santee Experimental Forest, South Carolina (Langdon 1981).

The effect of recurrent fire on Southern pine growth is important when considering fiber production (Chambers and others 1986). Studies have shown reduced growth after prescription fire (Boyer 1987; Boyer and Miller 1994; Zahner 1989) as well as increased growth after damaging fire (Johansen 1975). Grelen (1978) reported no height growth effects after introducing fire in 5 year old slash pine plantations and then burning at various frequencies through 9 years of age. Care must be exercised when interpreting the literature pertaining to the effects of fire on growth. Some fires were conducted at inappropriate times, and the methodology used to analyze results of some studies was flawed (Streng and others 1993; Wade and Johansen 1986). High fireline intensities that consume foliage (fig. 4-12) and high-severity fires that cause root damage (by burning at low duff moistures) most likely will cause mortality and reduce growth of survivors (Wade 1985). Ill-timed growing season burns and firing techniques will kill mature longleaf pine (Boyer 1990b). A guide recommended for firing is to consider a maximum air temperature of 99 °F, a minimum relative humidity of 34 percent, and flank firing technique. Reducing hazardous buildup of the forest floor by alternative methods such as commercial pine-straw raking (baled and sold on the retail landscape market as an ornamental mulch) will also reduce longleaf pine growth the following year (McLeod and others 1979), because scarce nutrients are removed instead of recycled. The literature pertaining to

Figure 4-11—Trumpet pitcherplant in a savanna burned the previous year on the Apalachicola National Forest, Florida. Photo by Sharon Hermann.

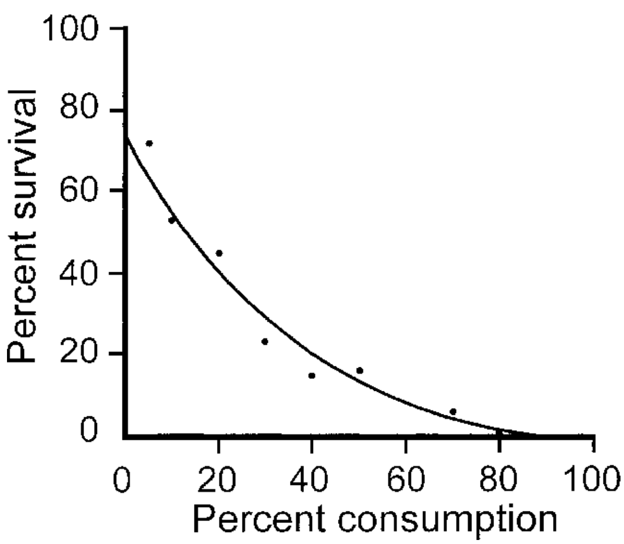

Figure 4-12—Percent survival of 1 to 8 year old planted loblolly pine after the first postfire year relates to percentage of live foliage consumption (Wade 1985).

the effects of fire on Southern pine was summarized and critiqued by Wade and Johansen (1986).

Chronic burning regimes result in an open parklike forest with a groundcover dominated by grasses and forbs (fig. 4-13). Not only are such plant communities aesthetically pleasing, but they also provide excellent wildlife habitat and high quality forage for cattle and deer. Many such landholdings are profitably managed for the simultaneous production of cattle, wildlife, and timber resources (Wade and Lewis 1987; Wade and Moss 1999). Detailed restoration and management options for this species across its range are presented in Brockway and others (1998), Farrar (1990), Hermann (1993), and Landers and others (1990, 1995); by accessing the Longleaf Alliance through the home page for the Auburn University School of Forestry; or by contacting the Tall Timbers Research Station in Tallahassee, FL, or the Joseph Jones Ecological Research Center in Newton, GA.

Slash Pine

Vegetation Dynamics—Slash pine (*var. elliottii*) is the chief conifer associate of longleaf pine on wet Coastal Plain sites throughout its natural range, which extends from coastal South Carolina through southern Georgia, the flatwoods of Florida, and across the Gulf Coastal Plain into Mississippi and Louisiana

Figure 4-13—Longleaf pine stand maintained by biennial growing-season fires near Thomasville, Georgia. Lack of longleaf pine reproduction is an artifact of decades of annual dormant-season burns that ended in 1986. Photo by Ken Outcalt, 1993.

(Lohrey and Kossuth 1990). It is a prolific seeder, has rapid juvenile growth, and becomes increasingly tolerant of fire with age. Because of these traits and the ease of handling seedlings, slash pine has been extensively planted. A 1980 survey showed that almost half of the 13 million acres (5.2 million ha) of slash pine resource had been planted (Haeussler 1983). Slash pine is perhaps the only native North American tree whose commercial range has undergone a successful large-scale expansion, through planting northward into Tennessee and westward into Texas where it is now naturalized (Stone 1983). Sheffield and others (1983) estimated slash pine occupied over 40 percent of the commercial forestland in Florida in 1980. A variety of slash pine (*densa*), which goes through an abbreviated grass stage somewhat like longleaf pine and is almost as fire tolerant, is confined to southern Florida and the Keys where it replaces longleaf pine (see chapter 7). General descriptions of the slash pine ecosystem can be found in Lohrey and Kossuth (1990), Grelen (1980), Myers and Ewel (1990), Stone (1983), and Stout and Marion (1993).

The most hydric slash pine sites are depressions such as bays, bayheads, titi swamps, and cypress pond margins embedded within the flatwoods matrix. Common associates are pond pine, loblolly pine, sweet bay, loblolly-bay, swamp bay, pondcypress and swamp tupelo, and bottomland hardwoods such as red maple, Carolina ash, American elm, sweetgum, and water oak. On such sites slash pine generally develops a pronounced buttress (comprised mostly of bark) that protects the tree from heat-girdling during drought fires. Species diversity is low. Characteristic understory species include cyrillas, sweetpepperbush, lyonias, and buckwheat tree. Little groundcover is present except for sphagnum, ferns, and various greenbriers.

Slash pine was historically confined to wet sites because its susceptibility to fire when young kept it from successfully competing with longleaf pine on upland sites. With removal of the longleaf pine and efforts to exclude fire, slash pine has successfully invaded many of these drier sites. The suspected historical successional scenario is that most fires favored slash pine. The occasional severe drought fire topkilled the woody vegetation (although a few slash pine survived), slash pine reestablished when a seed source was nearby, hardwood rootstocks resprouted (except in deep-burning peat fires), and the cycle repeated over the next several decades (Clewell 1980; Hodges 1980). Without fire the slash pine component eventually disappeared because a good seedbed was lacking and existing woody vegetation shaded out established slash pine seedlings.

Boggy flatwoods border creek swamps and acidic depressions. Fire enters these ecosystems during extended dry periods, which occur every couple of decades. Longleaf pine is the primary overstory associate with an understory of fire-adapted shrubs such as wax myrtle, gallberry, buckwheat tree, dahoon, and yaupon, and a ground cover of pitcherplants. These sites grade into slash pine flatwoods as the hydroperiod (the length of time the water table is above the soil surface) decreases with a several-inch rise in elevation and concomitant increase in fire frequency; with another several-inch rise, sites grade into pure longleaf pine stands where the fire regime is chronic. The more mesic sites support a rank understory of flammable vegetation, which produces some of the highest fuel loadings encountered in the South (table 4-2). Fuel consumption during extended dry-period wildfires can exceed 15 tons/acre (37 t/ha) after a decade or so of fire exclusion in fully stocked natural stands. Even high-intensity prescription fires do not approach this figure because burn plans almost invariably call for Keetch Byram Drought Index (Keetch and Byram 1968) values below 500 to avoid killing the overstory pine feeder roots that colonize the developing humus layer.

Extensive planting on old-field sites and decreased fire frequency have led to development of many other understory associations including saw palmetto, dwarf huckleberry, ground blueberry, and grasses such as Curtis' dropseed, broomsedge, and chalky bluestem and the creeping variety of little bluestem. Planting on xeric, nutrient-poor deep sands (ancient sand dunes) has resulted in associations with sand pine and numerous oaks such as post, blackjack, sand live, myrtle, bluejack, and turkey (Lohrey and Kossuth 1990). Planting of slash pine outside its natural range has resulted in many additional plant associations, virtually all of which are also fire-adapted.

Management Considerations—Harvesting of longleaf pine and fire exclusion policies in the first half of the 20th century, followed by several decades of extensive planting of slash pine coupled with dormant-season backing fires at wider than needed time intervals, resulted in a significant invasion of slash pine and its accompanying rank understory into former mesic longleaf pine sites. Most landowners were reluctant to use head fires, or in many cases any fire, because of high fuel loadings. Low-intensity backing fires rarely penetrated into the more mesic flatwoods because of high water conditions during the winter from frequent rainfall and reduced evapotranspiration.

During the 1980s, fire management of this pine resource changed dramatically due to the advent of aerial ignition, use of growing-season prescribed fires, recognition that the aesthetically pleasing longleaf pine ecosystems had virtually disappeared, and an increased appreciation of ecosystem management. The objectives included an increase in fire frequency to reduce the potential for catastrophic fire and the desire to directly or indirectly promote longleaf pine and its associated biologically diverse groundcover. On many lands, plowed firelines are no longer the norm, the fire-free interval has been shortened and high-intensity headfires are purposely run into the encroaching walls of underbrush to push the ecotone back toward historic boundaries (Ferguson 1998). These ecotones are not static, and as water tables are permanently lowered, they move further downslope.

The relationship between fire and major pests of slash pine has received limited study. However, studies suggest that use of prescribed fire can reduce the severity of annosus root rot (Froelich and others 1978) and fusiform rust (Siggers 1949; Wade and Wilhite 1981), two serious diseases of slash pine. Southern pine beetle infestations are not influenced by the judicious use of fire, but fires that result in severe crown scorch or root damage are often a precursor to attack.

Fire management in slash pine stands is straightforward. Fire has to be withheld for 3 to 5 years until sapling girth at ground level exceeds 2 inches (Johansen and Wade 1987a). After 8 or 9 years, slash pine usually recovers from complete crown scorch during all seasons except early fall (Weise and others 1990); however, its growth will be severely impacted (Johansen and Wade 1987b). Slash pine will not recover if buds are killed by complete consumption of foliage. Eye-level winds within stands are necessary to give the flame front direction and dissipate heat, but the controlling variable for effective use of fire is forest floor moisture content; if it is too high, fires will not back, whereas if too low, significant damage occurs to feeder roots. The water table level as measured by the Keetch-Byram Drought Index (KBDI) should be considered in planning prescribed fires because during severe droughts (KBDI above about 600), fires will burn through boggy flats and bayheads consuming underlying organic soils and destroying hardwood rootstocks as well as those of overstory trees. General guides for using fire in slash pine can be found in de Ronde and others (1990) and Wade (1983).

Loblolly Pine

Vegetation Dynamics—Loblolly pine dominates about 30 million acres (12 million ha) from New Jersey south to Florida and then west to Texas with excursions into Oklahoma, Arkansas, and Tennessee interrupted only by the flood plain of the Mississippi River (Baker and Langdon 1990). Loblolly pine historically was confined to much of the same wet landscape as slash pine for the same reason—its susceptibility to fire when young. It is also common along stream bottoms in the Piedmont where fire-free intervals historically exceeded 5 to 6 years. With increased fire protection, acreage dominated by this species dramatically increased as it seeded into former longleaf pine sites and abandoned agricultural lands. Loblolly pine was planted even more extensively than slash pine. It is currently the leading commercial tree in the Southern United States, comprising more than half of the standing pine volume in the region (Baker and Langdon 1990).

Loblolly pine is currently found on many of the same sites as slash pine except deep sands, but because of its greater range it occurs in association with many more species, especially where fire frequency has been reduced. Wahlenberg (1960) lists over 60 principal associates. Descriptions of various ecosystems where loblolly is a major component can also be found in Baker and Langdon (1990), Crow (1980), and Skeen and others (1993). On drier sites, longleaf, shortleaf, and Virginia pines; southern red, white, post, and blackjack oaks; mockernut hickory; and common persimmon are all common overstory associates. At the other end of the elevation-moisture gradient, associates include slash pine and pond pine; water, willow, southern red, swamp chestnut, and laurel oaks; southern magnolia; swamp tupelo; American elm; and red maple.

Because loblolly pine favors sites where soil moisture is not limiting, plant community composition usually includes a dense species-rich understory. Common understory hardwoods and vines usually resprout after fire. Herbaceous groundcover is typically sparse and includes bluestems, panicums, and longleaf uniola. Oosting (1944) compared differences in vegetation between an unburned area and two levels of fire intensity 9 years after a wildfire in a 35 year old stand of old-field loblolly pine. Species composition was about the same on all three areas. Although the crown fire killed virtually all overstory pines, many pine seedlings established and were successfully competing with hardwood sprouts suggesting that the future stand would again be mixed pine-hardwood. Twenty years of prescribed burning on the Georgia Piedmont at a 4 to 5 year cycle showed that the understory was drastically reduced and wildlife habitat improved (Wade and others 1989). Species composition in comparison to adjacent unburned stands was not altered. The same trends were noted on the Coastal Plain of South Carolina except where annual growing-season fires extirpated some hardwood species (Langdon 1981; Waldrop and others 1987).

Management Considerations—The key to managing loblolly pine is control of understory hardwoods, which is usually accomplished with periodic fire (fig. 4-14), mechanical methods, and chemicals (Chaiken and LeGrande 1949; Chapman 1947; Grano 1970; Harshbarger and Lewis 1976; Trousdell 1970). Where precipitation is a limiting factor, especially toward the western edge of its range, hardwood control can increase the growth of overstory pine (Bower and Ferguson 1968; Grano 1970). Loblolly pine is easily killed by fire when young, although seedlings will sometimes resprout at the root collar. On all but the poorest sites, juvenile height growth is several feet per year and its bark rapidly thickens, becoming resistant to light surface fires by the time stem diameter at ground level reaches 2 inches (5 cm) (Wade 1993).

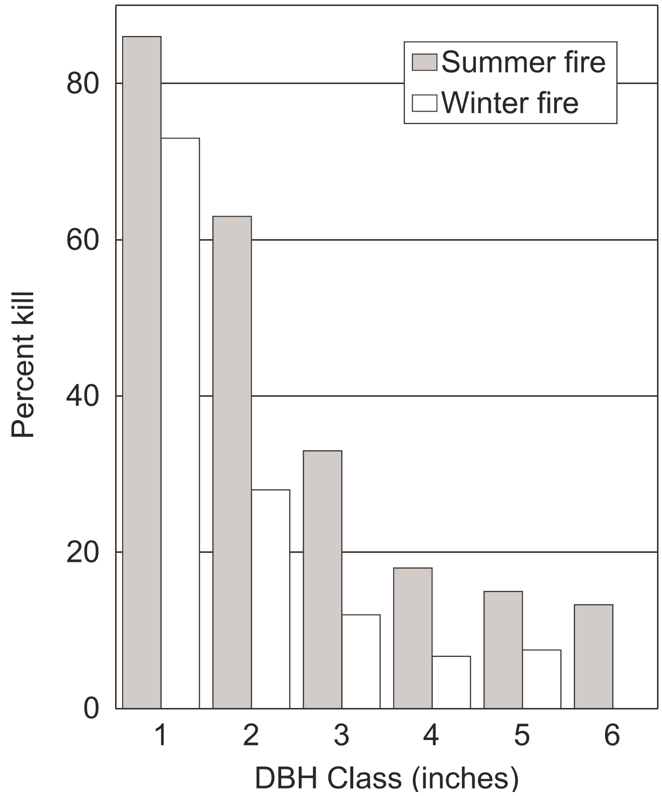

Figure 4-14—Growing-season prescribed fires topkilled more hardwoods and larger ones than dormant-season fires. Data collected on Brender Demonstration Forest, Georgia.

Operational underburning in loblolly pine is usually not advocated until stands reach about 10 years of age, but the sooner prescribed fire is used, the easier it is to topkill competing hardwoods. If care is taken in selecting burning conditions and in executing the burn to avoid crown scorch, plantations can be safely burned at half this age. Fire has been advocated as an integral part of the conversion from low-grade hardwood stands that developed with fire exclusion, back to loblolly pine or shortleaf pine. Planted loblolly exhibited increased growth on sites where hardwoods were cleared and burned (Applequist 1960). Numerous authors including Brender and Cooper (1968) and Cooper (1973b) described the use of fire to prepare mineral soil seedbeds to facilitate natural regeneration of loblolly pine.

Low-intensity, low-severity fires can be used to prepare seedbeds on steep slopes without triggering erosion problems (Douglass and Van Lear 1982); Van Lear and others 1985). The benefits to wildlife from repeated prescribed fires in loblolly pine stands are described well by Cushwa and Redd (1966) and Cushwa and others (1966). Guides to the management of this species include Brender (1973), Schultz (1997), and Wahlenberg (1960).

Shortleaf Pine

Vegetation Dynamics—The range of this species extends from southern New York southwest across southern Pennsylvania, Ohio, Indiana, Illinois, and Missouri, then south through eastern Oklahoma and Texas, and all States to the east including northern Florida (Mattoon 1915). It has the widest range of any of the eastern pines, occurring in 22 States. It occupies a wide variety of soils and environmental conditions but does not tolerate poorly drained sites. Shortleaf pine does well only on mineral soil seedbeds. It is a prolific seeder, often forming dense sapling stands that are favored over competing hardwoods by frequent fire.

Shortleaf pine can sprout repeatedly from its base if the tree is topkilled (see chapter 2). Trees 30 years old can produce basal sprouts (Little and Somes 1956); however, Matoon (1915) and Wakeley (1954) reported that this species looses its ability to sprout when less than half that age. Many sprouts can arise (70 or more), but virtually all die once a central leader assumes dominance (Komarek 1982). In a 1912 survey of shortleaf pine in Arkansas, few stands were found that originated from seedlings because of pervasive fire. There were more than three age classes of coppice in many stands (Matoon 1915). Ability to resprout, abundant seed crops, rapid juvenile growth (especially of sprouts), and a low resin content of the wood make this species markedly tolerant of fire (Mattoon 1915). Trees larger than 0.5 inch (1.3 cm) dbh are somewhat resistant to fire; mortality is negligible once trees reach 4 inches (10 cm) dbh (Walker and Wiant 1966). Like other Southern pines, trees over 5 feet (1.5 m) tall rarely die when crown scorch is less than 70 percent and buds are not killed by consumption of foliage.

Loblolly pine is the chief associate of shortleaf pine at lower elevations throughout the Mid-South and Southeast. Loblolly pine dominates the heavier, moist soils while shortleaf pine dominates the lighter, drier soils. Loblolly drops out at about 400 feet (122 m) elevation in the Ozarks and Ouachitas, resulting in pure stands of shortleaf pine up to about 2,000 feet (610 m) on south-facing slopes, above which hardwoods begin to dominate with shortleaf pine disappearing at about 3,000 feet (914 m). In the Appalachians and Upper Piedmont, Virginia pine replaces shortleaf pine on drier nutrient-poor sites east of the Appalachian divide. On the New Jersey Coastal Plain, pitch pine is the chief associate of shortleaf pine. Common hardwood associates are oaks (red, black, post, and chestnut), hickories (pignut and mockernut), sweetgum, yellow poplar, red maple, common persimmon, flowering dogwood, and sassafras.

Management Considerations—Mineral soil seedbeds are preferred but not required for seed germination (Boggs and Wittwer 1993; Ferguson 1958; Loyd and others 1978). Burns that expose mineral soil seedbeds may compact the surface soil as observed in the Arkansas mountains (Bower and Smith 1962). Care should be taken when using fire in hilly terrain because of the potential for erosion, especially on drier sites in the western part of its range (Moehring and others 1966). When low-severity fires are used to avoid consuming the humus layer, soil movement is negligible, even on steep Appalachian sites (Van Lear and Danielovich 1988). Shortleaf pine competes poorly with other plants (Williston and Balmer 1980). Periodic low-intensity fires, herbicides, or both are necessary to control the relentless encroachment of hardwoods and improve growth of the pines (Bower and Ferguson 1968; Crow and Shilling 1980; Ferguson 1957; Grano 1970; Hodgkins and Whipple 1963; Little and Moore 1950; Rogers and Brinkman 1965; Yocom 1972). Understory hardwoods deplete soil water in pine stands (Zahner 1958).

Somes and Moorhead (1950) showed that prescribed fire does not reduce the yield from oak-pine stands in New Jersey. Additional evidence showed that timber harvest and prescribed fire are not nutrient depleting practices and can enhance soil nutrient levels (Masters and others 1993). Phillips and Abercrombie (1987) demonstrated that excellent stands of shortleaf pine and mixed hardwoods could be produced by the fell and burn technique. Guides for managing shortleaf pine include Chen and others (1975, 1977), Nickels and others (1981), and Walker and Wiant (1966).

Oak-Hickory Forests

Pre-1900 Succession—Frequent fires ignited by Native Americans maintained open oak-hickory forests with a groundcover of grasses and forbs (fig. 4-15). Oaks and hickories were favored because of their thick bark. These species dominated the canopy as old, large, fire-resistant trees. Densities of dominant trees probably varied from 20 to 40 per acre. Shrubs, understory trees, and woody debris were rare (Barden 1997; Buckner 1983; Denevan 1992; Pyne 1997). Hardwood regeneration comprised seedling sprouts dominated by oak and hickory because these species initially emphasize root development over stem growth and have the ability to sprout repeatedly (Barnes and Van

Figure 4-15—Oak-dominated stand after 10 annual spring burns on the Cumberland Plateau, eastern Tennessee. Photo by Ivan Thor, 1976.

Lear 1998; Brown 1960; Van Lear 1991). With fire excluded for a few years, the well-developed rootstocks sent up vigorous stems that often developed sufficient size and bark thickness to withstand future fires. Where windstorms blew down trees over large areas, the replacement stand was even aged. Consequently, the forest was uneven aged, consisting of even aged patches.

Post-1900 Succession—The exclusion of fire from fire-dependent oak ecosystems should be considered a catastrophic disturbance according to Packard (1993). Reduction of fire has profoundly changed the oak-hickory forest by allowing the forest to succeed to mixed mesophytic and northern hardwood species such as red maple, eastern white pine, sugar maple, and beech (fig. 4-7). In the absence of fire, these species become established in the understory, grow into the midstory, and eventually change the composition of the canopy. Stem densities are often hundreds per acre. During the growing season, the dense shade from these fire-sensitive species reduces the abundance and richness of forbs and grasses and inhibits development of oak and hickory regeneration. Consequently, when a dominant oak or hickory dies, its reproduction is not capable of sufficient growth to capture the canopy opening. Instead, the growing space is filled by mesophytic and northern hardwood species (Abrams and Downs 1990; Crow 1988; Lorimer 1985; McGee 1984). According to Olson (1996), the brushy character of many sites is the result of an interruption in the chronic fire regime that allows shrubs and hardwoods to capture the site. When the area again burns several years later, these stems are top killed producing a dense growth of sprouts that can dominate the site for decades, especially with occasional fire.

On drier mountainous sites, fire exclusion allows ericaceous shrubs such as mountain laurel and rhododendron to move from riparian areas into upland forests (Elliott and others 1999). These shrubs are shade tolerant and evergreen, shading the forest floor throughout the year. Hardwoods cannot regenerate beneath them (Baker and Van Lear 1998), and without disturbance, these heath thickets are the climax plant community on some sites. Although the forest floor rarely dries enough to support surface fire, the ericaceous shrub layer is flammable; and when it burns, it typically supports intense, stand-replacement fires that alter successional pathways, reduce site productivity, negatively impact involved streams, and threaten human life and property (such slopes are favored building sites). Altered fire cycles have also impacted the "low elevation rocky summit" vegetation type where fire historically maintained the hardwood scrub savanna (Hallisey and Wood 1976). Fire exclusion over the past 50 years resulted in an increased hardwood overstory and a dramatic decline in herbs such as blazing star and some woody scrub species such as bear oak (Barden 2000).

Management Considerations—Until recently, foresters failed to appreciate the role of fire in maintaining open oak-hickory forests and in facilitating regeneration of these species (Lorimer 1993). Regeneration of oak was attempted only with timber harvesting and herbicides, which generally hastened the successional replacement of oaks by mixed mesophytic species (Abrams and Scott 1989). Research indicates that fire can be used in hardwood stands to establish and release oak-hickory regeneration (Barnes and Van Lear 1998; Brose and Van Lear 1998, 1999; Christianson 1969).

Understory burning of mature, uncut hardwood stands can help establish oak and hickory regeneration by preparing a seedbed (Barnes and Van Lear 1998). Acorns and hickory nuts are often buried by wildlife, particularly squirrels and blue jays, which prefer burned areas because of the thin root mat. The fires also top-kill or eliminate many of the shrubs and small trees that shade the forest floor. In a less shaded environment, the acorns and hickory nuts germinate and the new seedlings begin developing their root systems. Eventually, the regeneration replaces canopy trees. In this approach, fires are initially applied at a frequent interval (annual or biennial) depending on season of burn and severity of the shade. Once oak seedlings are established, fire is withheld for a few years (Cottam 1949), and then periodically reapplied once or twice a decade. This minimizes mortality of the oak regeneration by allowing time for root systems to develop. This approach may take 15 to 20 years for results to be apparent.

If oak and hickory regeneration is present in the understory, a two-step shelterwood harvest combined with a prescribed fire can be used for release (Brose and others 1999a,b). The initial shelterwood cut reduces the basal area to about 50 sq. feet/acre (11.5 sq. m/ha) removing low-value stems. The regeneration is allowed to develop for 3 to 5 years. During this time, oak and hickory regeneration develop large root systems but exhibit little height growth while their competitors do the opposite. When the root collar diameter of the oak regeneration is about 0.75 inch (2 cm), a growing-season prescribed fire with flame lengths of 3 to 4 feet (about 1 m) is used to kill the regeneration layer. This treatment will completely kill the less fire-tolerant competing hardwoods (Christianson 1969), invading eastern white pine (Blankenship and Arthur 1999), and rhododendron and mountain laurel. Few oaks and hickories will be killed by the fire, and most will sprout and grow vigorously. Regeneration should be inventoried 2 to 3 years later to determine whether additional fires are needed.

When using the shelterwood-burn technique, care must be taken to protect dominant oaks from basal fire damage. Directional felling during the logging operation is recommended so that the resultant slash is not abutting the trees. Otherwise, slash must be removed from the bases of dominant oaks to prevent fire damage. Generally, damage to dominant oaks is not a problem when burning in uncut stands because fuel loadings are considerably lighter. Graphs or equations can be used to predict mortality of several oak species after fires of varying intensity (Loomis 1973). The shelterwood-burn technique appears to be a reasonable mimic to the disturbance regime of oak-hickory forests before Euro-American influence. It has considerable value as a silvicultural method, a wildlife management tool, and a means for restoring habitats such as oak savannas and open woodlands.

Mixed Fire Regimes

Major Vegetation Types

The mixed fire regime best represents the presettlement fire history for several hardwood and conifer dominated ecosystems. The conifers include pitch pine and Virginia pine of Kuchler's oak-pine association and pond pine, a dominant tree of the pocosin association (table 4-1). The conifer types fit the mixed fire regime because fire intensities are generally greater than in the understory fire regime and cause mortality ranging from 20 to 80 percent of the overstory. The hardwood ecosystems comprise mesophytic hardwoods, Northern hardwoods, and elm-ash-cottonwood ecosystems (table 4-1). Although the hardwoods are prone to fire injury, many survive numerous fires before eventually being girdled. These fires tend to be low-intensity due to less flammable fuels than found in ecosystems having a substantial conifer component. We believe that the low-intensity presettlement fires that wounded and killed many trees did not cause enough mortality (>80 percent according to our criteria) to be considered stand-replacement.

Pines

Pitch Pine—Pitch pine grows on poor, generally sandy, gravelly, and shallow soils, primarily south of the glaciated region in southern New England in a fairly wide swath following the Appalachians and Upper Piedmont into Georgia where it occurs below 3,000 feet (914 m) elevation (Little 1959; Little and Garrett 1990). In New Jersey, pitch pine commonly exists in two forms, as a tree interspersed with hardwood trees (Pine Barrens) or as a member of a scrub oak community (Pine Plains).

Virginia Pine—The natural range of this species stretches from New Jersey across southern Pennsylvania to Indiana, then southward into central Alabama, and then northeasterly up the eastern slope of the Appalachians with more than half the standing inventory in western Maryland, Virginia, and North Carolina (Sternitzke and Nelson 1970). Virginia pine characteristically occupies poor sites where it often forms pure stands. Common associates include shortleaf, loblolly, and pitch pines; eastern redcedar; numerous oaks (SAF cover type Virginia pine-southern red oak); and other hardwoods.

Pond Pine—This forest cover type (SAF 98) stretches along the Coastal Plain from New Jersey to Alabama dominating poorly drained sites characterized by organic soils such as pocosins, bays, and shrub bogs where it often forms pure stands (Wenger 1958). About 80 percent of the pond pine forest is in the Carolinas (Sternitzke and Nelson 1970). Pond pine communities are often referred to by the understory vegetation such as shrub bogs or pocosins. The Native American name pocosin means swamp on a hill and they are just that. They occur on divides between rivers and sounds but are not alluvial. They all have long hydroperiods, burn periodically and are underlain by sandy humus or organic peat or muck soils (Richardson and Gibbons 1993). Pocosins, shrub bogs, and Carolina bays are often found within the loblolly pine, slash pine, slash pine-hardwood, and sweetbay-swamp tupelo-redbay cover types. On well-drained sites, pond pine is usually a minor component.

Hardwoods

Mixed Mesophytic Hardwoods—These forests occupy the transition zone from the oak-hickory forest to the northern hardwood forest. They are among the most diverse in the United States containing more than 30 canopy tree species. This type lies west of the Appalachians and transitions from the more northern sugar maple-beech-birch forest in northern West Virginia, southwestern Pennsylvania, and southern Ohio southward down the Allegheny Mountains, across the Allegheny Plateau including all of the Cumberland Plateau, and into northern Alabama where it transitions to the oak-hickory-pine type of the Southern Mixed Hardwood Forest. Common overstory species include sugar maple, red maple, basswood, northern red oak, chestnut oak, white oak, yellow poplar, American ash, silverbell, yellow birch, southern magnolia, Blackgum, black walnut, beech, yellow buckeye, and butternut.

Northern Hardwoods—The maple-beech-birch FRES ecosystem type, commonly known as Northern hardwoods, occurs on mesic and fire protected sites in the Lake States, Northeast, and Southeastern Canada (fig. 4-16). The dominant hardwood species include sugar maple, yellow birch, beech, and basswood in the Midwest. Northern hardwoods mix with boreal spruce and fir to the Northeast and with eastern hemlock, eastern white pine and northern oaks to the south and west. Component species, especially beech and sugar maple, extend south at mid elevations in the Appalachian Mountains to western Virginia and North Carolina where they occur in Kuchler's mixed mesophytic forest type.

Bottomland Hardwoods—This is the FRES elm-ash-cottonwood ecosystem type that occurs in narrow belts along major streams or scattered areas of dry swamps. The major portion is on the lower terraces and flood plains of the Mississippi, Missouri, Platte, Kansas and Ohio Rivers (Garrison and others 1977). This type comprises Kuchler's Southern and Northern flood-plain forest types and the elm-ash forest. Nineteen SAF forest cover types are included in these bottomland hardwood forests (Shartz and Mitsch 1993). Length of hydroperiod, which determines the anaerobic gradient (Wharton and others 1982), rather than fire frequency, determines plant distribution. Common canopy species include numerous oaks, sugarberry, American elm, eastern cottonwood, green ash, sweetgum, sycamore, and in deeper water, swamp and water tupelos, and bald cypress.

Fire Regime Characteristics

Pitch and Virginia Pines—Mixed severity fires were probably prevalent over much of the range of pitch and Virginia pines. Where Native American burning was common, pitch pine existed as an understory fire regime type. Understory fires were common in pitch pine forests where burning by Native Americans resulted in a 2 to 10 year fire interval. This

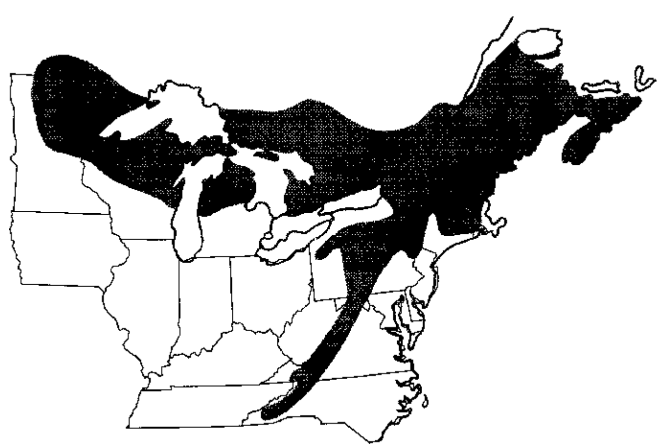

Figure 4-16—The extent of northern hardwood forests in northeastern North America. Redrafted from Bormann and Likens (1979).

frequency maintained stands with relatively large pines, scattered smaller pines and oaks, and little understory besides low ericaceous shrubs and herbs (Little 1946, 1973). Today, the mixed fire regime type applies, at least in the New Jersey Pine Barrens, because fire return intervals are longer and the majority of wildfires occur during the growing season when damage is greater. The historical fire regime in Virginia pine is unknown but was probably less frequent and resulted in higher mortality.

Pond Pine—Most pocosins burn on a 20 to 50 year cycle (Christensen and others 1988). On better sites, fire-return intervals range from about 3 to 10 years and at the short end, can result in pine savanna with a grass understory. On organic soil sites, such short return intervals result in herb bogs. Historically, more frequent fire in the adjacent longleaf-dominated uplands killed encroaching seedlings thereby confining this species to wetter areas. These wet sites, however, burned whenever they were dry enough. The rank shrub layer characteristic of these mesic areas comprises many ericaceous evergreen shrubs that tend to burn intensely, resulting in the topkill or death of all vegetation except pond pine. Pond pine has the ability to resprout from its base (fig. 4-17) and along its stem and branches (Wenger 1958); thus, its aboveground stem survives higher intensity fires than stems of other pine species. This trait allows the species to dominate wet areas such as pocosins, which support intense fires. Summer fires during severe drought usually eliminate the pond pine as well, because the underlying organic soil burns, destroying the root systems.

Mixed Mesophytic Hardwoods—Although little is known about presettlement fire, it appears that fire was much more common in the mesophytic forests west of the Appalachian divide than in those to the east. Harmon (1984) reported the fire return interval on south-facing slopes in extreme western Great Smoky Mountain National Park averaged 10 to 12 years between 1850 and 1940 when fire exclusion was begun in earnest. In a summary of fire in the Appalachians, Van Lear and Waldrop (1989) stated, "Forests of the Southern Appalachians probably did not burn as frequently as the pine-grasslands of the adjacent Piedmont. However, there can be no doubt that they did burn periodically." Buckner (1989) described fire's evolutionary importance in determining the vegetative mosaic of this region. Harmon and others (1983) thought that fires in the mixed mesophytic region were small and restricted to drier sites.

Northern Hardwoods—Although data are limited, evidence suggests that fires rarely occurred in presettlement Northern hardwood forests (Foster and Zebryk 1993; Patterson and Backman 1988). Fire return intervals of many centuries are consistent with

Figure 4-17—Pond pine basal sprouts 1 year after wildfire topkilled the overstory. Photo by Dale Wade, 1972.

land survey records (Siccama 1971; Lorimer 1977) and paleoecological data (Patterson and others 1983) in the Northeast. Lorimer (1977) calculated a fire rotation of 806 years, but argued that this figure was too low due to the effect of land clearing on his data. Based on a study of maple-beech forests in Ohio, Runkle (1990) concluded that the minimum fire-return interval was greater than a canopy generation. In the Bigwoods maple-basswood forest of Minnesota and Southeastern Ontario, fire probably occurred more frequently due to Native American burning practices that allowed prairie fires to spread eastward into the Bigwoods (Grimm 1984; Vankat 1979).

Where conifers such as hemlock and spruce were substantial components of the hardwood forests, stand-replacement fires probably occurred more often (Nichols 1913), in which case the stand-replacement fire regime may be a better representation of presettlement fire. Portions of the Northern hardwood forest were visited so infrequently by fire prior to settlement that a strong case could also be made to place this forest type in the nonfire regime (table 4-1).

Bottomland Hardwoods—The historical role of fire in the bottomland hardwood ecosystem is unclear. In Mississippi, Lentz (1931) stated that low- to moderate-intensity wildfires were frequent and that 80 to 90 percent of the Mississippi delta hardwood forest showed evidence of damage. Gustafson (1946) and Toole (1959) presented evidence of disastrous consequences to the hardwoods from repeated fires. In Louisiana, Kaufert (1933) dated fire scars on stumps back prior to the Civil War, but based on the recollections of "old-timers," he did not think widespread burning of these bottoms occurred until about 1890. Low-intensity fires are the norm in these forests because fuel loadings are generally light (except after damaging wind storms) due to rapid decomposition on these moist, humid sites. In the canebrakes, fire intensity was much higher although fire severity was low except during drought. Large fires occur only after extended drought, usually a dry fall followed by a dry spring.

Fuels and Fire Behavior

Pines—The typical lowland wildfire in **pitch pine** on the Pine Plains of New Jersey advances as a wall of flame consuming overstory pine crowns and leaving a stubble of shrub skeletons unless the underlying organic soil is also consumed, which occurs during severe drought fires (Little 1979). Fires in the Pine Barrens tend to be of lower intensity and more like fires elsewhere in pitch pine stands. Even on steep slopes of the Southern Appalachians where pitch and Table Mountain pine grow together, more than 20 percent of the overstory trees typically survive, although more intense stand-replacement fires occasionally occur. The short needles of **Virginia pine** form a relatively compact forest floor, which dries slowly and is conducive only to light surface fires (Little 1974).

Fuels in pocosins occupied by **pond pine** comprise varying proportions of shrubs, switchcane, and grasses. High intensity fires can occur where high fuel loadings accumulate, often including a rank growth of ericaceous shrubs. Typical live fuel and litter loadings are 6 to 8 tons/acre (13 to 18 t/ha) in low pocosins (about 4 feet high) (fig. 4-18), 8 to 10 tons/acre (18 to 22 t/ha) in medium height pocosins (about 5 feet high), and upwards of 15 tons/acre (34 t/ha) in high pocosins (about 14 feet high) (Wendel and others 1962). Some pocosin sites are depicted in a stereo photo series (Ottmar and Vihnanek, in press). The probability of blowup fires occurring in pocosins ranges from low in low pocosins to high in high pocosins.

Northern Hardwoods—Hart and others (1962) estimated average annual litterfall in a New Hampshire Northern hardwood stand at 1.4 tons/acre (3.16 t/ha). Leaf litter decomposition rates reported as half-times (years required to lose one-half of the original dry weight) ranged from 1.1 years for yellow birch to 2.5 years for beech, which are high (Gosz and others 1973). Accumulated duff is typically 2 to 3 inches (5-8 cm) in depth (Hart and others 1962).

Gore and Patterson (1986) sampled downed wood in Northern hardwood stands including a large tract of old-growth Northern hardwoods, and a recently clear-cut stand in New Hampshire. Loadings of material <1 inch (2.5cm) in diameter were low, ranging from 0.4 to 2.7 tons/acre (1 to 6 t/ha) across all but the recently cut stand (table 4-4). The total mass of downed wood declined precipitously in the first 10 years following cutting and stabilized at 18.7 tons/acre (42 t/ha) in the old-growth stand (fig. 4-19). Patterson and others (1983) reported similarly low total woody fuel loadings for Northern hardwood stands burned in 1947 and about 1880 in Acadia National Park, Maine. Total dead fuel loadings averaged 10.0 tons/acre (22.4 t/ha) and 11.8 tons/acre (26.5 t/ha), respectively, with duff depths of 2.3 and 1.0 inches (5.8 and 2.5 cm).

Postfire Plant Communities

Pitch Pine

Vegetation Dynamics—Pitch pine is well adapted to fire having thick bark, serotinous cones, and the ability to refoliate from dormant buds located along the stem and branches, or from the basal crook located just below ground line (Little 1979). A majority of the trees have serotinous cones especially where fires are severe (Little 1974). Seed is produced at an early age, 3 to 4 years for sprouts and 10 years for seedlings

Figure 4-18—Low pocosins maintained by periodic wildfire, eastern North Carolina. Photo by Walter Hough, 1973.

(Little 1953). Pitch pine is much more resistant to fire than its hardwood competitors. Fire every 4 to 6 decades will ensure a pitch pine component. As fire frequency increases, the importance of pitch pine in the stand also increases. Repeated intense fires at less than 20 year intervals will eliminate even its fire-adapted associates such as shortleaf pine, which take longer to produce viable seed (Little 1974). Shortleaf pine also produces basal sprouts when topkilled, but loses this ability with age while pitch pine can do so indefinitely (Little and Somes 1956). In the Pine Barrens, most associates are sprouters so that total cover surpasses 100 percent the first year after wildfire, severely restricting light and space for obligate seeders (Boerner 1981). The relationship between fire frequency, intensity, and severity, and their effect on postfire succession is discussed by Little (1979).

Common associates on upland sites in New Jersey include shortleaf, Virginia, Table Mountain and eastern white pines; black, white, northern red, southern red, chestnut, bear, post, scarlet, and blackjack oaks; and various hickories (Little 1973,1979; Little and Garrett 1990; Murphy and Nowacki 1997; Wright and Bailey 1982). Elsewhere eastern white pine is a common associate and outcompetes pitch pine in the absence of continued fire.

Successional trends following fire are toward domination by hardwood species. Without disturbance, pitch pine declines (Smith 1991; Vose and others 1994). On upland sites, trees usually invade much

Table 4-4—Average loading (tons/acre) of downed wood by diameter class (inch) and time since cutting for New Hampshire northern hardwood stands (Gore and Patterson 1986).

	Years since stand was clearcut				
Diameter	**1**	**15**	**50**	**100**	**Uncut**
0 to 0.25	3.4	1.1	1.5	0.5	0.9
0.25 to 1	6.5	2.2	2.4	2.7	2.4
1 to 3	17.8	1.7	2.9	1.0	2.1
>3	10.8	9.5	7.6	20.1	13.3
Total	38.5	14.5	14.3	24.3	18.7

Figure 4-19—Downed wood in old growth Northern hardwoods on the Bowl Research Natural Area, White Mountain National Forest.

faster than understory shrubs such as huckleberries and blueberries (Little 1979; Little and Moore 1949). On mesic to wet sites, fire exclusion leads to replacement by red maple, blackgum, sweetbay, American holly, and gray birch (Little 1979). Shrub encroachment is also much faster, and dense understories of sheep laurel, piedmont staggerbush, gallberry, and leatherleaf often quickly develop.

Management Considerations—On xeric mixed pine-hardwood ridges in the Southern Appalachians, fire has been advocated to restore diversity and productivity (Swift and others 1993; Vose and others 1994, 1997). Where pitch pine occurs with oaks and shortleaf pine in New Jersey, it is favored by winter fires that produce good seedbeds. In the Pine Barrens, a fire return interval of 12 to 16 years is used to maintain pine-oak stands. However, low-intensity winter prescribed burns have little effect on hardwoods (Little 1973). Boerner and others (1988) found that hardwood growth was actually increased by winter prescribed burns, suggesting that these fires were counterproductive. Competing hardwoods can be best controlled by applying a winter fire that reduces surface fuels followed by a summer fire. Deep-burning fires are needed on deep organic soils to prepare seedbeds and kill competing hardwoods; however, smoke management constraints generally severely curtail opportunities for such burns. Several underburns 8 to 16 years apart will reduce huckleberries while favoring an herbaceous ground cover including mosses and lichens (Buell and Cantlon 1953) that benefit many wildlife species from butterflies to quail. But according to Boerner (1981), species richness peaks the first year postfire and declines precipitously as resprouting heath cover closes toward the end of that first growing season.

In the middle of the Pine Barrens, intense, frequent fires (every 8 years or so) over a long period have eliminated virtually all large trees. The result is a low shrubby plant community consisting of a 3 to 7 foot (1 to 2 m) tall coppice growth of pitch pine, blackjack oak, bear oak, and mountain-laurel (Little 1946; Little and Somes 1964; Lutz 1934; Windisch and Good 1991). Repeated surface fires with moderate fire intensities can transform Pine Plains on xeric uplands into taller, less dense stands due to selective survival and growth of taller stems. Frequent crown fires (<20 year intervals) will maintain the Plains type. To preserve this unique ecosystem, fire frequency and severity should be varied to produce a diversity of stand ages, structures, and patch types (Windisch and Good 1991). Without fire, scrub type pitch pine eventually forms an overstory (Little 1998).

Roughly 4,000 acres of relic pitch pine barren ecosystem known as the Albany Pine Bush occurs in upstate New York. With periodic fire, pitch pine and scrub oaks (bear and dwarf chinkapin) predominate; but in the absence of fire, broadleaved hardwoods including red and white oaks, red maple, and white ash become established (Milne 1985). Prescribed fire is currently used on a 10 year return interval to restore and maintain this ecosystem, which provides habitat for the Federally endangered Karner blue butterfly. See Walker (1967) for discussion of management recommedations pertaining to pitch, Virginia, and pond pines.

Virginia Pine

Vegetation Dynamics—Virginia pine tends to dominate only on nutrient-poor, xeric sites (Mattoon 1915; Williston and Balmer 1980) where other species have trouble surviving. Virginia pine has only localized commercial importance. Although classed as a southern yellow pine and often exhibiting more than one flush during the growing season, it is much less tolerant of fire than the major Southern pine species because of its thin bark. Young trees sometimes produce basal sprouts when topkilled.

Management Considerations—Although pole-sized stands have been treated with low-intensity, winter season prescribed fire without overstory mortality in New Jersey, fire use should be considered experimental because of the likelihood of mortality. Most wildfires kill a majority of the stand because the thin, compact forest floor will only burn under relatively hazardous conditions. These fires, however, are usually responsible for regenerating the species. Fire is an effective tool for eliminating Virginia pine in mixed pine stands (Slocum and Miller 1953). Prescribed fire was recommended for preparing a seedbed for the next crop after harvest (Church 1955), and to increase seedling vigor (Sucoff 1961).

Pond Pine

Vegetation Dynamics—Pond pine has semi-serotinous cones, which are often produced by age 4 to 6 and open slowly over a period of years in the absence of fire (Wenger 1958). Seeds released from cones opened by fire almost invariably result in a blanket of reproduction, some of which survive the next fire if given 5 to 10 years to develop. Seedlings tend not to resprout, although they can in some situations.

Common associates are loblolly and slash pines, cabbage palmetto, Atlantic white-cedar, pond cypress, bald cypress, swamp tupelo, sweetbay, loblolly-bay, redbay, sweetgum, and red maple. Greenbrier is almost always a component of the understory along with switchcane, gallberry, large gallberry, swamp cyrilla, wax myrtle, saw palmetto, and sweetpepperbush (Bramlett 1990; Wenger 1958).

Different successional pathways producing various community types result from the interaction between fire frequency, fire intensity, hydrology, and organic soil depth (McKevlin 1996; Wharton 1977). The original pocosins once covered more than 2.5 million acres (1 million ha) in North Carolina alone (Richardson 1981), but only a fraction of that remains because of peat mining, drainage, and conversion to pine plantations or row crops. For overviews of this ecosystem see Richardson and Gibbons (1993) and Stout and Marion (1993).

Management Considerations—The prescription fire window is narrow between conditions too wet to carry fire and fires that sweep through the overstory completely consuming many live understory stems 0.25 to 0.5 inch (0.6 to 1.3 cm) diameter. Among several seedbed preparation techniques, prescribed fire was judged the most risky, but it also produced the best results (Crutchfield and Trew 1961). Where pond pine is not commercially utilized, stands are often burned every 10 to 20 years to regulate fuel buildup and restore fire to the ecosystem.

One of the most important pond pine understory communities is composed of switchcane. Cane also occurs as open-grown thickets thought to have originated on abandoned Native American agricultural fields and from Native American burning practices (Platt and Brantley 1997). Early explorers often mentioned canebrakes because of their distinctive character; apparently they were once widespread ranging from the valleys of the Appalachians to the Pocosins of the coastal Plain. They have largely disappeared because of overgrazing, inappropriate fire management, or deliberate type conversion. According to Wharton (1977), river cane burns about every 5 years but reaches its maximum fuel storage of 5 to 7 tons/acre (11 to 16 t/ha) in 3 years. Regularly burned cane provides some of the most nutritious native grazing in the South (Biswell and others 1942; Hilmon and Hughes 1965b; Hughes 1966; Shepherd and others 1951). If native range improvement is an objective, fires must be frequent; otherwise shrubs will overtop the cane within a decade. Continued exclusion results in one of the most flammable fuel complexes in the South. Guidelines for using prescribed fire in the Pine Barrens can be found in Little and Moore (1945).

Mixed Mesophytic Forest

Vegetation Dynamics—The hardwood forests of the Appalachian Mountains, the Ozark Mountains, and upland hardwoods of the Coastal Plain and the Piedmont have been regularly grazed and burned from the earliest settlement times (Komarek 1982; Van Lear and Waldrop 1989). The vast Shenandoah

Valley was burned annually by Native Americans to keep it from reverting to forest (Leyburn 1962). Foley (1901) noted that fire along with logging and grazing were major determinants of the species composition at the turn of the century. In the absence of fire, a mixed mesophytic forest develops. In the old-growth stage, pine regeneration is precluded and the forest slowly moves toward a hardwood climax (Cain and Shelton 1994). Literature reviews of the effects of fire on Eastern hardwood forests are provided by Christianson (1969) and Fennell and Hutnik (1970).

Management Considerations—The use of fire in hardwood stands generally has not been recommended because of the fear of damaging stem quality and because of the danger of erosion, particularly on steep slopes (Van Lear and Waldrop 1989). This recommendation is largely based on postburn observations of wildfires, which often burn with higher intensity and severity than prescribed fires. For example, in a survey of almost 6,000 harvested upland hardwood trees nearly half had basal wounds, 97 percent of them caused by fire (Hepting and Hedgcock 1937). The incidence of decay originating in basal wounds was greater for basswood and yellow poplar than oaks. Seventy percent of the trees with basal wounds had butt rot that resulted in an average cull of more than 15 percent. But the costs of decay are greater than just reduced board feet, because fire-damaged trees take up space that could be utilized by trees of superior form (Gustafson 1946).

Reviews of fire research on Southern Appalachian and Upper Piedmont sites showed that prescribed fires had little negative impact on soil (Van Lear and Johnson 1983; Van Lear and Waldrop 1989). Although numerous questions about fire effects on soils remain unanswered, generally fires that expose mineral soil create the potential for erosion, while those that leave a portion of the forest floor do not appear to have deleterious soil or water consequences. Van Lear and others (1985) found that Piedmont sites can be harvested following a series of low-intensity prescribed burns with minimal soil loss and degradation of water quality.

Augspurger and others (1987) and Waldrop and others (1985) demonstrated that single fires have little effect on the composition of young coppice stands. Roth and Hepting (1943) and Roth and Sleeth (1939) examined numerous hardwood stands of sprout origin and found that sprouts on burned areas were forced to develop at or below the ground line, which resulted in well-anchored stems free from decay. Thor and Nichols (1974) found that both the number of stems per sprout-clump and the total number of clumps, especially oaks, increased with annual and periodic burning in comparison to unburned stands.

Low-intensity prescribed fires have also been shown to stimulate germination of yellow poplar seed (Little 1967), which can remain viable in the forest floor for more than a decade, and produce more faster growing seedlings than those on unburned sites (Shearin and others 1972). Although fire has been demonstrated to be useful in the regeneration of some mixed mesophytic forests, no references were found that advocate underburning in the management of these forests. However, research on the application of prescribed fire in this type continues. For example, a moderately intense prescribed burn was applied to a south-facing slope in the Southern Appalachians to test its effectiveness for restoring a degraded pine/hardwood community and stimulating forage production after 70 years of fire exclusion (Elliott and others 1999).

Studies conducted in the Upper Piedmont and Southern Appalachians have shown that fire can be safely used to dispose of logging debris and prepare seedbeds (Swift and others 1993; Van Lear and Waldrop 1989). For example, Sanders and others (1987) found that low-intensity dormant-season fires had little adverse effect on bole quality of mature hardwood stems. Van Lear and Danielovich (1988) noted little visible evidence of erosion on mountain slopes up to 45 percent following prescribed summer burning designed to reduce heavy logging debris and prepare the site for planting. Sanders and Van Lear (1987) showed that the judicious use of fire reduces the large amount of highly flammable fine woody material present after clearcutting by more than 90 percent. The fell-and-burn technique that gained prominence in the late 1980s can regenerate mixed pine-hardwood stands after clearcutting with minimal adverse site effects (Abercrombie and Sims 1986; Danielovich and others 1987; Phillips and Abercrombie 1987).

Northern Hardwoods

Pre-1900 Succession—Paleoecological studies suggest that Northern hardwood species such as beech, sugar maple, and birch decline following fire. Pollen and charcoal samples from Lake Wood in Acadia National Park, Maine, show that during the period about 2,000 to 6,000 BP (before present time), Northern hardwoods and hemlock were dominant. During that period, fires indicated by charcoal analysis occurred in conjunction with sharp declines in hemlock probably as a result of an insect or disease outbreak (Davis 1981). Declines of hemlock about 4,800 and 3,000 BP were followed by periods in which one or more fires burned the watershed. With the rise in importance of spruce and cedar about 2,000 BP, the incidence of fire increased with return intervals of 200 to 400 years. The abundance of maple, beech, and hemlock declined simultaneously. The watershed of Lake Wood burned in a catastrophic fire in 1947; today it contains only one small stand (about 5 acres) of hemlock and no Northern hardwood stands. The forest is dominated by seral hardwoods (aspen, paper birch, and gray birch), northern red oak, white pine, and red pine.

Northern hardwoods such as the Bigwoods of Minnesota generally are not very flammable (Grimm 1984); fires burn as patchy, creeping ground fires. Grimm (1984) noted that "the fire regimes of deciduous forests, such as the Bigwoods, are much different from the commonly perceived model of a forest fire regime, in which fuels and fire danger increase with time and in which intense crown fires commonly cause great destruction of forest." This is consistent with our observations in Northern hardwood stands in Maine other than at Lake Wood (table 4-5). Stands burned in 1947 currently support forests dominated by sprouts of beech and sugar maple rather than seral hardwoods (Patterson and others 1983).

A vigorous debate exists about whether Native American cutting and burning practices or climatic cooling caused shifts from Northern hardwoods to oak and pine at Crawford Lake, Ontario (Campbell and McAndrews 1995; Clark 1995; Clark and Royal 1995; McAndrews and Boyko-Diakonow 1989). Although the relative importance of Native American burning versus climate change as an influence on the larger Northern hardwood region remains open, it seems likely that changes evident in the Crawford Lake pollen profiles were partly the product of human manipulation of the forest. Clark (1995) concluded that additional studies of Native American effects on Northern hardwood forest composition are needed, but there is little evidence that Native American burning alone (without accompanying agricultural activity) was as important in Northern hardwoods as it apparently was in oak forests to the south (Abrams 1992).

Post-1900 Succession—Based on fire records from 1945 to 1976, Bormann and Likens (1979) concluded that forests in the Green and White Mountains are "among the least burnable in the 'northern hardwood region'." On average, only 7 to 10 acres (3 to 4 ha) burn annually per million acres (405,000 ha) on the Green Mountain and White Mountain National Forests. Fahey and Reiners (1981) calculated fire rotations in Northern hardwoods of 910 years for Maine and 770 years for New Hampshire. Current work (Patterson 1999) documents the continued trend toward less area burned (longer rotations) during the later half of the 20th century in New Hampshire and a low 20th century fire occurrence in Vermont, which has the largest representation of Northern hardwoods. Stearns (1949), who examined a virgin Northern hardwood stand in northern Wisconsin, noted that although hot slash fires "burned to the edge of the virgin stand they did not penetrate into it more than a few rods."

Although ecologists believe fire has been a relatively unimportant ecological factor in Northern hardwoods (Bormann and Likens 1979; Fahey and Reiners 1981), they acknowledge the fact that Northern hardwoods have burned in the past, especially when adjacent stands were clearcut during the logging period and when stands accumulated fuels from blowdown (Lorimer 1977; Stearns 1949). Records suggest that modern stands have been more influenced by fire (chiefly as a result of anthropogenic fire during the period 1850 to 1950) than stands will be in the future. Even present Northern hardwood stands have been influenced to a far smaller degree by fire than have other vegetation types in the Northeast.

After the 1947 fire in Acadia National Park, beech and sugar maple stands have returned to their original stand composition more rapidly than any other

Table 4-5—Average basal area (sq ft/acre) by species for Mt. Desert Island northern hardwood stands burned in 1947 and before 1880 (Patterson 1999).

	Burned in 1947		Burned before 1880	
	Sample year			
Species	1980	1992	1980	1992
Red spruce	—	0.1	0.6	1.6
Hemlock	0.1	0.1	0.4	—
Paper birch	4.8	4.3	4.0	4.2
Yellow birch	0.1	0.3	0.1	0.7
Red maple	0.3	1.2	0.4	0.9
Sugar maple	4.4	3.1	7.0	5.0
White ash	0.1	—	1.5	1.1
American beech	11.7	15.0	9.6	9.2
Striped maple	0.5	1.6	0.9	2.2
Bigtooth maple	0.7	1.5	0.2	0.4
Others	0.5	0.2	0.3	0.2
Total	23.2	27.4	25.0	25.5

forest types (table 4-5) (Patterson 1999; Patterson and others 1983). Although Northern hardwood species are widely viewed as having little resistance to fire, maple and birch sprout vigorously from the stump; beech suckers from the root system as vigorously as aspen (Fowells 1965). This capacity for rapid vegetative reproduction appears to limit invasion of Northern hardwoods by seral aspen, paper birch, and gray birch. These species are short lived and cannot persist in competition with beech, maple, and yellow birch in the absence of frequent, stand-replacing disturbances (Patterson and others 1983). The present dominance of white birch on some sites in the White Mountains is probably more a reflection of increased incidence of fire and logging in the 1800s than it is an indicator of the long-term importance of fire on the landscape.

Management Considerations—As management shifts toward longer harvest rotations and reduced volume removal, Northern hardwoods will likely regain their historic position of importance on mesic, fire-protected sites in the Northeast. Northern hardwoods are susceptible to fire (Swan 1970). Where Northern hardwoods mix with conifers including hemlock, white pine, red spruce, and balsam fir, fires are likely to be more common, especially in the wake of catastrophic wind storms (Foster 1988; Lorimer 1977; Stearns 1949). If climate warms and incidence of fire is reduced, Northern hardwoods may return to some sites at the present hardwood-boreal forest boundary while giving way to transition hardwood-conifers to the south. However, the increased presence of human ignition sources may alter fire-vegetation relationships evident in presettlement forests.

Bottomland Hardwoods

Vegetation Dynamics—In young pole size stands, fires often result in basal wounds. Although these wounds often heal over, internal decay continues with decay height closely related to time since fire. Kaufert (1933) estimated that 90 to 95 percent of the decay in merchantable Southern bottomland hardwood stands was the result of past fires. When mature oaks die in areas protected from fire, species such as red maple, American elm, and green ash tend to replace them (Abrams 1992). Lotan and others (1981) stated that because the elm-ash forest is moderately fire-prone, prescribed fire should be tested for its ability to control insect and disease pests and unwanted understory. Keep in mind that most bottomland hardwoods, even large ones, are sensitive to fire. Low-intensity fires appear benign at first glance, but the cambium has been damaged and incipient decay begins even though the bark remains intact for several years after fire.

Stand-Replacement Fire Regimes

Major Vegetation Types

Vegetation types in the Eastern United States represented by stand-replacement fire regimes, where fire typically kills more than 80 percent of the overstory, include prairie, wet grassland, and portions of the oak-gum-cypress (bay forests) FRES ecosystem types (table 4-1). The conifer cover types include sand pine, Table Mountain pine, Atlantic white-cedar, and spruce-fir. Pocosins without a significant component of pond pine are also a stand-replacement fire regime type.

Wet Grasslands

Kuchler (1964) recognized two major regions of herbaceous wetlands in the Eastern United States, exclusive of the Everglades of southern Florida (see chapter 7). These regions include the northern cordgrass prairie, which extends along the Atlantic coast from Maine to southern Florida, and the southern cordgrass prairie, which spans the Gulf of Mexico from southern Florida to southern Texas. Numerous marshes, some quite extensive, occur in inland areas in the Eastern United States, and many of these possess characteristics similar to freshwater coastal marshes.

It is convenient to distinguish salt and brackish from oligohaline (tolerant of moderate salinities) and fresh marshes because of consistent differences in species composition and fire behavior. Along the Atlantic seaboard, salt marshes can be further subdivided into the New England group and the Coastal Plain group (Mitsch and Gosselink 1993). Salt marshes of the New England group extend from Maine to New Jersey and are built mainly on marine sediments and marsh peat, with relatively little sediment discharge from distributaries. Salt marshes of the Coastal Plain group extend southward from New Jersey to Florida, where they are replaced by Mangrove forests at the southern tip of Florida.

Salt and brackish marshes are largely dominated by species of cordgrass and rush. The regularly flooded, tidal salt marshes in the Eastern United States are dominated almost entirely by smooth cordgrass. Regularly flooded areas typically are referred to as low salt marshes to distinguish them from high salt marshes, which occur inland from the low salt marshes, are less frequently flooded, and often contain more stressful soil conditions due to stagnation and evaporative concentration of salts (Mitsch and Gosselink 1993). Low salt marshes reach their greatest extent in South Carolina, Georgia, and along the Gulf of Mexico (Teal

1986). High salt marshes can be dominated by several species such as smooth cordgrass, needlegrass rush, pickleweed species, inland saltgrass, saltmeadow cordgrass, and saltmeadow rush. Along the Texas coast, high salt marsh can include extensive stands of gulf cordgrass, which is also the characteristic dominant of salty prairie, an upland community type. In brackish areas, salt marsh species yield dominance to species of slightly less salt tolerance, and a greater variety of both dominant and subordinate species can be found (Gosselink 1984). In addition to a shift in the herbaceous layer, brackish marshes often include woody species, especially eastern baccharis and bigleaf sumpweed. A more detailed discussion of the geographic variations in salt marsh geomorphology and vegetation can be found in Mitsch and Gosselink (1993).

Coastal marshes that receive freshwater and are removed from the direct influence of salt water form the inner band of coastal marshes. These marshes reach their greatest extent along the middle and southern Atlantic Coast and along the northern Gulf Coast. The Atlantic Coast freshwater marshes include about 405,200 acres (164,000 ha), while those in the northern Gulf of Mexico cover about 1,156,400 acres (468,000 ha). Generally, fresh and oligohaline wetlands occur where salinities are less than 5 ppt, but wetland types are more easily recognized by the known salinity tolerances of the vegetation rather than the average soil or water salinity (Brewer and Grace 1990). Many plant associations exist because of the high diversity of species found in fresh and oligohaline marshes. The lowest fresh marshes are characterized by plants that root in relatively deep water, such as species of pond-lily, waterlily, wildrice, and cutgrass. Along the Atlantic Coast, tidal fresh marshes include both annual streamside associations and perennial associations of green arrow arum, pickerelweed, arrowhead species, and cattail species. In fresh marshes of the Gulf Coast, bulltongue arrowhead, maidencane, spikerush species, and numerous sedge and forb species are predominant.

Prairie

The prairie ecosystem in the United States, referred to by many as the tallgrass prairie, forms a rough triangle from the Minnesota and North Dakota border south to the Texas Gulf Coast and eastward into northern Indiana (Reichman 1987). It is dominated by big bluestem, Indiangrass, and switchgrass. Drier sites are dominated by little bluestem, and wet, lowland sites are often dominated by prairie cordgrass. Within this grass matrix are more than 250 forb species (Freeman 1998). The local species composition at any one place is dependent on burn history, grazing history, soils, aspect, and topographic position. In Indiana, Illinois, and Wisconsin only 0.1 percent of this ecosystem remains. The Flint Hills region of Kansas has the most remaining prairie, 3 million acres (1.2 million ha), which is only 17 percent of the presettlement prairie in Kansas (Samson and Knopf 1994). The prairies that do remain outside of the Flint Hills region are widely scattered and often smaller than 1.2 acres (0.5 ha) (Betz and Lamp 1989). The tallgrass prairie is one of the youngest ecosystems in North America. Much of the region was glaciated only 10,000 years ago. This region has few endemic species, so most plant and animal species have migrated into this region from neighboring ecosystems (Risser and others 1981).

Portions of the tallgrass prairie consist of oak savannas and glades or barrens. Herbaceous species found in savannas are a mixture of forest and prairie species (Curtis 1959); there are no endemic savanna species. In wetter periods, or periods with reduced fire frequency, savannas can be converted to forests. In drier years or with shortened fire intervals, savannas can be converted to grasslands. Whether savannas are stable ecosystems or simply an unstable continuum between closed canopy forest and open grasslands is debatable (Nuzzo 1986). Today, Nuzzo (1986) estimates that only 0.02 percent of presettlement savanna survives and that the largest savannas are only about 50 acres (20 ha).

Glades or barrens are patches of prairie within a forest matrix. The barrens region stretches across Missouri, Tennessee, Kentucky, southern Illinois, Indiana, and Ohio (Baskin and Baskin 1978). These grasslands occur only on very dry, shallow soils, which are usually found on south- or west-facing slopes. Anderson and Schwegman (1971) stated that barrens are "degraded forests that had been invaded by prairie plants as a result of fire" and that in the absence of fire the areas quickly revert to forest. However, Wade and Menges (1987) stated that these glades don't support woody vegetation except for the shallow rooted eastern redcedar; thus, they are often termed cedar glades (Baskin and Baskin 1978). If fire is excluded for long periods, trees encroach around the edges of glades and reduce their size.

Bay Forests

This ecosystem type occurs primarily in North and South Carolina but can be found along the Atlantic Coast from Virginia to Alabama. Carolina bays and pocosins without an overstory of pine or Atlantic white-cedar are the major vegetation types. Much of the original type contained a merchantable overstory that was harvested, thereby altering the fire cycle. This type is characterized by a dense tangle of evergreen and deciduous shrubs and vines (Richardson and Gibbons 1993). Carolina bays are swamps dominated by bay species. Numerous species of special concern including several Federal and State listed species occur in this vegetation type.

Conifers

Sand Pine—Two varieties of sand pine, Choctawhatchee and Ocala, are recognized here because their fire ecology requires different management. The natural range of Choctawhatchee sand pine is confined to the panhandle of Florida and Baldwin County, Alabama (Brendemuehl 1990). It originally was restricted to the Gulf shoreline (islands and dunes) where fire was infrequent (Outcalt 1997). The Ocala variety has serotinous cones and is confined to the central Florida ridge and on old sand dunes down both coasts from central to southern Florida. It is easily distinguished from the Choctawhatchee variety, which generally lacks serotinous cones.

Table Mountain Pine—This serotinous cone species is endemic to the Appalachians from Pennsylvania to northeastern Georgia, with local populations in New Jersey and Delaware (Little 1978). According to Sternitzke and Nelson (1970), about 90 percent of the standing inventory is in West Virginia, Virginia, and North Carolina. Table mountain pine forms even-aged pure stands or shares dominance with pitch pine. Common hardwood associates include red maple, blackgum, sourwood, chestnut oak, and scarlet oak (Della-Bianca 1990), which eventually dominate these sites in the absence of fire. A dense shrub layer of mountain-laurel, which will burn within a week of good drying conditions during the dormant season, is often present along with other ericaceous shrubs such as blueberries and huckleberries.

Spruce-Fir—This is a high-elevation forest type of the Appalachians and is one of the rarest and most threatened ecosystems in the South (White and others 1993). In the Southern and Central Appalachians, stands comprise red spruce, the endemic Fraser fir, and balsam fir. In the Northeast, red and white spruces and balsam fir make up this forest type.

Atlantic White-cedar—This species tends to form pure stands throughout a narrow coastal belt from southern Maine to northern Florida (Little 1959) and westward to the Mississippi (Little 1978). However, its distribution is spotty because it avoids substrates underlain with clay (Little 1950). It requires moist sites with a long hydroperiod but without stagnant water, generally in swamps with organic soils.

Fire Regime Characteristics

Wet Grasslands—Much of the coastal region of the Southern United States, from Virginia to Texas, is characterized by a presettlement fire frequency of 1 to 3 years (Frost 1995). Coastal marsh landscapes are typically extensive, a factor that aids in the propagation of an individual fire. Natural barriers to fire spread are relatively common and vary from wide river channels to small stream channels and narrow animal trails. Depending on the fuel and wind speeds, fires may either bridge small to moderate-sized natural breaks or be stopped by them. Thus, the extent of natural fires varies greatly as does the ease of accomplishing a prescribed burn. Lightning-strike fires are thought to be common in coastal wetlands (Frost 1995), and often fire from the adjacent upland can spread into the marsh. Spontaneous combustion has been reported to occur in coastal marshes (Viosca 1931), though how frequently it happens is largely unknown.

Away from the coastal influence of a persistently high water table, peat fires can be common during prolonged dry periods and can represent a substantially more severe fire, leading to loss of substrate and protracted subterranean burning. While such fires may not be the norm for coastal marshes, they can be important under some circumstances (Hungerford and others 1995). For all fires, coastal or inland, groundwater levels are important for both the behavior of the fire and its effects on vegetation and soil (Bacchus 1995).

Prairie—Historically, Native Americans contributed to the creation and maintenance of the tallgrass prairie ecosystem by frequently burning these ecosystems, which controlled woody vegetation and maintained dominance by herbaceous plants. In the Eastern tallgrass prairie, Native Americans were probably a far more important source of ignition than lightning. With grasses remaining green through late summer and a low incidence of dry lightning storms, lightning-caused fires were probably relatively infrequent.

Few studies of the pre-Euro-American tallgrass prairie have been conducted. Most existing data are primarily anecdotal, based on widely scattered accounts of early explorers. Existing data on burn frequency in the tallgrass prairie come from studying tree rings on the prairie-forest margin. In Missouri, Guyette and McGinnes (1982) determined that fires occurred every 3.2 years prior to 1870. After 1870, the average fire return-interval increased to 22 years. On the Mark Twain National Forest, Guyette and Cutter (1991) determined that from 1710 to 1810, fires occurred on average every 4.3 years and that severe fires (fire scars on three or more trees) occurred every 11 years. After 1810 fires occurred every 6.4 years, and no fire scarred more than two trees. But fire scar studies are typically conservative and tend to underestimate fire frequency, which could be the case here because historical accounts reported annual burning in the tallgrass prairie

(Pyne 1997). After Euro-American settlement, fires were less frequent and burned smaller areas. The reduction in fire frequency resulted in replacement of large areas of grasslands by woodlands (Beilmann and Brenner 1951; Muir 1965; Pyne 1997).

Some notion of the seasonality of historic fires can be learned from early explorers and missionaries dating back to the late 1600s. According to Pyne (1997), Wells stated in 1819 that fires in western Pennsylvania were set by Native Americans at the end of the Indian summer, presumably October; Michaux observed in 1805 that fires occurred in March or April in Kentucky; and James reported in 1819 that fires in Missouri usually occurred in the fall. In Illinois, McClain and Elzinga (1994) reported that historically almost all fires were ignited during the Indian summer, late October to early November. Euro-Americans continued the burning practices they learned from Native Americans until early in the 20th century when these historic fire patterns were altered through fire suppression, planting of cool season grasses, and the use of prescribed burning concentrated in the spring to improve forage for livestock. It appears from these historical accounts that the season when fires were common varied regionally and through time as a result of the cultural practices of the people living in a particular area.

In addition to fire, bison and other large grazers influenced the tallgrass prairie. Bison were present throughout the tallgrass region (McClain and Elzinga 1994); thus periodic grazing was a significant influence on plant communities (fig. 4-20). Bison prefer burned to unburned grassland for grazing during the growing season and can contribute to the pattern of burning in prairie (Vinton and others 1993). The variability of grazing by bison may have created variability in the patchiness of fuels and severity of subsequent fires. The effects of grazers in tallgrass prairie plant communities are reviewed by Hartnett and others (1996), Hartnett and Fay (1998), Howe (1999), and Knapp and others (1999).

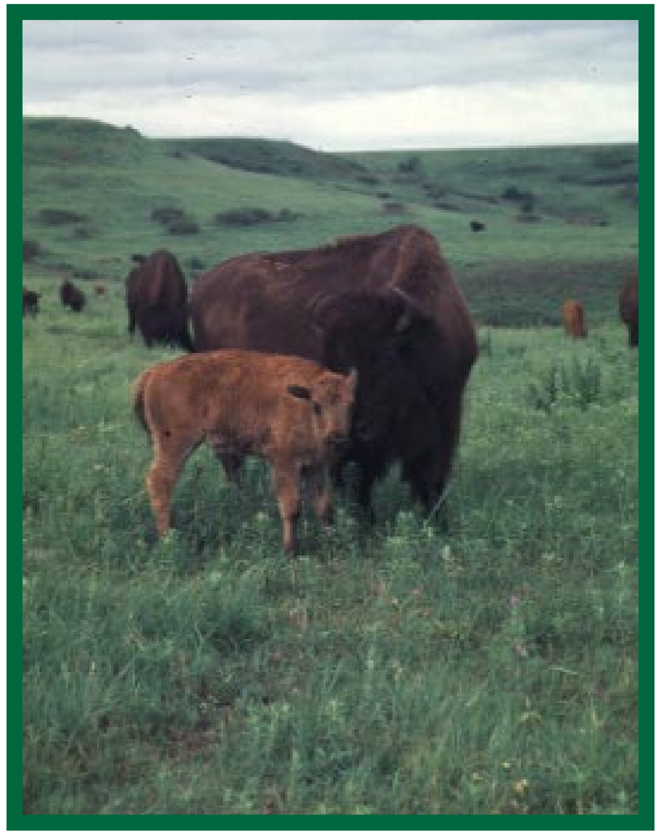

Figure 4-20—Grazers are attracted to freshly burned areas because of increased forage quality, Konza Prairie Biological Station, Riley County Kansas. Fire management can control the distribution of grazers on the landscape. Photo by Greg Hoch.

Bay Forests—This type now burns on about a 20 to 100 year cycle, but uncertainty exists about the historic fire frequency. Wharton (1977) stated that 50 to 150 years are required for mature bay forests to develop. McKevlin (1996) believes fire frequency is probably less now than it was 200 years ago in spite of lowered water tables. The presence of charcoal lenses at various depths in the underlying organic soil is evidence of extensive fires in the past (Dolman and Buol 1967).

Conifers—The fire cycle for Ocala **sand pine** corresponds roughly to stand longevity, which is 30 to 60 years (Christensen 1981). The historic fire frequency for the Choctawhatchee variety is unknown but lightning fires were rare. This variety grows in pure stands, directly inland from the beach, separated from more pyrophilic vegetation types by wet intradune swales and sparse dune vegetation.

Little is known about the historical fire frequency in **Table Mountain pine**. Barden and Woods (1974) found that fires in Table Mountain pine were more frequent, more intense, and probably larger earlier this century. Between about 1800 and 1944, the fire interval in sampled stands averaged 10 to 12 years. Now the fire return interval is between 7 and 70 years with an average of 40 years based on USDA Forest Service records. A commonly accepted hypothesis is that the fire practices of the large Native American population inhabiting the Southern Appalachians exposed both Table Mountain and pitch pine to frequent understory burns keeping the stands open, thereby ensuring successful regeneration after the occasional stand-replacement fire. Zobel (1969) suggested an evolutionary link between fire and Table Mountain pine because of its fire adaptations.

Fires rarely occurred in **spruce-fir** forests prior to Euro-American settlement. Two relatively recent instances in Maine were the 4 days in October 1947 when about 7,000 acres (2,830 ha), including spruce-fir, burned, and the summer drought fire that involved Mount Katahdin in the early 1990s. Before Euro-Americans harvested **Atlantic white-cedar**, this prized species was mostly perpetuated by major disturbances, probably crown fire that occurred at 25 to 300 year intervals. Mature cones can develop by age three in open stands, which supports the crown fire hypothesis.

Fuels and Fire Behavior

Wet Grasslands—Coastal marshes typically support high fine fuel loadings. Estimates of standing biomass range from 4 to 18 tons/acre (10 to 40 t/ha) (Gosselink 1984; Odum and others 1984) and higher (Gough and others 1994). Hackney and de la Cruz (1978) have reported that for some species, full recovery of standing biomass to preburn levels can occur in a single growing season, while it may take several years for other species. Informal observations of fire behavior indicate that marsh fires are generally consistent with other grassland fires in the Eastern United States. Rates of lateral spread vary with wind speeds and can exceed 3 feet/second (1 m/s). Flame heights can vary from less than 3.3 feet (1 m) for short or sparse vegetation and up to 13 feet (4 m) for dense or tall vegetation (fig. 4-21). In typical areas of fine fuels, marsh fires produce fewer flying embers than shrubland or forest fires and are generally less likely to produce spot fires.

Coastal marshes typically contain large quantities of herbaceous vegetation and are considered highly flammable. But not all marsh types and plant species are equally conducive to propagating a fire. The vegetation of salt and brackish marshes is relatively similar throughout the coastal region of the Eastern United States, with species of cordgrass, rush, and saltgrass as typical dominants. The cordgrass species, which are generally the most widely represented group throughout coastal salt and brackish marshes, tend to be quite flammable. Species such as saltmeadow cordgrass and gulf cordgrass can burn readily while tissues are green and are capable of burning more than once in a single growing season. All of the cordgrass communities are capable of carrying fire over standing water.

Flammability of vegetation throughout fresh and oligohaline marshes can be highly variable due to the considerable plant diversity (Gough and others 1994). In general, grasses support much more intense and continuous fires than do forb and sedge associations. Some exceptions to this include dense stands of sawgrass and cattail, which are capable of supporting extensive fires. Common species that generally provide good fuels

Figure 4-21—Headfires in sawgrass behave similarly in other Eastern wet grasslands, Everglades National Park. Photo by Wayne Adkins.

include reed, maidencane, and switchgrass. Dominant species that represent poor fuels include bulltongue arrowhead, spikerush species, alligatorweed, hydrocotyle species, pickerelweed, and green arrow arum. Generally, grass-dominated associations burn more readily than those dominated by sedges or forbs. Among grass-dominated associations, fresh and oligohaline marshes (for example, panicums) tend not to burn as reliably as those in the brackish and salt marshes (Ford and Grace 1998a). Forb associations, such as the extensive bulltongue arrowhead community of the lower Mississippi River wetlands, typically provide poor fuel for fires during the dormant season when plant tissues have decomposed and fuel is sparse. In other forb-dominated plant associations, propagation of a fire may only be successful during the dormant season when aboveground tissues have senesced.

The presence of woody plants in coastal marshes can alter fire behavior substantially in some cases. Where the woody plant component is high, reduced airflow to fine fuels may reduce the completeness of the burn and the rate of fire spread. Nonetheless, fire generally kills the aboveground portions of these woody species (Scifres and Hamilton 1993). Succession to woody associations in fresh marshes is common, particularly where water depths are not excessive and substrates are firm. A heavy dominance by species such as willow, maple, cypress, tupelo, cedar, Chinese tallow, and bayberry can substantially reduce herbaceous fuels. These communities are likely to burn only under very dry conditions.

Prairie—Early explorers and settlers described abundant grasslands, prairies, savannas, and open woodlands with grass and herbaceous understories throughout the Eastern United States. The exact characteristics of tallgrass prairie such as height, loading, and percent cover are unknown but can be estimated from historical accounts. Grasses were 3 to 6 feet (1 to 2 m) high; they were often described as tall as a man on horseback. Percent cover was substantial (>50 percent) over hundreds of square miles. From these characteristics, loadings are conservatively estimated to range from 2 to 5 tons/acre (4.5 to 11.0 t/ha).

Fuel loadings ranged from 1.1 tons/acre (2.5 t/ha) for sparse communities to 3.4 tons/acre (7.6 t/ha) for abundant communities in fuel models developed for fire behavior modeling (Reinhardt and others 1997). These loadings were based on estimates of current production (Garrison and others 1977) and accumulated thatch or litter. Flame lengths of 12 feet (3.6 m) could be expected assuming these loadings, 6 percent fuel moisture content, and windspeed of 20 mph (32 km/hr) (Rothermel 1983). Thus, historical prairie fires probably burned much of the time with flame lengths of 8 to 15 feet (2.4 to 3.6 m), too hot for direct frontal attack with hand tools.

Bay Forests—Forest floor and downed woody fuels averaged 10 tons/acre (22 t/ha) in the Piedmont of North Carolina (Albrecht and Mattson 1977). In the Great Dismal Swamp in Virginia, Day (1979) found loadings of accumulated dead and live understory fuels (< 1 inch diameter) in a mixed hardwood stand of 9.7 tons/acre (21.7 t/ha) and on a maple-gum site of 15.6 tons/acre (35.1 t/ha). Also, see pond pine fuels and fire behavior in the Mixed Fire Regime section.

Conifers—Sand pine needles are short and form a flat mat on the forest floor that burns poorly and supports creeping fires. Needle-drape is not a flammability problem in either variety. Mature stands average about 10 to 15 tons/acre (22 to 34 t/ha) of available fuel composed mainly of live understory biomass (Custer and Thorsen 1996). Lightning fires are common, typically small, low-intensity creeping fires that go out at night when the humidity rises. The primary ground fuel is deer moss, which can produce flame lengths of 2 to 3 feet (<1 m) when dry, but when humidity is high, the moss absorbs moisture from the air and will not even smolder. The Ocala variety often supports a well-developed understory of scrub oaks, ericaceous shrubs, and rosemary that will burn intensely when dry and pushed by a strong wind. This variety also exhibits a "varnish stage" when the needles unexplainably exude a sap-like substance that is exceedingly flammable (Hough 1973). This condition is usually observed in the fall and occurrences can be decades apart. The Ocala fuel complex typically burns with intense fires that almost always enter the stand from more pyrogenic adjacent communities (fig. 4-22). The fastest spreading wildfire recorded in the United States occurred in this fuel type; it covered 35,000 acres (14,170 ha) in 4 hours with a spread rate of 6 mph.

Under **Atlantic white-cedar** stands in the Great Dismal Swamp of Virginia, Day (1979) found average loadings of 1.8 tons/acre (4.0 t/ha) for the litter layer, 1.2 tons/acre (2.7 t/ha) for woody fuels <0.75 inch (2 cm) diameter, and 22.4 tons/acre (50 t/ha) for woody fuels >0.75 inch (which included stumps as well as logs).

Postfire Plant Communities

Salt and Brackish Marshes

Pre- and Post-1900 Succession—Many coastal marshes are dependent on fires that are lethal to aboveground tissues and that reduce or eliminate woody plants. Woody plants can, however, be excluded from an area by excess salinity. Saline and hypersaline soil conditions generally preclude most native woody species in both coastal and inland wetlands throughout the temperate zone, with the exception of salt-tolerant mangroves, which are restricted to subtropical-tropical latitudes in southern Florida (Mitsch

Figure 4-22—Stand replacement burn in Ocala variety sand pine on the Ocala National Forest, Florida. Photo by George Custer, 1993.

and Gosselink 1993). Marshes formed on unconsolidated or flotant substrates are also typically unsuitable for the long-term success of trees even in the absence of fire (Doyle 1995). In some situations, woody succession may be precluded by herbivores. For example, in much of southern Louisiana wild populations of nutria are believed to prevent successful reestablishment of baldcypress (Connor and Toliver 1987). The herbaceous communities that develop in marshes, as a result of fire, edaphic influences, or herbivores, tend to be readily flammable and are well adapted to frequent fires.

Despite the frequent presence of standing water, fire is able to propagate in both low and high salt marsh and in brackish vegetation. As a result, these systems are frequently burned, both naturally and with prescription. Postfire succession patterns vary somewhat with salinity and preexisting vegetation. In low salt marshes, little species replacement occurs and smooth cordgrass typically retains dominance. Thus, the main effect on vegetation is a replacement of old tissues with younger tissues that are more palatable to wildlife species. In high salt marshes, fire causes few long-term shifts in vegetation, although it may provide a brief period when dominance by cordgrass is reduced and interstitial species increase. Successional patterns are more pronounced in brackish marshes following fire. In areas of the central and western Gulf coast, succession from herbaceous dominance to dominance by eastern baccharis takes place over several years. Frequent fires in these systems keep woody species from dominating. On a shorter time span, fire in brackish marshes reduces dominance by cordgrass and rush species temporarily and favors earlier successional species such as chairmaker's bullrush (Ford and Grace 1998a).

Management Considerations—According to Nyman and Chabreck (1995), the frequency of intentional fires increased around 1910 as burning in coastal marshes became a more common practice for promoting wildlife populations and reducing the hazards of wildfires. Today, prescribed fire is commonly and frequently used in salt and brackish marshes to enhance productivity (Hackney and de la Cruz 1981), manage food sources for wildlife and cattle, reduce plant cover, reduce fuel loadings, and eliminate woody species such as baccharis (Chabreck 1988).

In the absence of fire, succession to woody dominance can take place in only a few years. Where the control of woody plant succession is of highest priority, growing-season burns approximately every 3 years are the most effective. Observations indicate that baccharis-dominated systems remain flammable, in contrast to those overtaken by the introduced exotic, Chinese tallow. This species can greatly reduce fire intensity and fire propagation in dense stands (Grace 1998). Only frequent fires are likely to effectively prevent its invasion into wetlands. Once Chinese tallow exceeds a certain density, it typically becomes nonflammable and acts as a firebreak. Once this density threshold is reached, herbicides or mechanical means will be required for its removal.

Waterfowl and mammals alike generally prefer early successional plant species as well as the younger tissues of resprouting plants. Some fire-promoted species such as chairmaker's bullrush are considered of exceptionally high value to muskrat (*Ondatra zibethicus*), nutria (*Myocaster coypus*), and snow geese (*Chen caerulescens*). In areas where burning is commonly used to promote wildlife populations, burns may be conducted in the fall and winter to provide a steady supply of young tissues throughout the winter. For cattle grazing, burning is often recommended as a method of temporarily increasing dietary crude protein and forage quality (Angell and others 1986; McAtee and others 1979).

Marsh fires can be classified as cover burns, root burns, and peat burns (Lynch 1941; O'Neil 1949). Water levels control the depth of influence of fire. Thus, proximity to the water table, tidal conditions, and drought cycles can determine the severity of impact to belowground plant parts and to substrate. While more common in inland areas such as the Everglades, peat burns have been reported in coastal wetlands along the Gulf of Mexico (Hoffpauir 1968; Lynch 1941). During dry periods it may be possible to create fires that could substantially damage plant roots. Documented success in using such burns to control plant species, however, is lacking (Nyman and Chabreck 1995).

Prescribed fire for promoting desired wildlife forage species such as chairmaker's bullrush appears to be more successful when conducted during the fall or winter. Spring burns are believed to damage the regrowth of this species and lead to more persistent dominance by saltmeadow cordgrass (Chabreck 1981). Fall and winter burns may be used to avoid destroying nests or killing young wildlife (Nyman and Chabreck 1995). Fires aimed at promoting nutria and many other species are often limited in size to produce a landscape mosaic of burned and unburned habitat (Kinler and others 1987). Postburn water levels can substantially influence the effects of fire on vegetation. When stubble is submersed for an extended period, complete death can occur for many wetland species (Herndon and others 1991; Sale and Wetzel 1983). See Chabreck (1988) for more information on the use of fire to manage wildlife, and Kirby and others (1988) for an extensive bibliography of literature dealing with fire effects on wildlife.

Avoiding use of fire to favor wildlife may be wise under certain circumstances. For example, in the Mississippi delta, geologic subsidence rates have contributed to extremely high rates of marsh loss because coastal areas have subsided faster than accretion occurs. In this situation, the feeding activities of nutria and other mammals can contribute to habitat loss (Ford and Grace 1998b). Nyman and Chabreck (1995) noted that the potential exists for deleterious effects of marsh burning in this region. Hypothetically, fire could accelerate rates of wetland loss through the removal of organic matter that might otherwise contribute to sediment accretion and through the promotion of wildlife populations that lead to consumption of vegetation. At present, experimental evaluation of this hypothesis is lacking.

Another factor is the possibly deleterious grazing by snow geese. Currently, excessive numbers of snow geese are causing extensive damage to the northern wetlands where they breed and they are also known to cause extensive eat-outs of southern coastal wetlands. There is no evidence at present that burning is significantly affecting their population. However, burning of coastal wetlands can attract wintering snow geese to recently burned areas and increase the potential for localized damage. Finally, because of the important role of marshes as sources of organic matter for estuarine food webs, high fire frequencies are not necessarily ideal for near-shore systems (Hackney and de la Cruz 1978; Nyman and Chabreck 1995). A more complete discussion of salt marsh fire ecology can be found in Lynch (1941), Bendell (1974), Daiber (1974), Frost (1995), Nyman and Chabreck (1995).

Fresh and Oligohaline Marshes

Pre- and Post-1900 Succession—Various successional sequences are found from fresh marshes to forested wetlands for the most inland of the fresh marshes. Fresh and oligohaline wetlands may succeed to dominance by baldcypress, swamp tupelo, water tupelo, and red maple as well as to Chinese tallow (Frost 1995). A number of factors may impede woody plant development including unconsolidated substrate, scouring by waves, and periodic fires. Fire-driven successions in freshwater coastal marshes are poorly documented. Evidence suggests that as with brackish marshes, fire releases a diversity of early successional species that are more palatable to wildlife (Ford and Grace 1998a). Succession of the herbaceous communities in fresh and oligohaline marshes tends to be rapid as it is in salt and brackish marshes. Preburn vegetation can regain its dominance in a few years. Van Arman and Goodrick (1979) reported that 6 months after prescribed burning a Florida freshwater marsh, vegetative recovery was almost complete, and total numbers of animal species and individuals were significantly higher in the burned area than in the adjacent unburned marsh. Increases in these macroinvertebrates and smaller fish populations at the lower end of the food chain suggest that these increases potentially could be passed on up the scale. See Ewel (1995) and Frost (1995) for a more detailed consideration of the role of fire in regulating succession in forested freshwater wetlands.

An increasingly common successional pattern in fresh marshes is due to invasion of Chinese tallow, which has limited tolerance to salinity and is largely confined to fresh and oligohaline wetlands. Evidence indicates that when Chinese tallow invades a wetland or upland grassland it causes a shift from a grass-dominated herbaceous layer to a sparse forb-dominated layer that is much less capable of carrying a fire. As a result, stands of Chinese tallow act as firebreaks. Below some minimum stand density, fire can be used to effectively control Chinese tallow as long as adequate fuel remains (Grace 1998).

Management Considerations—The use of prescribed fire in coastal fresh marshes is much less extensive than in salt and brackish marshes (Chabreck 1988). Fires in salt and brackish marshes, however, often spread into fresh marsh areas, resulting in a relatively frequent burn regime for associations that will propagate fire. Prescribed burning in fresh marshes is more likely to be used for fuel reduction and to control woody plants such as wax myrtle, thinleaf alder, and Chinese tallow than to promote wildlife populations. Nonetheless, when wildlife or cattle production is the goal, the same management recommendations apply to fresh marshes as described previously for brackish and salt marshes.

Prairie

Vegetation Dynamics—A primary effect of fire in tallgrass prairie ecosystems is the control of invading woody species (Anderson and Van Valkenburg 1977; Wade and Menges 1987). The rapid conversion from prairie to forest with the removal of fire was noted in the early 1800s. Muir (1965) observed in Wisconsin that as soon as sufficient firebreaks were created, a thick oak forest invaded the prairie. In 1822, botanist Edwin James reported, "Whenever the dominion of man is sufficiently established in these vast plains, to prevent the annual ravages of fire, trees will spring up" (Pyne 1997). One of the most aggressive woody species in the prairie is eastern redcedar. In the absence of fire this species could quickly become the dominant tree over much of the Ozark region (Beilmann and Brenner 1951). Cedar forests now occupy 6.4 million acres (2.6 million ha) in five Midwestern States, an increase of 113 percent during the last three decades (Schmidt and Leatherberry 1995). In as little as 30 years after fire, a treeless pasture can be converted to a closed canopy cedar forest (Hoch and Briggs 1999).

In savannas, frequent fires tend to be of low intensity, do not kill overstory trees, and create an open understory. Infrequent fires are more intense due to litter accumulation, can kill overstory trees, and promote vigorous sprouting of woody species, often creating a thicket. Fires every 2 to 3 years held woody canopy at a constant level (Faber-Langendon and Davis 1995). Generally, more frequent fires reduced tree canopy while less frequent fires increased tree canopy. Faber-Langendon and Davis (1995) suggested that a 4 year fire interval might be best for controlling tree spread because at this interval fires would burn more intensely than annual or biennial fires.

Fire frequency affects species differently. Generally, big bluestem shows no response to time since fire, while little bluestem, Indiangrass, and switchgrass all decrease with time since fire (Collins and others 1995). Gibson (1988) found that perennial forbs and cool-season grasses increased with time since fire, while annual forbs and warm-season grasses decreased. Annual or frequent burning tends to decrease herbaceous plant diversity in tallgrass prairie. Knapp and others (1999) found that annually burned areas had lower species richness than unburned areas and areas burned every 4 years. Collins and others (1995) found that species richness increases for 7 to 8 years after burning, and that time since burning was the primary agent in determining variation in species composition. Gibson (1988) argued that burning every 4 years might be the best strategy for maintaining maximum diversity (fig. 4-23). In annually burned areas fewer species can become established. In unburned areas litter accumulation creates too much shade for many species. Burning every 3 to 4 years ensures that most of the species present will have at least one "optimal" year for growth and reproduction. Collins (1987) showed that a combination of burning and grazing resulted in higher plant species diversity than burning or grazing alone.

Generally, net primary production increases the growing season following burning (fig. 4-24). If a prairie remains unburned, detritus accumulates that shades the soil, especially at the beginning of the growing season, and limits production (Knapp and Seastedt 1986; Seastedt and Knapp 1993). However, the decomposing detritus adds nitrogen to the soil, and the detrital layer insulates the soil from drying. Researchers have theorized that long-term annual burning would reduce soil nitrogen levels and lead to decreased productivity. However, even after 20 years of annual burning at Konza Prairie, Kansas, productivity has not decreased (Blair and others 1998) probably because a large pool of soil nitrogen buffers the system. Only now are declines in soil nitrogen becoming evident in these annually burned grasslands. The influence of burning on net primary production also depends on interactions with other factors such as drought. In drought years, fire can decrease net primary production (Briggs and Knapp 1995). Following fire, higher levels of net primary production will occur on a long-term unburned area than on annually burned areas, primarily due to the accumulation of soil nitrogen.

Seasonality Influences—Seasonality of fire can have a dramatic effect on species composition and diversity (Platt and others 1988a). Henderson and others (1983) found that early spring burning did not affect cool season grasses while late spring burning reduced flowering up to 70 percent. In contrast, burning in the fall or spring increased flowering in the warm season grasses over flowering in unburned areas; late spring burning treatments showed the greatest flowering activity. Henderson (1992a) determined that late spring burning significantly reduced diversity compared to early spring or fall burning due to

Figure 4-23—Periodic fires increase the diversity of tallgrass prairie plants, Smith Pioneer Cemetery, Vermilion County, Indiana. In its eastern range, the tallgrass prairie is found in isolated “remnant” prairies, old cemeteries, and along railroads. Many are smaller than 1 or 2 acres (0.5 ha) and surrounded by agricultural fields.

Figure 4-24—Productivity of all plants, especially the grasses, increases following fire, Smith Pioneer Cemetery, Vermilion County, Indiana. In the Eastern tallgrass region, dominant grasses such as big bluestem pictured here, could reach heights of 11 feet (3.4 m) or greater. Photo by Greg Hoch.

removal of cool season grasses and several mid- and late-season flowering forb species. Prairie violet and blue-eyed grass, early flowering species prone to damage by spring burning, were not lost. This study also suggested that burning in the fall might benefit prairie violet, blue-eyed grass, and sky blue aster. Henderson (1992b) found that pasque flower was favored by early spring fires that occurred before bud emergence. However, this pattern was reversed in years having high rabbit herbivory or late frosts. Areas burned early in the spring were preferentially grazed by rabbits. In years with late frost the thatch from the previous year seemed to insulate the emerging buds from late frosts.

Howe (1994, 1995) determined that species richness was higher on plots burned in July than on plots burned in March or unburned plots in a restored prairie. Two species, blackeyed Susan and annual fleabane, were found only in the summer burn plots. Summer fires dramatically increased the seedling establishment of forbs but had no effect on seedling establishment of grasses.

Seasonality of burning can affect productivity, especially if the fire influences soil drying and moisture levels (Adams and others 1982; Towne and Owensby 1984). Burning in winter or early spring removes the insulation provided by accumulated litter and allows the upper layers of the soil to dry. Burning late in the spring (just after the warm season grasses emerge) removes the thatch, warms the soil, and allows sufficient light to the soil surface; but there is insufficient time for the soil to dry before the grasses begin growing. The effects of the seasonal timing of fire on the tallgrass prairie can be complex depending on time since last fire, latitude, and rainfall. In many cases, timing of rainfall (Benning and Bragg 1993) and microclimate (James 1985) have a greater influence on productivity than season of burning.

Management Considerations—Most studies in the tallgrass prairie show that prescribed fires should be conducted in the fall, early spring, or summer to maximize plant species diversity. However, burning during these times may lead to direct mortality of some animals or to indirect mortality from the removal of protective cover. If weight gain of livestock is the primary goal, the area should be burned in late spring to maximize production of warm season grasses. Additionally, grazing can significantly influence ecosystem responses to fire. Management should strive to preserve heterogeneity through maintenance or simulation of natural disturbance regimes. Where feasible, mowing or grazing (Collins and others 1998) should be considered in conjunction with burning. Steuter (1990) provides an excellent example of the use of a “natural” disturbance regime for grassland conservation and management.

In remnant prairies, heterogeneity within remnants and regionally between remnants is an important management objective. Larger areas of prairie should be managed as patchworks to promote diversity within the area (Coppedge and Shaw 1998; Steinauer and Collins 1996). Ideally, some element of randomness can be incorporated in the fire management program. This can be accomplished by burning some remnants every 1 to 2 years, some every 4 to 5 years, and some areas every 10 years at different times of year. Attempts should be made to burn only part of an area at a given time. This allows refugia for animal species in the unburned thatch. Many insects, which are important pollinators to prairie plants, winter aboveground. If the entire area is burned, whole populations of these species can be destroyed.

The tallgrass prairie is an extremely dynamic ecosystem. Grassland plants have shown genetic changes as a result of management in as little as 60 years (Painter and others 1989). Many species in remnant prairies have become extinct within the last 60 years probably due to fragmentation and certain management practices; more species probably became extinct prior to this (Leach and Givnish 1996). Gibson (1988) observed that even after several years of identical treatment two areas may still have different plant communities, probably a result of past land use. Thus, it is difficult or impossible to answer, “What plant species were originally found in this general area at the time of settlement by Euro-Americans?” Climate variability can have an even stronger influence on the prairie than careful management. Henderson (1992a) stated that “The vegetation of the prairie seemed to change more from one year of severe drought than from 10 years of frequent early spring, late spring, or early fall burning.”

Bay Forests

Vegetation Dynamics—The overstory was historically dominated by one or more of the following: Atlantic white-cedar, bald cypress, pond pine, slash pine, swamp tupelo, and Blackgum. Other overstory species include sweetbay, red bay, loblolly bay, red maple, and sweetgum. The almost impenetrable understory includes lyonias, titi, swamp cyrilla, gallberries, bays, blueberries, huckleberries, wax myrtle, sweetpepperbush, hollies, Virginia willow, various species of greenbrier, and sphagnum moss. The vegetation of all communities is highly correlated to time since the last fire (Christensen and others 1988). Fire is necessary to cycle nutrients, especially on sites with deep organic soils; the absence of fire is more of a disturbance in pocosins than intense fire (Christensen 1981). During drought conditions, the underlying organic soils burn creating substantial smoke problems, often for weeks at a time. The various plant associations that make up this ecosystem and their response to fire are discussed by Wells (1928), Richardson (1981), Christensen and others (1988), and Richardson and Gibbons (1993).

Management Considerations—Although the original overstory was usually harvested, few bays are currently managed for forest products. In the decades following World War II, management usually consisted of draining, bedding, and planting loblolly or slash pine, or in some cases row crops. On the remaining bays, fire periodicity determines the successional state of the site. If bay forests burn every 2 to 5 decades, they are called shrub bogs (Christensen 1977; Wharton and others 1976), and if burned at least once a decade, herb bogs. If the underlying organic soils are completely consumed, both pocosins and bays will revert to marsh (Richardson and Gibbons 1993). Herb bogs must be burned at least once every decade or they will succeed to shrub bogs (Wharton and others 1976). Where the objective is to maintain herb bogs and their suite of showy herbaceous species, many of special concern, they should be burned on a 1 to 3 year rotation. As succession proceeds, the fire prescription window narrows and becomes less defined. Wind is the major factor determining whether fire will carry through the early successional stages, while drought becomes more important as the flashy herbaceous groundcover is shaded out by the developing shrub layer. Where shrub composition is primarily wax myrtle, gallberry, fetterbush, and other flammable species, such as in the pocosins of North Carolina, these communities remain receptive to wind-driven fires.

Sand Pine

Cone serotiny is weak in the Choctawhatchee variety and strong in the Ocala variety. Both species are thin-barked and easily killed by fire. The Ocala variety recaptures the site with seed from the freshly opened serotinous cones. The Choctawhatchee variety seeds in from adjacent unburned stands. Both varieties are prolific seeders, producing viable seed by age 5 or 6 (Brendemuehl 1990). Two fires in quick succession (less than about 6 years apart) will eradicate this species from a site. Sand pine is somewhat shade-tolerant and rapidly establishes in the direction of the prevailing wind. It often grows in pure, even-aged stands.

Choctawhatchee—This variety is the principal overstory species. Common understory associates include occasional xeric oaks (turkey, bluejack, sand, post) and prickly pear, with a sparse groundcover of wiregrass and bluestems (Brendemuehl 1990).

Young sand pines are sensitive to fire, so when a fire burns through a stand, it usually kills the overstory; however, overstory species composition does not change because of the copious amount of seed available for regeneration. In fact this variety is often described as a weed species because it tends to invade other vegetation types that are downwind. Because of its thin bark, fire is often used to eradicate it when it seeds into other stands.

Ocala—This variety occurs on nutrient-deficient sands as the dominant overstory species in evenaged stands. Common understory associates include evergreen shrubs such as myrtle, sand live oak, Chapman oak, turkey oak, rusty staggerbush, rosemary, scrub palmetto, and saw palmetto (Christensen 1981, Outcalt 1997). Little groundcover is found on these xeric sites although cup lichen, or deer moss as it is locally known (same species as in the Artic), is fairly common.

These stands usually burn with an intense, wind-driven fire that generally consumes all live foliage, kills thin-barked trees, and opens cones allowing the stand to regenerate (Cooper 1951, 1965; Cooper and others 1959).

Management Considerations—Industrial plantations of the **Choctawhatchee** variety have been established in inland Florida and about 200 miles north of its natural range on xeric sand-hill sites in west-central Georgia. This variety can withstand light surface fires once it reaches pole size; prescribed fire under mild burning conditions is sometimes used for hazard reduction in plantations. Care must be taken, however, because the boles of even mature trees are quite susceptible to severe surface fires. Because of the lack of understory competition and slow buildup of fuels, the use of fire in the management of this variety is generally not a high priority. Prescribed fire, however, is advocated for regenerating the **Ocala** variety (Cooper 1953,1973a; Price 1973). A tight prescription and experienced crew are necessary to confine the fire to its intended boundaries and to manage smoke from these stand-replacement fires because of their proximity to urban areas (Custer and Thorsen 1996). See Walker (1967) for a broader discussion of management recommendations for both varieties.

Table Mountain Pine

Vegetation Dynamics—Table Mountain pine is found on xeric, typically south- to west-facing sites (Whittaker 1956) where it is perpetuated by fire, although it will colonize more mesic sites following fire (Williams and Johnson 1990). This species depends upon fire to: (1) melt the wax seal on serotinous cones to release seeds; (2) consume a large portion of the accumulated forest floor to create a receptive seedbed; and (3) reduce competition for sunlight, water, and the pulse of mineralized nutrients important on sterile soils (Zobel 1969). Groeschl and others (1993) stated that fire reduces overall site quality, which results in a more favorable environment for pine than for xeric-site hardwoods.

After examining fire records from 1960 to 1971 in the southern Appalachians, Barden and Woods (1974) found no occurrence of lightning-caused crown fires, perhaps because fine herbaceous fuels had vanished after decades of fire exclusion. Of 85 lightning fire records, only two fires killed a majority of the overstory and none resulted in more than token pine reproduction (Barden and Woods 1976). Several human-caused crown fires during this period killed most of the overstory and resulted in much better pine recruitment. Sutherland and others (1995) found the same situation upon examining two centuries of evidence in a southwestern Virginia Table Mountain pine community. Most existing stands have resulted from fires associated with logging early in the 20th century (Williams 1998).

Management Considerations—The long-term effects of fire exclusion in the Appalachians are becoming more apparent (Williams 1998). Most stands are now degraded and succeeding toward hardwood dominance (Williams 1998; Williams and Johnson 1992). The increased incidence of bark beetle attacks in these stressed, aging stands is accelerating this successional trend. The Southern Appalachian Mountains Assessment (SAMAB 1996) listed Table Mountain pine as a rare community.

In the 1990s interest increased in restoring Table Mountain pine communities (Waldrop and Brose 1999; Welch and Waldrop, in press). Although other natural events such as ice storms can create canopy gaps, reduced duff depths are the overriding requirement for seedling establishment (Williams and Johnson 1992; Zobel 1969); for this, periodic fire is generally responsible (although see Barden 1977 and Williams 1998). Questions regarding the necessity of crown fires are still unresolved (Waldrop and Brose 1999; Whittaker 1956; Zobel 1969) but may be answered by future research.

Spruce-Fir

Spruce-fir forests are a related variant of the extensive boreal forest biome described in chapter 3. Historically, the spruce-fir type was virtually fireproof (Harmon and others 1983; Korstian 1937), but logging followed by fire has devastated this forest type in the Southern Appalachians (Korstian 1937). In many cases, the deep duff layers have been consumed down to bare rock, and species composition has shifted to yellow birch, pin cherry, and mountain ash. According to Minckler (1944), rehabilitation of these sites will take 500 to 1,000 years.

Mature spruce may initially survive low-intensity fires, but Stickel and Marco (1936) found that over half the survivors had been attacked by fungi, insects, or both within 3 years postburn. Thus underburning does not appear to be an appropriate practice in management of this forest type. See Walker (1967) for more discussion of management recommendations.

Atlantic White-Cedar

Vegetation Dynamics—Once trees reach pole size, copious amounts of seed are produced (about 500,000 per pound) from several thousand cones per tree. The seed is released every fall and stored in the forest floor where it remains viable for about 3 years. Stands tend to be exceedingly dense. Lower branches die at an early age but persist for several decades before being sloughed off. Under normal (wet) conditions, crown fires destroy the aboveground vegetation. As droughts get progressively worse, more of the forest floor and stored seed are consumed. Two fires in close succession, before the seed bank is replenished, will produce herb bog, shrub bog, or bay forest depending upon the future fire return interval.

Reestablishment after fire depends upon fire severity and age of the stand. When the water table is high and fires just skim off the top of the forest floor, the replacement stand can be Atlantic white-cedar, pond pine, or sprouting hardwood trees and shrubs. Succession depends upon the amount of stored cedar seed, preburn species composition, and the composition of unburned adjacent stands. Fires during severe drought, which consume much of the organic soil, result in open water and a dense cover of leatherleaf, hardwoods capable of sprouting, and cedar, depending on the factors just mentioned and postfire precipitation (Little 1959).

Generally, dense stands of Atlantic white-cedar are formed after fire when the water table is neither much above nor below the top of the peat. A high water table allows sprouting hardwoods to gain a competitive advantage before cedar seeds germinate. A low water table allows most of the cedar seed to be consumed. In both cases, regeneration may be insufficient to produce a monotypic cedar canopy.

Management Considerations—Current fire management of Atlantic white-cedar stands is to exclude all fire until the stands are harvested. Then fire can be used to dispose of logging debris and prepare the site for the next crop. See Walker (1967) for more discussion of management recommendations.

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Chapter 5: Fire in Western Forest Ecosystems

Understory Fire Regimes

Major Vegetation Types

Major forest types that are characterized by nonlethal understory fire regimes include those where ponderosa pine or Jeffrey pine has been a major component either as a fire-maintained seral type or as the self-perpetuating climax (table 5-1). This includes extensive areas throughout the Western United States from northern Mexico to southern British Columbia, Canada (Little 1971). Also, sizeable areas of open woodlands dominated by Oregon white oak, California black oak, blue oak, or Digger pine were characterized by frequent understory fires largely due to deliberate burning by Native Americans (Boyd 1986; Lewis 1973). These occurred in relatively dry areas west of the Cascades and Sierra Nevada from the southwest corner of British Columbia to southern California. Recent studies suggest that large areas of the redwood forest in coastal northern California were characterized by frequent understory fires resulting from burning by Native Americans (Brown and Swetnam 1994; Duncan 1992; Finney and Martin 1989; Greenlee and Langenheim 1990). Additionally, portions of other forest types may also have had understory fire regimes. For example, some areas of interior Douglas-fir near the drought-caused lower timberline in the higher valleys of the Rocky Mountains may have been maintained in open condition in understory fire regimes (Arno and Gruell 1983; Arno and Hammerly 1984). Nevertheless, most of this type is best represented by the mixed regime.

Fire Regime Characteristics

Fires were frequent, with mean intervals between 5 and 30 years in most areas (Kilgore 1987; Martin 1982) in the ponderosa pine type and at similar intervals in the redwood type; fires occurred even more frequently in some of the oak-prairie communities. At one extreme, fire intervals averaged 1 to 2 years in an area of northern Arizona and no more than 10 years throughout the Southwestern ponderosa pine type due to abundant lightning activity (Dieterich 1980). Conversely, near the cold limits of the ponderosa pine type in western Montana, mean fire intervals averaged between 25 and 50 years (Arno and others 1995b). Relatively short mean intervals occurred where many ignitions were made by Native Americans, while longer

Table 5-1—Occurrence and frequency of presettlement fire regime types by Forest and Range Environmental Study (FRES) ecosystems, Kuchler potential natural vegetation classes (1975 map codes), and Society of American Foresters (SAF) cover types. Occurrence is an approximation of the proportion of a vegetation class represented by a fire regime type. Frequency is shown as fire interval classes defined by Hardy and others (1998) followed by a range in fire intervals where data are sufficient. The range is based on study data with extreme values disregarded. The vegetation classifications are aligned to show equivalents; however, some corresponding Kuchler and SAF types may not be shown.

			Fire regime types						
			Understory		Mixed		Stand-replacement		
FRES	Kuchler	SAF	Occur[a]	Freq[b]	Occur	Freq	Occur	Freq	Nonfire
Coastal[c] Douglas-fir 20	Cedar-hemlock-Douglas-fir K022	Douglas-fir-w. hemlock 230			M	2: 40-150	M	3	
	Mosaic of above and Oregon oak woods K028	Pacific Douglas-fir 229			M	2: 40-150	M	3: 200-500	
		Red alder 221							
	Calif. Mixed evergreen K029	Douglas-fir-tanoak-Pacific madrone 234			M	1			
Redwood 27	Redwood K006	Redwood 23	M	1: 5-25	m	2			
Hemlock-Sitka spruce 24	Spruce-cedar-hemlock K001	Sitka spruce 223			m	2	M	3	m
		W. hemlock 224					M	3	
		W. hemlock-Sitka spruce 225					M	3	
		W. redcedar-w. hemlock 228					M	3	
W. hardwoods 28	Oregon oakwoods K026	Oregon white oak 233	M	1					
	California oakwoods K030	Blue oak-digger pine 250	M	1					
		Canyon live oak 249			M	1,2			
		California coast live oak 255			M	1,2			
Coastal[c] fir-spruce 23	Silver fir-Douglas-fir K003	True fir-hemlock 226					M	3	m
	Fir-hemlock K004	Mountain hemlock 205					M	2,3	m
Inland forests									
Ponderosa pine 21	W. ponderosa pine K011	Pacific ponderosa pine 245	M	1: 5-30	m	2			
	Pine-Douglas-fir K018	Pacific ponderosa-Douglas-fir 244	M	1: 5-30	m	2			
	Mixed conifer K005	Sierra Nevada mixed conifer 243	M	1: 5-30	m	2			
		Jeffrey pine 247	M	1: 5-30	m	2			
		California black oak 246	M	1: 5-30	m	2			
	Arizona pine K019	Interior ponderosa pine 237	M	1: 1-25	m	2			
	E. ponderosa K016	Interior ponderosa pine 237			M	2			
	Black Hills pine K017	Interior ponderosa pine 237	m	1	M	2	M	2	
Interior[c] Douglas-fir 20	Douglas-fir K012	Interior Douglas-fir 210	m	1,2	M	2: 25-100			
Larch 25	Grand fir-Douglas-fir K014	W. larch 212			M	2: 25-200	M	2,3	
		Grand fir 213			m	2	M	2,3	
W. white pine 22	Cedar-hemlock-pine K013	W. white pine 215			M	2: 50-200	M	3: 130-300	

(con.)

Table 5-1—Con.

FRES	Kuchler	SAF	Fire regime types: Understory Occur[a]	Understory Freq[b]	Mixed Occur	Mixed Freq	Stand-replacement Occur	Stand-replacement Freq	Nonfire
Lodgepole pine 26	Lodgepole pine-subalpine K008	California mixed subalpine 256			M	2			
Rocky Mountain lodgepole pine[c] 26	W. spruce-fir K015	Lodgepole pine 218			M	2: 25-75	M	2,3: 100-300	
	W. spruce-fir K015	Whitebark pine 208			M	2: 50-200	M	3: 150-300	
Interior[c] fir-spruce 23	W. spruce-fir K015	Engelmann spruce-subalpine fir 206					M	2,3: 100-400	m
	Spruce-fir-Douglas-fir K020	White fir 211			M	2	M	2,3	
		Blue spruce 216			M	2	M	2,3	
W. aspen[c] 28	W. spruce-fir K015	Aspen 217			m	2	M	2	

[a]M: major, occupies >25% of vegetation class; m: minor, occupies <25% of vegetation class.
[b]Classes are 1: <35 year, 2: 35 to 200 years, 3: >200 years.
[c]Added subdivision of FRES.

intervals occurred on similar forest sites that were more remote from aboriginal occupation. For example, in a western Montana study the mean fire interval from 10 heavily used areas was 9 years while the mean interval from 10 remote areas on similar sites was 18 years (Barrett and Arno 1982).

In the Southwestern ponderosa pine type, major fire seasons occur after snow melt (April and May) and in mid-summer just before the monsoon rains begin, and a secondary season exists in the fall. In most other areas the main lightning fire season is summer; whereas Indian burning apparently occurred to some extent in spring, summer, and fall (Barrett and Arno 1982; Gruell 1985a, 1985b). Low-intensity surface fires were characteristic and may have been quite large where dry forests and adjacent grasslands were extensive—for example, on the gentle topography of high plateaus in northern Arizona and New Mexico. In contrast, in rugged mountainous topography, the understory fire regime was often confined to small areas of dry sites on south-facing slopes (Arno 1980). The adjacent moist sites supported other, denser forest types, which burned less often and in mixed or stand-replacement fire regimes. When stand-replacement fires burned the adjacent moist types in 1889 and 1910 in the Northern Rocky Mountains, the dry forest types still burned primarily in a nonlethal manner. See chapter 6 for additional information about the Southwestern ponderosa pine type.

Fuels

During periods of high fire frequency, fuels were primarily herbaceous material and forest floor litter. After fire suppression became effective, forest floor duff and live fuels such as shrubs and conifer regeneration accumulated. Measurements in recent decades (Brown 1970; Brown and Bevins 1986; Sackett 1979) show that litter typically ranges from 0.6 to 1.4 tons/acre (1.3 to 3.1 t/ha) and the entire forest floor of litter and duff averages about 12 tons/acre (27 t/ha) in both Arizona and Northern Rocky Mountain areas. Forest floor quantities as high as 40 tons/acre (90 t/ha) have been measured (Harrington 1987b). During periods of frequent fire, forest floor quantities would typically range from 1 to 4 tons/acre (2.2 to 9.0 t/ha). Herbaceous fuels range from practically none in dense stands to as much as 0.5 tons/acre (1.1 t/ha) in open stands on productive sites. In the Black Hills of South Dakota, herbaceous fuel quantities in open stands of ponderosa pine averaged 440 lb/acre (490 kg/ha), which was six times greater than in closed stands (Uresk and Severson 1989). Herbaceous fuel quantities are typically about 400 lb/acre (448 kg/ha).

Frequent low-intensity surface fires perpetuated open stands of trees whose lower branches were killed by fire. With fire suppression, accumulated fuels support higher intensity fire including torching and crowning behavior and longer periods of burnout. The increased burn severity results in greater mortality to plants and soil organisms. Managers can easily overlook the significance of forest floor fuels; the upper layer (litter) and part of the middle (fermentation) layer provide the highly combustible surface fuel for flaming combustion and extreme fire behavior during severe fire weather. The lower part of the fermentation layer and the humus layer make up the ground fuel that generally burns as glowing combustion. A substantial amount of forest floor material can remain after an area is initially burned (Sackett and Haase 1996).

Postfire Plant Communities

Ponderosa Pine/Jeffrey Pine and Ponderosa Pine-Mixed Conifer

Pre-1900 Succession—These semiarid forest types are widespread in the inland portions of western North America. They include pure pine climax types, which are abundant in plateau areas of northern Arizona and New Mexico, central Oregon, and eastern Washington. They also encompass many sites in the inland mountains where pines are seral to more shade-tolerant conifers—interior Douglas-fir, white fir, grand fir, or incense-cedar. Prior to 1900 these pine communities experienced frequent fires as a result of highly combustible leaf litter, an abundance of cured herbaceous vegetation, and a long season of favorable burning weather. Stands had an open, parklike appearance, dominated by large old, fire-resistant trees. Shrubs, understory trees, and downed logs were sparse, as testified to by dozens of historical photographs and narrative accounts (Cooper 1960; Leiberg 1899; Wickman 1992). Travelers often rode horseback or pulled wagons for miles through these forests without the need of cutting trails.

Undergrowth was primarily of fire-resistant grasses and forbs that resprouted after each burn. Shrubs were suppressed by the frequent burning coupled with overstory competition (Gruell and others 1982). In most stands, duff depth probably averaged only about half an inch (Keane and others 1990a). The majority of overstory trees survived each fire, while many of the understory trees were killed. The most fire-resistant species—ponderosa pine, Jeffrey pine, and western larch—were favored. In the large areas of this type where ponderosa pine is seral, it maintained dominance only because of the frequent fires.

In much of the pure ponderosa pine type and in the seral pine type on dry sites, pine regeneration occurred whenever overstory trees died, thereby creating small openings. These open microsites allowed a few seedlings to grow fast enough to gain resistance to survive the next fire (Cooper 1960; White 1985). Thus stands tended to be uneven-aged and often contained some 400 to 600 year old trees (Arno and others 1995b, 1997). Trees were often distributed in small even-aged clumps. When small patches of overstory were killed by fire or bark beetles, subsequent fires consumed these fuel concentrations, locally reducing grass competition and creating mineral soil seedbeds. This favored establishment of ponderosa pine seedlings, allowing a new age class to develop in a micromosaic pattern within a stand (Cooper 1960). These effects helped create an uneven-age stand structure composed of small, relatively even-aged groups (Cooper 1960; Arno and others 1995b).

Mixed-severity fire regimes were characteristic on some of the relatively moist sites and on steep slopes throughout the ponderosa pine type. Variable and mixed regimes were evidently widespread in ponderosa pine communities in the Front Range in Colorado and in the Black Hills of South Dakota and Wyoming, and stand-replacement regimes occurred in some situations in the Black Hills. These are described under the section "Mixed Fire Regimes," later in this chapter.

Post-1900 Succession—Important changes have occurred in these forests since 1900 due to interruption of frequent burning. Nonlethal fire has decreased while lethal fire has increased (fig. 5-1). Reduced fire began in the late 1800s as a result of (1) relocation of Native Americans and disruption of their traditional burning practices; (2) fuel removal by heavy and extensive livestock grazing; (3) disruption of fuel continuity on the landscape due to irrigation, cultivation, and development; and (4) adoption of "fire exclusion" as a management policy. The general result has been development of dense conifer understories, commonly adding 200 to 2,000 small trees per acre beneath old growth stands or thickets of 2,000 to 10,000 small trees per acre where the overstory was removed. Densely overstocked conditions have resulted in slow growth and poor vigor of most trees in a large proportion of the ponderosa pine type where adequate thinning treatments have not been applied. Stand stagnation is accompanied by a sparse representation of nonflowering herbs and shrubs, which reflects a loss of natural biodiversity and of forage for wildlife (Arno and others 1995a). Growth stagnation renders even the dominant trees highly vulnerable to mortality in epidemics of bark beetles, defoliating insects, diseases such as dwarf mistletoe, and various root rots (Biondi 1996; Byler and Zimmer-Grove 1991; Cochran and Barrett 1998). For example, in the 1980s about a million acres of ponderosa pine-fir forests in the Blue Mountains of eastern Oregon suffered heavy mortality from the above agents as a result of overstocking and growth stagnation related to fire exclusion (Mutch and others 1993).

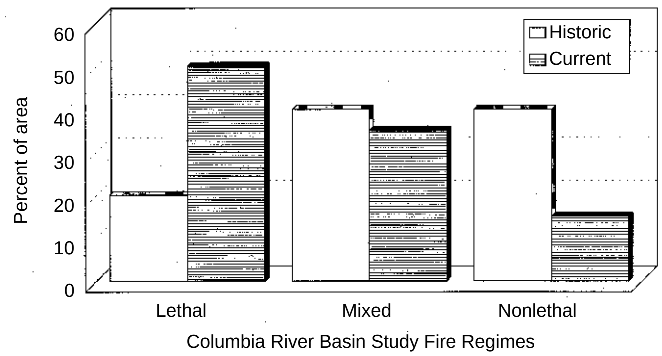

Figure 5-1—Change in fire severity between historic (pre-1900) and current conditions for Forest Service and Bureau of Land Management administered forested potential vegetation groups in the Interior Columbia Basin and portions of the Klamath and Great Basins (Quigley and others 1996).

In many stands, duff mounds 6 to 24 inches deep have accumulated under old growth trees, and burning these mounds may girdle and kill the trees (fig. 5-2). Stands of small, slow-growing pines have also colonized former grasslands (Gruell 1983). Overstory trees have been removed in more than a century of logging, primarily partial cutting, and this has aided development of thickets of small trees. On sites where ponderosa pine is seral, there has been a compositional shift to the shade-tolerant species. These successional changes have resulted in a buildup of understory or ladder fuels that now allow wildfires to burn as stand-replacing crown fires.

A combination of heavy forest floor fuels and dense sapling thickets acting as ladder fuels, coupled with the normally dry climate and frequent lightning- and human-caused ignitions, has resulted in a dramatic increase of severe wildfires in the ponderosa pine type in recent decades (Arno 1996; Williams 1995). For example, approximately 1 million acres (405,000 ha), largely in this type, burned in severe wildfires in central in Idaho between 1986 and 1996 (Barbouletos and others 1998).

Management Considerations—Fires of the past were important to the evolution of ponderosa and Jeffrey pine forests (Keane and others 1990a; Mutch 1970; Weaver 1967). Today, prescribed fire and wildland fire are the obvious and most feasible substitutes for filling the ecological role of historic fires in restoring these wildland ecosystems (fig. 5-3). Many alternatives exist for employing prescribed fire and fuels management treatments to improve forest health and reduce excessive ladder fuels in ponderosa pine and pine-mixed conifer types (Arno 1988). Different prescriptions can be chosen to enhance critical resources or values (Fiedler and others 1996), such as maintenance of old growth in a natural area, encouraging

Figure 5-2—A prescribed burn during October in central Idaho removed duff as thick as the distance between the hands, in this case with little damage to large ponderosa pine trees.

Figure 5-3—Prescribed fire during May is being used to reduce fuel loadings after a retention shelterwood harvest in a ponderosa pine/Douglas-fir forest.

browse and cover on a big game winter range, maintaining forest structure favored by neotropical migrant bird species, northern goshawk, flammulated owl, and other species of concern; protecting homes and watersheds from severe wildfire while maintaining visual screening; or providing a continuous supply of forest products without disruption of esthetics. However, most long-term management goals for wildlands in these forest types can be enhanced by some form of prescribed fire and fuels management.

Many contemporary stands have such an altered stand structure and composition, including a buildup of understory fuels, that it is difficult to restore forest health with prescribed fire alone. Silvicultural cutting and pile-burning or removal of excess small trees may be necessary to allow successful application of prescribed fire and to return to more open structures dominated by vigorous trees of seral species (Arno and others 1995a). Failed attempts to restore more natural stand conditions with prescribed burning alone may result from inappropriate use of fire as a selective thinning tool in dense fire-excluded stands, or from burning too little or too much of the accumulated forest floor fuels. A better approach to the latter problem may be to apply two or even three burns to incrementally reduce loadings (Harrington and Sackett 1990). Once a semblance of the desired stand and fuel conditions have been established, stands can thereafter be maintained more routinely with periodic burning or a combination of cutting and fire treatments. Prescribed fire can be used in wildernesses and natural areas to maintain natural processes.

The advantages of using fire and improvement cuttings to restore and maintain seral, fire-resistant species include: (1) resistance to insect and disease epidemics and severe wildfire; (2) providing continual forest cover for esthetics and wildlife habitat; (3) frequent harvests for timber products; (4) stimulation of forage species; and (5) moderate site disturbances that allow for tree regeneration (Mutch and others 1993). Frequent prescribed fires will not produce heavy screening or hiding cover, which is not sustainable over large areas in these forests. But such fires can help maintain moderate cover and screening indefinitely (Martin and others 1988). Management of a large proportion of the forest in open conditions can help ensure protection of strategically located patches of heavier cover (Camp and others 1996). The frequent disturbance cycles can also produce and maintain large old trees characteristic of pre-1900 forests and of high value for wildlife habitat, esthetics, and selective harvesting for lumber. In other words, the management approach of using a modified selection system and periodic burning can be used to maintain remnant old growth stands and to create future old growth (Fiedler 1996; Fiedler and Cully 1995).

Prescribed fire and wildland fire must be introduced cautiously in stands where leave trees have poor vigor or where tree roots are located in a deep duff layer (Harrington and Sackett 1992). Burning thick forest floor fuel layers can mortally injure roots and boles of old pines that in past centuries survived many fires (Sackett and Haase 1996). Exceptionally poor tree vigor is reflected by growth stagnation over two or more decades in the dominant trees in both second growth and old growth stands. This widespread stand stagnation has resulted from basal area stocking levels often two or more times those of historic conditions (Arno and others 1995b; Biondi 1996; Cochran and Barrett 1998; Habeck 1994). Fire can be highly stressful to trees that are suffering from growth stagnation, and may, for instance, allow bark beetles to inflict major damage to the weakened trees. If vigor of desired leave trees is poor, it may be wise to thin mechanically and allow the leave trees to recover or release somewhat before applying fire, in perhaps 2 years (Fiedler and others 1996).

Options for alleviating the condition of poor vigor or deep duff are not ideal. Managers can simply accept a 20 to 50 percent loss of old growth in a single fuel-reduction burn as being a cost of decades of fuel buildup. Most old growth stands are now more heavily stocked than in pre-1900 times, and some of the trees would probably have been killed by natural fires had suppression not intervened. Alternatively, fuels could be manually removed or raked away and dispersed from around the boles of old growth trees. The use of a burn prescription that reliably removes a portion of the fuel mound around a big tree has not been found. If glowing combustion is able to establish in the deeper mounds, it can continue even with high moisture content of that material and result in total consumption and prolonged heat release.

Improvement cutting, thinning, and understory cutting with whole-tree removal or pile burning may be necessary to achieve open stocking levels that will sustain vigorous tree growth and to reduce ladder fuels (Fiedler and others 1996) (fig. 5-4). Harvesting and thinning should be designed to retain the most vigorous trees. If stems are tall and slender, as in dense second growth stands, it may be necessary to leave clumps of three to five trees for mutual protection against breakage by wet snow and windstorms. The restoration cutting process may require thinning in two to three steps over 15 to 20 years. Spot planting of the desired seral tree species in open burned microsites can be used when shade-tolerant species have taken over (Fiedler and others 1996).

Prescribed fire can be used in a variety of seasons to meet management objectives (Harrington 1991; Kalabokidis and Wakimoto 1992; Kauffman and Martin 1990; Kilgore and Curtis 1987; Martin and Dell

Figure 5-4—(A) An unwanted ponderosa pine-fir stand that resulted from partial cutting of large pines followed by development of fir thickets. (B) Similar stand after the first restoration treatment consisting of thinning to favor remaining pines and jackpot burning.

1978; Weatherspoon 1990). This includes burning in late winter following snowmelt, when understory Douglas-fir or true firs may burn readily and are easily killed because of low foliar moisture content; spring; cloudy-humid days in summer; or autumn. Trees are less susceptible to fire damage when entering dormancy in late summer and fall. Potential for fire damage can be reduced by thinning and whole-tree removal, burning when slash is still green, successive burns starting with damp fuels, and raking duff mounds away from boles of old growth trees. Fire intensity can be reduced by burning at dawn, late evening, or at night. Conversely, to enhance burning in stands where surface fuels are inadequate, such as in fir thickets, thinning can be used to create sufficient loadings of cured slash. Alternatively, waiting for grass fuels to cure in the fall may be effective. If enclaves of dense growth are desired, such as for wildlife habitat, these can be protected from wildfire if buffered within a matrix of treated stands that have light fuel loadings.

During the past decade tens of thousands of acres of the ponderosa pine type have been treated with prescribed fire in central Oregon, northwestern Montana, and parts of Arizona and New Mexico (Kilgore and Curtis 1987; Losensky 1989; Simmerman and Fischer 1990). If these applications can be greatly expanded they could correct some of the severe forest health and wildfire problems that exist in the Inland West (Everett 1994; McLean 1992; Mutch and others 1993).

Redwood

Pre-1940 Succession—Forests where redwood was the most abundant tree covered about 1 million acres in a narrow strip along the coastal fog belt from the extreme southwestern corner of Oregon to Monterey County, California (Roy 1980). It has long been recognized that relatively frequent understory fires were historically a common feature of the redwood forest and that fires seldom killed many of the large redwood trees (Fritz 1931). Redwood is even more fire resistant than coast Douglas-fir, its primary associate, and unlike most conifers in this region, redwood sprouts vigorously from dormant buds when the crown is heavily scorched. Pacific madrone and tanoak, the most abundant hardwoods in the redwood type, are fire susceptible. They resprout when top-killed by fire, but burning probably taxed them physiologically when they were growing beneath an overstory.

Recent advances in dating fire scars on redwood stumps have shown that presettlement (pre-1850) fire intervals averaged between about 5 and 25 years, shorter than previously estimated (Brown and Swetnam 1994; Finney and Martin 1989, 1992). The pattern of frequent fires on fire-scarred tree stumps was traced back to about 1300 A.D. in one study area (Finney and Martin 1989) and to about 800 A.D. in another (Fritz 1931). There is a convergence of evidence from historical journals and archeological and anthropological discoveries that Indian burning was primarily responsible for the frequent fires prior to the mid-1800s (Duncan 1992; Greenlee and Langenheim 1990; Lewis 1973; Stuart 1987). Coincidentally, the redwood forest is located close to interior coastal valleys where detailed journal accounts of Indian burning have been assembled (Boyd 1986, 1999). In the presettlement period, before logging occurred, the frequent fires probably kept understories open and reduced forest floor fuels. Overstories of the tall trees with branch-free lower boles were relatively dense because of high site productivity and redwood's fire resistance.

During the settlement period, in the later 1800s and early 1900s, Anglo-Americans conducted logging that created large quantities of slash. These inhabitants continued a pattern of frequent burning in conjunction with logging and for grazing or other purposes. They also allowed accidental fires to spread.

Post-1940 Succession—Starting in the 1930s and 1940s, land use patterns changed with the implementation of the California Forest Practices Act and with more vigorous fire suppression (Greenlee and Langenheim 1990; Stuart 1987). The number of escaped fires from agricultural or logging activities was reduced greatly; but the effects of this reduction of fires have received little evaluation. It is generally assumed that the long-lived, shade-tolerant redwoods will continue to dominate regardless of removal of underburning. Successional relationships in redwood forest communities (Zinke 1977) suggest that with suppression of underburning, shade-tolerant shrubs, understory hardwoods, and western hemlock will increase as will forest fuels in general. In the extensive parks and preserves where redwood forests are protected from logging, continued exclusion of fire will probably allow fuels to accumulate to levels that support higher intensity wildfires, which can escape suppression. Thus, the removal of underburning may ultimately result in a mixed fire regime where wildfires kill significant portions of the overstory and may induce soil damage on the steep slopes associated with a large portion of the redwood type (Agee 1993; Atzet and Wheeler 1982). Additionally, removal of underburning will probably reduce the abundance of early seral species (including chaparral species) and will allow trees to colonize the small openings and prairies associated with the redwood forest belt (Finney and Martin 1992; Greenlee and Langenheim 1990; Zinke 1977).

Oregon Oak Woodlands

Pre-1850 Succession—Open woodlands dominated by Oregon white oak once occupied the driest climatic areas throughout the Puget Sound-Willamette Valley lowlands and southward in dry valleys behind the coastal mountains to the California border. (Farther south this woodland expands to cover large areas of dry hilly terrain and it is dominated by several species of oak in addition to other hardwood trees; those communities are described under “Western Oaks” in chapter 6.) Oregon white oak dominated in open woodlands and savannas associated with valley grasslands, and isolated small prairies that were surrounded by the extensive coast Douglas-fir forest. Oak woodlands also were associated with droughty sites such as bedrock with shallow soils on the southeast coast of Vancouver Island and in the San Juan archipelago, in the rain shadow of the Olympic Mountains.

Ample evidence from journal accounts and archeological sources shows the extensive oak woodlands of the Willamette and other major valleys persisted as a “fire climax” maintained by frequent aboriginal burning (Agee 1993; Boyd 1986, 1999; Habeck 1961). Prior to the influx of Euro-American settlers in the mid-1840s, the Kalapuyan Indians and other tribes typically set fire to large areas of the Oregon oak woodlands to aid hunting, food plant harvest, and for other purposes (Boyd 1986). Firing was commonly done in September, and many areas were burned at short intervals, perhaps annually in some areas. Similar patterns of frequent burning to maintain valley grasslands, isolated prairies, and open oak woodlands are described from northwestern Washington southward to central California (Lewis 1973; Boyd 1986, 1999), but these practices are well documented only in the Willamette Valley of northwestern Oregon.

Most of these fires must have been characterized by short duration flaming as they quickly consumed grass and litter that had accumulated since the previous burn. The thick-barked oaks survived, but regeneration of all shrubs and trees would have been heavily thinned by frequent burning. Grass flourished. The effect of frequent burning on oak regeneration from sprouts and seedlings is not known, but this species was more successful in establishment than Douglas-fir or other competitors under a regime of frequent burning. Results of experimental burning suggest that heavily burned microsites are favorable for oak seedling establishment (Agee 1993). Also, about half of the 1 to 9 year old seedlings burned in a prescribed fire resprouted and were alive 3 years later.

Post-1850 Succession and Management Considerations—In former oak savannas, fire exclusion has led to an increased density of shrubs and oaks, transforming them into woodlands (Agee 1993). In former oak woodlands, shrubs, Douglas-fir, and other tree species are replacing oaks. Livestock grazing and fire exclusion have been major factors in the successional change that has occurred in Oregon oak woodlands since Euro-American settlement. Logging, clearing, and firewood harvest also have changed many woodlands. Additionally, large areas of oak woodlands have been displaced by agricultural, commercial, industrial, and residential development.

Today, 150 years after Euro-American settlement began, only a general idea of presettlement conditions can be hypothesized for most stands. Much of the remaining undeveloped area in this type will be replaced successionally by the year 2010 unless prescribed fire and other restoration treatments are conducted (Agee 1993). To complicate restoration of oak communities, a variety of introduced herbaceous plants and the shrub Scotch broom are now established. Introduced plants can increase as a result of some treatments. Nevertheless, some strategies of burning carefully coordinated with cutting, seeding of native plants, and other treatments hold some promise (Agee 1993; Sugihara and Reed 1987). Restoration of structural characteristics of the oak communities is an attainable goal (see Agee 1993).

Mixed Fire Regimes

Major Vegetation Types

Major forest types include coast Douglas-fir, redwood, California red fir, interior Douglas-fir, western larch, lodgepole pine, whitebark pine, and ponderosa pine types east of the continental divide in Montana, South Dakota, Wyoming, and Colorado. (There is evidence that a mixed regime may have been important for perpetuation of giant sequoia groves in the Sierra Nevada; Stephenson and others 1991; Swetnam 1993.) These forests are widespread in the upper Great Plains, the mountains of the Western United States, and in the northwest coastal regions, extending into southern British Columbia (fig. 1-2). In large portions of their distributions, some of these forest types are also characterized by stand-replacement fire regimes in large portions of their distributions, evidently as a result of differences in climate or topography. An example of this contrast is the larch-lodgepole pine forest in Glacier National Park, Montana, where the northern portion was under a mixed fire regime while the southern portion had a stand-replacement fire regime (Barrett and others 1991). A similar contrast is provided by lodgepole pine types in southwestern Alberta as described by Tande (1979) and Hawkes (1980).

Fire Regime Characteristics

Fires were variable in frequency and severity, and perhaps because these situations are difficult to characterize in simple terms, they have been largely overlooked in previous fire regime classifications (Brown 1995). Mean fire intervals were generally longer than those of understory fire regimes and shorter than those in stand-replacement fire regimes. However, some individual fire intervals were short (<30 years), while the maximum intervals could be quite long (>100 years). Our mixed category covers the spectrum of fire regimes between nonlethal regimes and those where stand-replacement fires were typical.

As described in chapter 1, mixed fire regimes may consist of a combination of understory and stand-replacement fires. Examples are the seral ponderosa pine-western larch forests in western Montana that burned in replacement fires at long intervals (150 to 400+ years) with nonlethal underburns at short intervals (20 to 30 year averages) in between (Arno and others 1995b, 1997).

Mixed severity fire regimes may also be characterized by fires that killed a large proportion of fire-susceptible species in the overstory (such as western hemlock, subalpine fir), but spared many of the fire-resistant trees (such as redwood, Douglas-fir, larch, ponderosa pine). These fires tended to burn in a fine-grained pattern of different severities, including patches where most of the moderately susceptible trees (such as California red fir, white fir, lodgepole pine) survived. Any given location within a mixed fire regime could experience some stand-replacement fires and some nonlethal fires along with a number of fires that burned at mixed severities.

Evidence of mixed severity fire in ponderosa pine is suggested in two landscape photocomparisons of central Montana from the 1880s to 1980 (Gruell 1983, plates 32 and 43). Relatively long fire intervals and mixed burning also occurred in ponderosa pine in the Colorado Front Range (Laven and others 1980) and in the Black Hills of South Dakota (Brown and Sieg 1996; Shinneman and Baker 1997). The mixed fire regime is reported to have been common in mixed-conifer forests of northern Idaho (Zack and Morgan 1994; Smith and Fischer 1997) and western Montana (Arno 1980).

Pre-1900 fires often covered large areas. The uneven burning pattern in mixed fire regimes was probably enhanced by mosaic patterns of stand structure and fuels resulting from previous mixed burning. Thus, past burn mosaics tended to increase the probability that subsequent fires would also burn in a mixed pattern. Complex mountainous topography also contributed to variable fuels and burning conditions, which favored nonuniform fire behavior.

Fuels

During the presettlement period fuels were probably quite variable spatially and temporally. At a given time, some segments of the vegetative mosaic would be patches of postfire regeneration that had arisen where the last fire killed much of the overstory. Fuel loadings in these patches might increase dramatically as dead trees and limbs fell into a developing patch of saplings. If these regenerated patches burned again, the resulting "double burn" might be an area cleared of most living and dead fuel and thereafter more likely to support nonlethal underburning in the next fire. In the presettlement period a given fire could burn day and night for 2 to 3 months under a great variety of weather conditions in a hodgepodge of different vegetation and fuel structures. Reburning might occur later in the same fire event. This could result in an intricate pattern of different fire effects on the landscape. Such complex burning patterns are difficult to imagine in modern stands where 60 to 100 years of fire exclusion have allowed most of the landscape mosaic to age and advance successionally. Patches of late-successional forests with accumulations of dead and living fuels have coalesced, increasing the likelihood of fires of unusual size and severity (Barrett and others 1991).

The ranges in fuel loadings observed in vegetation types characterized by mixed and stand-replacement fire regimes exhibit considerable overlap. Fuel loadings vary widely within broad vegetation classes due to stand history and site productivity. Dead woody fuels accumulate on the ground often in a haphazard manner due to irregular occurrence of natural mortality factors such as fire, insects, disease, tree suppression, and wind and snow damage. However, the greatest fuel loadings tend to occur on the most productive sites, which are predominately stand-replacement fire regimes. For example, in the Northern Rocky Mountains downed woody fuel loadings ranged from an average of about 10 tons/acre (22 t/ha) on low productivity (30 cu ft/acre/yr) sites to about 30 tons/acre (67 t/ha) on high productivity (90 cu ft/acre/yr) sites (Brown and See 1981).

Average fuel loadings determined from extensive forest surveys in the Northern Rocky Mountain National Forests (Brown and Bevins 1986; Brown and See 1981) indicate that quantities of duff and downed woody material differ between mixed and stand-replacement fire regimes (table 5-2). For example, total woody fuel loadings for the spruce-fir and cedar-hemlock types, which are stand-replacement regimes, averaged about 30 tons/acre. It averaged about 17 tons/acre in Douglas-fir and lodgepole pine types, which are characterized by mixed and stand-replacement regimes. Variability within stands and cover types was considerable. Fuel loading distributions for forest floor and all classes of downed woody material were highly skewed, with long right-handed tails. The ratios of medians-to-means averaged close to 0.6 (Brown and Bevins 1986). These statistics indicated that fuels were not uniformly distributed but concentrated in scattered patches.

Downed woody fuels greater than a 1 inch diameter are considered coarse woody debris, which has

Table 5-2—Average loadings of forest floor and downed woody fuel by diameter class for randomly located sample points in Northern Rocky Mountain forest types: Douglas-fir (DF), Lodgepole Pine (LP), Larch/grand fir (L/GF), Spruce/fir (S/F), and Cedar/hemlock (C/H).

	USDA Forest Service forest survey cover types				
Fuel	**DF**	**LP**	**L/GF**	**S/F**	**C/H**
	tons/acre				
Litter	0.56	0.35	0.66	0.52	0.85
Duff	13.0	16.0	21.8	25.4	25.4
Woody					
0 to ¼ inch	.18	.22	.22	.12	.30
¼ to 1 inch	1.0	1.0	1.3	1.0	1.3
1 to 3 inches	1.8	2.1	2.3	1.9	2.7
3+ inches	12.9	14.4	17.7	23.8	29.4
Total woody	15.9	17.7	21.5	26.8	33.7

important implications for managing biodiversity and nutrient potentials. Based on a survey of coarse woody debris knowledge (Harmon and others 1986) quantities of downed woody material are considerably higher in Cascade Mountains and coastal forests than in the Northern Rocky Mountains. Loadings of coarse woody debris ranged from 60 to 300 tons/acre (130 to 670 t/ha) in Sitka spruce and western hemlock of coastal British Columbia and from 60 to 240 tons/acre (130 to 540 t/ha) in the Douglas-fir type of the Cascade Mountains.

Postfire Plant Communities

Coast Douglas-fir and Douglas-fir/Hardwoods

Pre-1900 Succession—These humid maritime forests are extensive at low and middle elevations west of the crest of the Cascades and British Columbia Coast Range from northern California to southern British Columbia. Fire history studies (see Agee 1993) indicate that mixed fire regimes were common in the Douglas-fir type south from west-central Oregon; drier areas of the Douglas-fir type farther north; and in the Douglas-fir/hardwood types of northwestern California and southwestern Oregon (Wills and Stuart 1994) (fig. 5-5). Conversely, the cooler, wetter, more northerly portions of the coast Douglas-fir type tended to be associated with stand-replacing fire regimes. Mixed fire regimes were probably also associated with some areas of the redwood type, perhaps on steep terrain and in areas relatively remote from Native American use.

Mixed fire regimes favored development of stands dominated by large old, fire-resistant trees, such as coast Douglas-fir—and, where present, redwood. These regimes were characterized by patchy nonuniform burning (Morrison and Swanson 1990; Teensma 1987; Wills and Stuart 1994). Overall, most of the fire-susceptible trees (notably western hemlock) were killed while many of the resistant trees survived. Occasional nonlethal understory fires and stand-replacement fires also occurred. Effects of burning included removing understory conifers and ladder fuels, preventing successional replacement by shade-tolerant trees, and creating openings of all sizes that allowed regeneration of seral undergrowth (including berry-producing shrubs), hardwood trees (such as red alder, bigleaf maple, bitter cherry), Douglas-fir and a few other seral conifers (such as western white pine and shore pine).

In northwestern California and southwestern Oregon, dry Douglas-fir/hardwood types burned rather frequently and supported a variety of seral shrubs, including *Ceanothus* spp. as well as hardwood trees that typically resprout after fire—including tan oak, madrone, canyon live oak, and chinquapin (Agee 1993; Husari and Hawk 1993; Wills and Stuart 1994). A remarkable feature of the mixed fire regimes in dry sites was the prevalence of large (>6 feet in diameter), old Douglas-fir that had survived numerous fires. Although forests associated with this fire regime type are extensive and important for a variety of ecological values, little is known about landscape patterns and successional patterns associated with the presettlement fires. The existence of this mixed fire regime as a widespread type was documented only recently (Agee 1993; Means 1980; Morrison and Swanson 1990).

Post-1900 Succession—Better knowledge of the ecological role and importance of fire in these mixed fire regimes is needed (Kauffman 1990). Although major ecological changes in these forests due to clearcutting and short-rotation forest management have been recognized, other significant changes have also occurred as a result of interruption of the mixed fire regime (Agee 1990). Effects of fire exclusion are

Figure 5-5—Coastal Douglas-fir in a dry area of northwestern Washington, with scars from two different historic fires.

conspicuous in dry site coastal Douglas-fir communities which burned rather frequently, for example, those in southwestern Oregon (Agee 1991; Means 1982) and in the Puget Sound lowlands (Agee and Dunwiddie 1984; Boyd 1986). Such effects include loss of early seral shrub species, advanced successional development, increased stand density, and increased mortality. In older logged areas it is common to find dense second growth Douglas-fir and hemlock that are stagnating and succumbing to root rot, whereas the original stands consisted of large Douglas-fir in moderately open stands that had survived for several centuries in the presence of repeated fires.

Management Considerations—These forest types have high value for recreation, esthetics, endangered species habitat, and timber production. It is generally recognized that fire had an important role in creating these forests. A century ago Gifford Pinchot, Forester for the U.S. Department of Agriculture, argued for research into the ecological role of fire in these forests, but his call went largely unheeded (Pinchot 1899). Most fire research in these vegetation types has been related to burning of clearcuts (Kauffman 1990), and provides knowledge that is of limited value for modern ecological application. Other ecological aspects of these forests have been studied in detail at the H. J. Andrews Experimental Forest in west-central Oregon (Franklin and Others 1981; Teensma 1987) and in some redwood ecosystems (Zinke 1977). Major forest preservation efforts have focused on these forests, but ironically have generally accepted the continued exclusion of fire as consistent with maintenance of ecological values.

California Red Fir and Sierra/Cascade Lodgepole Pine

Pre-1900 Succession—These are high elevation forests characteristic of the southern Cascades, Klamath Mountains (in northwestern California), and the Sierra Nevada. Fire history studies conducted thus far suggest a variable or mixed fire regime.

The following summary of fire history in California red fir forests is paraphrased from Husari and Hawk (1993) with additional information from Taylor (1993). Estimates of natural fire frequency in California red fir range from 21 to 65 years (Taylor 1993; Taylor and Halpern 1991). Analysis of lightning fire occurrence in Yosemite National Park shows that, on a per acre basis, the California red fir forest type there experiences more lightning ignitions than any other vegetation type, but the fires are mostly small (van Wagtendonk 1986). This is probably attributable to lower overall productivity on these sites, a compact duff layer, a short fire season, and generally cool, moist conditions. The California red fir forest shows structural evidence of a combination of large intense fires, small fires and fires of mixed severity (Taylor 1993). This variety of fire characteristics has led to landscape diversity in California red fir forests (Agee 1989).

Agee (1993) and Chappell and Agee (1996) describe variable and mixed fire regimes in lodgepole pine forests of southern Oregon. Average fire intervals are estimated to be 60 to 80 years. Mountain pine beetle epidemics can result in an accumulation of heavy fuels, which in turn supports stand-replacement fire. Stands with moderate quantities of older downed logs and open space between tree canopies often experience smoldering fires ("cigarette burns") that spread primarily through decayed logs on the forest floor. Southward, in the California red fir-lodgepole pine-mixed conifer subalpine forest of California, small patchy fires are the norm (Parsons 1980). This is due in part to an abundance of broken topography, rock, bare mineral soil, and sparse fuels that hamper fire spread.

Post-1900 Succession—Suppression of the mixed, patchy fires in these high-elevation forests may eventually result in a landscape mosaic consisting largely of contiguous old stands with comparatively heavy loadings of dead trees (standing and fallen) and canopy fuels. This is probably the basis for Husari and Hawk's (1993) projection that in the future these forests will be characterized by infrequent high-intensity, stand-replacing fires.

Management Considerations—Intensive tree harvesting can be used to break up continuous heavy fuel loadings that might result in large, severe fires. Laacke and Fiske (1983) state that most silvicultural treatments have been designed to produce even-aged stands. Clearcutting offers the opportunity to control the spread of dwarf mistletoe and root disease, which are often a concern in this type. Recent interest in perpetual retention of some tree cover on high-elevation forest sites and in designing treatments consistent with historical disturbances will probably encourage attempts at uneven-aged silviculture or retention shelterwood. California red fir is a shade-tolerant species, which conceptually makes it appropriate to consider for uneven-aged silvicultural systems. Concern about root disease and dwarf mistletoe has worked against consideration of uneven-aged systems, but use of underburning might serve some function in controlling these pathogens (Koonce and Roth 1980; Petersen and Mohr 1985). Fire history and experience burning in white fir types (Petersen and Mohr 1985; Weatherspoon 1990) suggest that understory burning might be useful in some California red fir/lodgepole forests.

Interior Douglas-fir, Larch, Rocky Mountain Lodgepole Pine

Pre-1900 Succession—A broad range of mid-elevation mountain forests dominated by interior Douglas-fir, western larch, or Rocky Mountain lodgepole pine were characterized by mixed fire regimes. These occurred from central British Columbia (Strang and Parminter 1980) and Jasper National Park, Alberta (Tande 1979), southward at least to western Wyoming (Arno 1981; Loope and Gruell 1973). They were abundant and diverse in western and central Montana (Arno 1980; Barrett and others 1991, Arno and Gruell 1983). Mixed fire regimes allowed an open overstory of mature Douglas-fir and larch to survive many fires. Small trees and associated less fire-resistant species were heavily thinned by moderate-intensity burning. Additionally, some nonlethal underburns occurred in lodgepole pine stands having light fuels. Occasional stand-replacing fires were also part of the mixture making up this fire regime.

Effects of these variable fires often included maintaining a fine grained forest community mosaic on much of the landscape—as is illustrated by maps in Tande (1979), Barrett and others (1991), and Arno and others (1993) (fig. 5-6). Elements of this mosaic were small stands dominated by various age structures of seral coniferous species and seral hardwoods such as Scouler willow and aspen. Some stands experienced nonlethal underburns that maintained open understories by killing saplings and fire sensitive species. Others experienced patchy fire mortality that gave rise to patchy tree regeneration including seral species. Occasional large stand-replacement fires may have

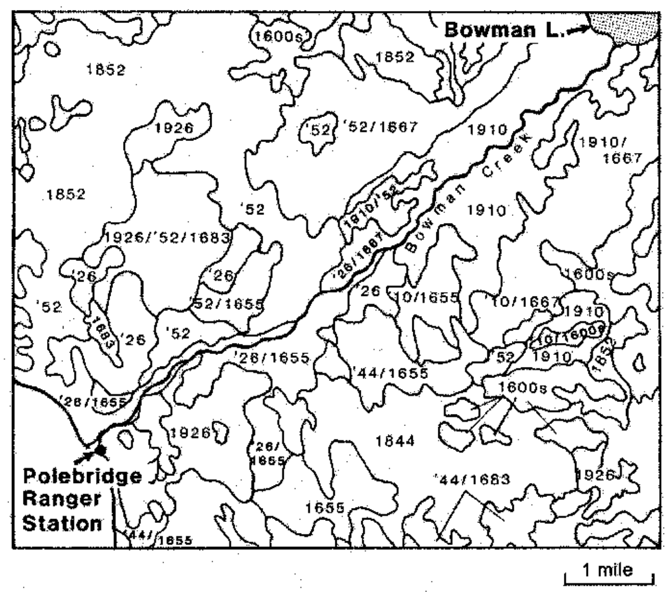

Figure 5-6—This mosaic of age-classes resulted from a mixed-severity fire regime in Glacier National Park, Montana. Dates indicate fire years that resulted in establishment of seral western larch and lodgepole pine age classes. In addition, some low-intensity underburns were detected in fire scars, but they did not thin the stand enough to allow new age classes to become established (from Barrett and others 1991).

reduced the spatial diversity, but the varying distribution of seed sources and sprouting shrubs in the preburn mosaic probably enhanced variability in postburn vegetation. A fire effect near the lower forest boundary was to maintain seral grasslands, shrublands, and aspen groves by periodically removing most of the invading young Douglas-fir or lodgepole pine (Arno and Gruell 1986; Patten 1963; Strang and Parminter 1980).

Post-1900 Succession—With a reduction in fire activity due to livestock grazing (removing fine fuels) and fire exclusion policies, young conifer stands have invaded former grasslands within or below the forest zone. The trees are densely stocked and subject to extreme drought stress. They often have poor vigor and are susceptible to western spruce budworm or other insect or disease attacks and to stand replacing fires. Productivity of seral herbs, shrubs, and aspen declines dramatically in the continuing absence of fire.

Stands within the forest zone may have undergone significant changes in recent decades. As a result of fire exclusion, the trees in most stands within the landscape mosaic have become older, and often have a buildup of down woody or ladder fuels. Recent wildfires have burned as larger stand-replacement fires than those detected in fire history studies (Arno and others 1993; Barrett and others 1991).

Management Considerations—Fire exclusion may move these communities toward a long-interval stand-replacement fire regime. This could decrease vegetation diversity on the landscape and may reduce values for wildlife habitat, watershed protection, and esthetics. Numerous alternatives exist for simulating more natural landscape structure and disturbance regimes using silvicultural cuttings and prescribed fire (Arno and Fischer 1995; Arno and others 2000; Brown 1989; Gruell and others 1986).

Whitebark Pine

Pre-1900 Succession—The whitebark pine type occurs near the highest elevations of forest growth from southern British Columbia south to western Wyoming in the Rocky Mountains and along the Cascades to northern California. In the drier mountain ranges and on rocky sites with sparse fuel, whitebark pine is characterized by a mixed fire regime, whereas in moist and more productive areas it is perpetuated by a stand-replacement fire regime. The mixed fire regime has been noted in whitebark pine stands in the Selway-Bitterroot Wilderness and nearby areas of central Idaho and western Montana (Barrett and Arno 1991; Murray 1996). Prior to 1900 these forests experienced a range of fire intensities from nonlethal underburns to large (but usually patchy) stand-replacement fires. Rugged terrain, including extensive rocklands and cool-moist north slopes hampered the spread of fires and resulted in a variable burn pattern.

Whitebark pine is a seral species on all but the harshest sites and is replaced successionally by subalpine fir, Engelmann spruce, or mountain hemlock (fig. 5-7). These species were readily killed in low-intensity fires, whereas whitebark pine often survived. Epidemics of mountain pine beetle periodically killed many of the older whitebark pines. Fuels created by beetle kills and successional ladder fuels contributed to patchy torching or stand-replacement burning. Underburns had a thinning effect that removed many of the competing fir, while more intense fires created open areas favorable for whitebark pine regeneration. This species' seeds are harvested and cached often in burned areas by Clarks Nutcrackers (*Nucifraga columbiana*). Many seed caches are not retrieved, so pines regenerate (Tomback 1982). Whitebark pine is hardier than its competitors and functions as a pioneer species on high-elevation burns.

Post-1900 Succession—Fire exclusion may have been particularly effective in many areas of the whitebark pine type where rugged terrain provides natural fire breaks. To cover large areas of the type, which is restricted to high ridges, fires had to burn across broad valleys. Many of the latter are now under forest management, agriculture, suburban or other development that prevents fire spread. Large wilderness areas and National Parks sometimes allow wildland fires with limited suppression as a substitute for natural fires. However, even the most successful natural fire program in a large wilderness has had a lower level of success in returning fire to whitebark pine than to other forest types (Brown and others 1994, 1995). Major fires in the whitebark pine type are confined largely to late summer in especially dry years, when prescribed natural fires usually are not allowed.

To complicate matters, white pine blister rust (*Cronartium ribicola*), an introduced disease, is killing cone-bearing limbs and entire trees of whitebark pine in about half of the species' natural range (Arno and Hoff 1990). A small percentage of the trees has some resistance, but without ample sites upon which to regenerate, resistant strains cannot develop and multiply.

Management Considerations—Fires may be more critical for whitebark pine's survival now that blister rust has inflicted heavy mortality across large areas of the Northwestern United States. Evidence from mortality observations (Kendall and Arno 1990), permanent plots (Keane and Arno 1993), and ecological process modeling (Keane and others 1990b) suggests that unless active management is carried out on a landscape scale, whitebark pine will continue to decline in a large part of its range and may virtually

Figure 5-7—Whitebark pine regenerating on a 26-year-old burn (foreground). In contrast, the old, unburned forest (background) consists of subalpine fir, Engelmann spruce, and dead whitebark pine snags.

disappear in some areas. Use of wildland fires in wilderness, prescribed fires, release cuttings to favor the pine over its competitors, and aiding the propagation of natural rust resistance are the most obvious alternatives. These measures are now being tested at several sites in Idaho and Montana by Robert Keane, Rocky Mountain Research Station, Missoula, Montana.

Ponderosa Pine

Pre-1900 Succession—Some ponderosa pine forests were historically characterized by mixed fire regimes, although the extent and ecological relationships of these mixed regimes are yet to be determined. It appears that mixed regimes were commonly associated with ponderosa pine growing east of the Continental Divide, and also with some forests west of the Divide, especially those on steep slopes and on relatively moist sites. The most compelling evidence for a large area of mixed fire regime comes from the Black Hills of South Dakota (Brown and Sieg 1996; Gartner and Thompson 1973; Shinneman and Baker 1997) and the Front Range of the Rocky Mountains in Colorado (Kaufmann 1998; Laven and others 1980). Many of the ponderosa pine stands in the Black Hills and nearby areas of northeastern Wyoming and southeastern Montana develop dense patches of pine regeneration after fire, which become thickets of small stagnant trees, susceptible to stand-replacing fire. Intervening areas with more open stocking presumably were more likely to underburn in the frequent fires of the presettlement era. Factors contributing to a mixed fire regime in ponderosa pine probably include relatively moist sites that tended to produce pine thickets soon after a fire, areas frequently exposed to high winds during the burning season, steep topography, and stands killed by bark beetle epidemics.

Post-1900 Succession—In the Black Hills, northeastern Wyoming, and southeastern and central Montana, fire exclusion coupled with livestock grazing has resulted in an expansion of ponderosa pine from its presettlement habitat on rocky ridges and other poor sites into the adjacent grassy plains. This invasion may involve over a million acres (Gartner and Thompson 1973; Gruell 1983), but no data are available. The stands on the ridges have also thickened with ingrowth of younger trees. Dense stands greatly reduce production of native grasses and forbs and diminished forage values for livestock and wildlife (Gartner and Thompson 1973). Thousands of homes are now located in these dense pine stands, and many are threatened annually by wildfires. Nearly 50 homes burned in a central Montana wildfire in 1984, and over

100 burned in 1991 in ponderosa pine woodlands near Spokane, Washington.

Management Considerations—The dense stands of small ponderosa pines that have expanded into grasslands in the upper Great Plains are at high risk of severe wildfire and have diminished forage values. Breaking up the continuity of these stands using silvicultural cuttings and prescribed fire treatments would allow more effective control of fire. It may be possible to thin some of these stands commercially while retaining the largest and healthiest trees. Prescribed burning in conjunction with cutting could reduce fuel loadings and stimulate forage (Gartner and Thompson 1973; Kilgore and Curtis 1987). Initial burning should be done with care not to injure leave trees (see discussion of ponderosa pine in the section "Understory Fire Regimes").

There may be objections to converting dense stands to open-growing ones, especially doing so on a landscape scale (Shinneman and Baker 1997). Patchy even-aged cuttings and prescribed burns are one alternative that could be used to simulate historical mixed regime patterns (Franklin and Forman 1987). Allowing dense forests on public lands to be killed by insect epidemics that encourage severe wildfires may be the eventual result of inaction. This latter alternative could be made less hazardous to adjacent lands if fuels management were carried out around the borders of such areas (Scott 1998).

Stand-Replacement Fire Regimes

Major Vegetation Types

Major forest types include coast Douglas-fir, true fir/hemlock, interior true fir/Douglas-fir/larch, Rocky Mountain lodgepole pine, white pine/western redcedar/hemlock, spruce/fir/whitebark pine, and aspen. Several minor forest types are also characterized by stand-replacement fires. These forests are widespread in wetter forest regions of the Northwestern United States and Western Canada and in subalpine forests associated with the major mountain ranges. Some of the same compositional types are also characterized by mixed fire regimes in large portions of their distributions. Differences in regional climate, fuels, and local topography can influence the resulting fire regime (for example, see Barrett and others 1991).

Fire Regime Characteristics

Stand-replacing fires kill most overstory trees, although the pattern of these fires on the landscape varies with topography, fuels, and burning conditions. Sometimes extensive areas burn uniformly in stand-replacing fire events, especially in wind-driven crown fires (Anderson 1968). However, a major proportion of stand-replacement is caused by lethal surface fire, as was the case with much of the 1988 Yellowstone Area fires. Lethal surface fire was responsible for about 60 percent (versus 40 percent in crown fire) of the stand-replacement burning in the Selway-Bitterroot Wilderness under the prescribed natural fire program (Brown and others 1995). Under different conditions, a complex landscape mosaic of replacement burning from crown fire and lethal surface fire is interwoven with areas of lighter burning or no burning. For instance, patchy burning patterns may be accentuated by rugged mountainous topography containing contrasting site types, microclimates, and vegetation. These mosaic elements represent diverse burning environments and the result is that stand-replacement burning is restricted to certain landscape elements. For example, in one area of northern Idaho, stand-replacement fires were associated with a mid-slope "thermal belt" on southern and western exposures while other slopes tended to burn in mixed severity fires (Arno and Davis 1980). Superimposed on the site mosaic is a fuels mosaic linked to the pattern of past fires. On gentle topography and more uniform landscapes, such as high plateaus, stand-replacement fires tend to be more uniform or at least to burn in large-scale patches.

Stand-replacement fires generally occur at long average intervals (table 5-1), ranging from about 70 years in some lower elevation Rocky Mountain lodgepole pine forests subject to extreme winds, to 300 to 400 years in some inland subalpine types, and over 500 years in some moist coastal mountain forests. Often the range of actual intervals is broad since the fires themselves depend on combinations of chance factors such as drought, ignitions, and high winds. Such combinations occur sporadically. In coastal forest types having long fire intervals, such as coast Douglas-fir and true fir-mountain hemlock, it appears that exceptional drought and perhaps an unusual abundance of lightning ignitions were linked to major fires (Agee 1993). The great length of intervals and findings of substantial climatic changes during the last few thousand years suggest that fire intervals varied with climatic changes (Johnson and others 1994).

The irregular timing of stand-replacement fire is heightened in several forest types by their propensity to support double or triple burns in the aftermath of an initial fire. For instance, the Yacolt fire (1902) of southwestern Washington and the Tillamook fire (1933) of northwestern Oregon reburned numerous times (Gray and Franklin 1997; Pyne 1982). A history of occasional replacement fires followed by severe reburns also is common in the Clearwater drainage of northern Idaho (Barrett 1982; Wellner 1970).

In the Rocky Mountain lodgepole pine type, fuel buildup is an important factor in length of fire intervals (Brown 1975; Romme 1982). Mean fire intervals in this type range from a low of about 70 years in productive lower elevation lodgepole pine forests in high wind environments on the east slope of the Rockies—for example, at Waterton Lakes National Park, Alberta; Glacier National Park, Montana; and perhaps in the Red Lodge area northeast of Yellowstone National Park. At the other extreme, mean fire intervals on unproductive sites such as the high-elevation rhyolite plateaus in Yellowstone National Park are 300 to 400 years (Millspaugh and Whitlock 1995; Romme 1982). The majority of studies in this type has found mean fire intervals between 100 and 250 years (Agee 1993; S. W. Barrett 1994; Hawks 1980; Kilgore 1987).

Fuels

Unlike understory and mixed fire regimes, fuels play a critical role in limiting the spread of fire in stand-replacement fire regimes. Accumulation of duff and down woody fuels increases the persistence of burning. This is important for keeping fire smoldering on a site until a wind event occurs (Brown and See 1981). Typically a certain level of fuel is required to allow fire to spread. This may be the result of dead and down fuels—from insect epidemics, windstorms, or a previous fire—or of extensive ladder fuels (fig. 5-8). In contrast, stands with few down or ladder fuels often fail to support fire (Brown 1975; Despain 1990). In lodgepole pine, dead and down woody fuel loadings of 15 to 20 tons/acre (34 to 45 t/ha) are generally near the lower threshold of what will support a stand-replacement through moderate-intensity surface fire (Fischer 1981). Ladder fuels and heavier loadings of down and dead woody fuels contribute to torching, and with winds a running crown fire may evolve.

In cover types supporting large trees such as Douglas-fir/hemlock and western white pine, large woody fuel loadings typically are 40 to 50 tons/acre (90 to 110 t/ha) and duff about 30 tons/acre (67 t/ha) (Keane and others 1997). In smaller tree cover types such as lodgepole pine and spruce/fir, large woody fuels typically are 15 to 20 tons/acre and duff about 15 to 30 tons/acre. However, the range in loadings may be considerably greater as reported for Kananaskis Provincial Park, Alberta (Hawkes 1979), where downed woody fuels ranged from 4 to 63 tons/acre (9 to 141 t/ha) and duff from 8 to 58 tons/acre (18 to 130 t/ha).

Figure 5-8—Downfall of mountain pine beetle killed lodgepole pine results in an accumulation of large downed woody fuels that increases the likelihood of stand-replacement fire.

Postfire Plant Communities

Coast Douglas-fir

Pre-1900 Succession—These humid maritime forests are extensive at lower and middle elevations west of the Cascades and British Columbia Coast Range. The cooler, wetter, and more northerly portions of the coastal Douglas-fir type (generally associated with the mountains of western Washington and southwestern British Columbia) burned in stand-replacement fires at long intervals, averaging 200 to several hundred years (Agee 1993). The range of pre-1900 fire intervals on a given site is unknown because in most cases only the most recent interval can be calculated due to decay of the previous stand. Long and others (1998) described fire intervals over the last 9,000 years, and Impara (1997) reports on the spatial patterns of historical fires in the Oregon Coast Range.

Western hemlock is the potential climax dominant tree in most of this type, but seral Douglas-fir, which arose after replacement fires during the last several hundred years, is the actual dominant. The greater size and longevity of Douglas-fir allows it to persist in considerable quantities for 700 to 1,000 years between major stand-opening disturbances such as fire or severe blowdowns (Agee 1993). Scattered individual Douglas-fir survived fires and served as seed sources in the burn. Seeds of this species may also survive and mature in the crowns of some trees whose foliage was killed (but not consumed) by a late-summer fire. The seeds are also wind-dispersed from unburned stands. Douglas-fir seedlings grow readily on burned seedbeds and outcompete other conifers in the postburn environment.

Often red alder becomes abundant and temporarily outgrows Douglas-fir in a recent burn. However, the fir grows up beneath and displaces alder within a few decades, benefitting from soil nitrogen fixed by symbiotic organisms associated with alder roots. Numerous other seral conifers (western white pine, shore pine, grand fir, and Sitka spruce) and hardwood species (bigleaf maple, mountain ash, cascara, and others), as well as seral shrubs (salmonberry, huckleberries) and herbaceous plants, appear in the postburn environment, greatly enriching the biological diversity of these forests (Fonda and Bliss 1969; Franklin and Dyrness 1973; Hemstrom and Franklin 1982; Huff 1984; Yamaguchi 1986).

Post-1900 Succession—Due to the great length of natural fire intervals it would seem unlikely that significant successional changes have occurred in most of these forests as a result of attempts to exclude fire during this century. Large areas of these forests have been clearcut in recent decades, sometimes followed by broadcast burning. This has given rise to large areas of early seral communities dominated by native flora, often with planted Douglas-fir, which might offset a shortage of early seral communities resulting from natural fires. However, natural burns and clearcuts differ ecologically, for example, in seedbed preparation, in providing residual large woody debris, and in having an overstory of dead trees (Kauffman 1990). Hansen and others (1991) point out that young communities arising after fires are rich in structural complexity and in species composition, but are the rarest successional state, much rarer in today's landscapes than is old growth. Although millions of acres have been set aside as reserves for old growth Douglas-fir, there are no measures for perpetuating these communities through the use of prescribed fire, and if present fire suppression policies succeed, young postfire communities will continue to be rare.

Management Considerations—Until the 1990s these forests were usually managed by clearcutting, site preparation, and growing even-aged stands at rotations (50 to 100 years) much shorter than those of presettlement fire intervals. This management approach failed to consider perpetuation of many ecological functions in these forests (Kauffman 1990). The need to develop more ecologically-based management strategies gave rise to concepts of "new forestry" treatments (Franklin and others 1986; Gillis 1990; Hopwood 1991). Most of these leave patches of overstory and understory trees after harvesting to increase structural diversity of the new stand and the "biological legacies" of large woody debris. In this structural respect, these treated stands are simulating the early seral community following a natural fire. However, many of the proposed treatments avoid actually utilizing fire due to a desire to limit smoke, increase woody residues on the site, avoid the operational difficulties in burning, and reduce treatment costs (Means and others 1996). Douglas-fir can be regenerated in heavily logged areas without burning. It is apparently assumed that the other effects of fire are expendable, for example, in soil nutrition and maintenance of diverse fire-dependent undergrowth species (Agee 1993; Kauffman 1990). However, there is little scientific basis for such an assumption. A more rigorous evaluation of the consequences of various management alternatives, including use and avoidance of burning, is needed for this ecological type.

Coastal True Fir-Mountain Hemlock

Pre-1900 Succession—These high-elevation maritime forests are found along and west of the crest of the Cascades (north of Crater Lake, Oregon) and the British Columbia Coast Range. Principal tree species are Pacific silver fir, mountain hemlock, western hemlock, and noble fir. This type is a cooler and wetter environment than the coastal Douglas-fir type that borders it at lower elevations and on warmer aspects. Annual precipitation is commonly >100 inches and a deep snowpack accumulates in winter and persists into early summer.

The principal tree species are fire-sensitive and seldom survive surface fires. Thus, fires were typically of the stand-replacement type, although scattered Douglas-fir found in this type often survived. Fire return intervals were evidently between about 125 and 600 years (Agee 1993). Shorter intervals (<200 years) were associated with drier environments where the type is confined to north-facing slopes and is surrounded by drier types. Fires in this type usually occur under conditions of severe summer drought accompanied by strong east (foehn) winds. Major blowdowns also initiate regeneration cycles and can contribute large amounts of fuels in this type.

Postfire stands often go through a shrub-dominated stage—commonly including early seral communities of mountain huckleberry (Agee 1993; Franklin and Dyrness 1973). A variety of early seral conifers becomes established in burns (noble fir, Douglas-fir, subalpine fir, Alaska-cedar, lodgepole pine, and western white pine). Eventually the shade-tolerant Pacific silver fir, mountain hemlock, and western hemlock become dominant.

Post-1900 Succession—The long intervals between stand-replacement fires and the remote location of much of this forest type suggest that fire suppression would have had a minor effect upon it, but no detailed evaluation of this question has been made.

Management Considerations—Few fire management programs that plan for wildland fire use exist in the National Parks and wilderness areas in this forest type. Where fires occurred in the past, they often resulted in shrubfield/open conifer stands. These persistent open, “old burn” communities have been an important component of wildlife habitat and natural diversity. Use of prescribed fire is limited in this type because fire generally burns only under extreme weather conditions. Because the principal tree species are readily killed by fire, any burning in standing trees increases loadings of dead and down fuels. Mechanical fuel manipulation (tree harvesting or removal of dead woody fuel) is necessary for creating fuel reduction zones to contain wildfires (Agee 1993).

Interior True Fir–Douglas-fir–Western Larch

Pre-1900 Succession—This is a diverse group of forests that have a stand-replacement fire regime and are dispersed throughout much of the Interior West, usually at middle elevations in the mountains. Principal tree species are white fir and grand fir (potential climax), interior Douglas-fir (seral or climax), and larch, lodgepole pine, and aspen (early seral associates). These forests commonly develop dense stands with accumulations of ladder fuels and they often occupy steep slopes on cool aspects. The forest floor fuels are primarily a compact duff layer that does not support low intensity surface fires. However, when down woody or ladder fuels accumulate and severe burning conditions arise, they can support a stand-replacing surface or crown fire. Such fires occurred at intervals averaging between 70 and 200 years. Similar compositional types in other geographic areas or on different topographic situations are associated with mixed fire regimes. The relative amounts of these types in mixed and stand-replacement fire regimes is unknown (Brown and others 1994). Also, the factors that determine whether one of these forests will have a mixed or stand-replacement regime is not known, but lack of receptiveness of surface fuels to burning, characteristically dense stands, steep slopes, and frequent strong winds probably favor the stand-replacement fire regime. For example, one area in the stand-replacement regime is on the eastern slope of the Continental Divide in Montana where dense Douglas-fir stands develop in an environment featuring severe winds (Gruell 1983).

Relatively frequent stand-replacement fires kept much of the landscape in open areas (seral grasslands or shrublands) and favored seral shrub species (such as serviceberry, willow, and bitterbrush) and aspen. Such plant communities are important forage for wildlife.

Post-1900 Succession—Photocomparison and fire history studies suggest that fire exclusion has allowed a greater proportion of these inland forests on the landscape to develop as dense stands. The spatial continuity of these stands may allow insect and disease epidemics and stand-replacement fires to become larger than in the past (Arno and Brown 1991; Byler and Zimmer-Grove 1991; Gruell 1983) (fig. 5-1). At the same time seral grassland species, shrubs, aspen, and seral conifers are being replaced by thickets of shade-tolerant conifers.

Management Considerations—This major forest cover type is divided into mixed and stand-replacement fire regimes, but the environmental characteristics linked to each fire regime type are poorly known. Knowledge of these characteristics would help land managers determine where stand-replacement fire is probable, which might help in setting priorities for management of fuels to confine potentially severe wildland fires. It should be possible to reduce frequency and extent of stand-replacement fires using a variety of fuel-reduction treatments. Prescribed fire in activity fuels (slash) can be useful in fuels reduction and in obtaining other desirable fire effects such as stimulation of wildlife forage. It may be possible to use fire for fuels reduction and habitat improvement for ungulates by felling a few trees per acre to create enough fine slash fuels to allow a prescribed fire to move through the stand. This is promising in Douglas-fir stands because of this species resistance to low intensity fire.

Rocky Mountain Lodgepole Pine

Pre-1900 Succession—This is a major type at middle to high elevations in the more continental mountain climatic areas of the Inland West, from the Yukon Territory, Canada, to southern Colorado. In parts of this geographic distribution, lodgepole pine forests burned in a mixed fire regime, primarily where fine surface fuels and dry climate allowed lower intensity fires to occur. Much of the lodgepole pine type, however, is resistant to burning except when there is an accumulation of down woody, ladder, and crown fuels. When fuel loadings are sufficient to support fire, it becomes a stand-replacing surface or crown fire.

Brown (1975) illustrated how fuel loadings are indirectly linked to stand age (fig. 5-9). Young dense stands containing ladder fuels of associated spruce and fir and accumulated downfall from a former, beetle-killed or fire-killed overstory have high potential to support a stand-replacement fire. Conversely, young pole-size stands of pure lodgepole pine (with sparse lower limbs) arising after a burn that removed most large fuels, have low potential to support fire. When a lodgepole pine stand becomes mature or overmature, tree growth and vigor declines markedly, and likelihood of a mountain pine beetle epidemic increases. Such epidemics kill many trees that begin falling in a few years, and within 10 to 15 years large amounts of dead woody fuels accumulate that greatly adds to the potential of stand-replacement fire. Dwarf mistletoe also builds up with stand age and adds highly flammable witches-brooms to the tree crowns. Stand-replacement fire destroys the dwarf mistletoe, except in surviving patches, and removes potential for mountain pine beetle until a mature stand has again developed.

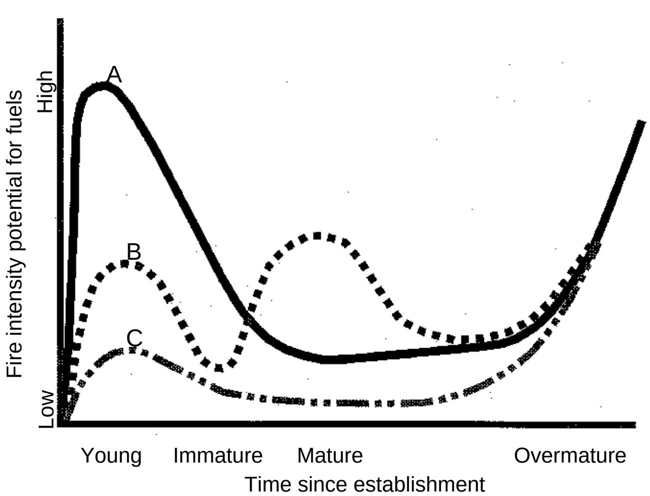

Figure 5-9—Fuel cycles and fire intensity potential in a lodgepole pine stand-replacement fire regime (Brown 1975).

Post-1900 Succession—Some studies indicate that attempts to exclude fire have had relatively little effect in this fire regime (Barrett and others 1991; Johnson and others 1990; Kilgore 1987). Certainly numerous fires have been successfully suppressed while quite small or while they were in adjacent, different fire regime types; but this is presumably offset by additional ignitions from large numbers of human-caused fires. The possibility exists that suppression could have appreciable effects in some geographic areas, especially where units of this type are small, isolated, and surrounded by other kinds of forest or vegetation where fires have been largely excluded. In the southern Canadian Rockies a decline in fire frequency in largely stand-replacement fire regime types was attributed at least partially to prevention and suppression of human-caused fires (Achuff and others 1996). However, in some geographic areas, the proportion of area burned by lethal seveity has increased (fig. 5-1).

Management Considerations—As illustrated by the political uproar regarding the Yellowstone Area fires of 1988, this forest type represents a challenge for fire management (Christensen and others 1989; Wakimoto 1989). Ecologists, land managers, and many environmental activists recognize the importance and inevitability of large stand-replacement fires. In contrast, such fires are often viewed as an unnecessary inconvenience or as a disaster by those who are unaware of the natural role of wildland fire.

Fires are critical to maintenance of biological diversity in this type. Many early seral species, including herbs, shrubs, and aspen, depend on occasional fires to remain as components of the lodgepole pine type (Habeck and Mutch 1973; Kay 1993). Black-backed Woodpeckers, many invertebrates, herbivores, small mammals, birds, and even some aquatic organisms depend upon fires for creation of seral communities, snag patches, and beneficial nutrient cycling (Agee 1993; Despain 1990).

Stand-replacement fire regimes in lodgepole pine forests can be influenced by management actions. For example, fuel breaks can be developed near critical property boundaries and to protect resorts and other facilities (Anderson and Brown 1988; Kalabokidis and Omi 1998; Schmidt and Wakimoto 1988). Wildland fire use programs coupled with prescribed stand-replacement fires could help develop landscape fuel mosaics that limit the ultimate size of wildfires (Weber and Taylor 1992; Zimmerman and others 1990).

Clearcutting and broadcast burning have long been a common timber management treatment in this type. If fine fuel consumption is complete, most of the seed source in the slash will be destroyed. Seeds from open-coned pines at the edge of units are often wind-distributed about 200 feet into clearcuts in sufficient quantities for forest regeneration (Lotan and Critchfield 1990). Leaving some cone-bearing trees standing throughout the burn can provide seed source as well as light shade and snags for wildlife. In salvage logging of fire-killed stands, it is important to leave ample dead trees for a variety of wildlife species.

There is increasing interest in planning for patches of trees to survive broadcast burning, to provide structural diversity in the post-treatment community (Arno and Harrington 1998; Hardy and others 2000). Underburning in a lodgepole pine shelterwood cut, although difficult, may be possible if slash fuels are light and moved away from the base of leave trees.

Western White Pine-Cedar-Hemlock

Pre-1900 Succession—This forest type is centered in northern Idaho and the interior wet zone of southeastern British Columbia (Krajina 1965; Shiplett and Neuenschwander 1994). It also extends into northeastern Washington and northwestern Montana. It occupies a "climatic peninsula" (Daubenmire 1969) or inland extension of Pacific maritime climate between about 46 and 53 °N latitude. This is the only area of the Rocky Mountain system where western white pine, western redcedar, western hemlock, and numerous other maritime associates (trees and undergrowth) are found. In this area the inland maritime species form the dominant vegetation mixed with Inland Rocky Mountain species such as ponderosa pine, interior Douglas-fir, Engelmann spruce, western larch, and whitebark pine.

Traditionally this type has been considered to mainly represent a stand-replacement fire regime, but detailed fire history studies are few (Smith and Fischer 1997). The most characteristic, large, and influential fires were stand-replacing, and because of scarcity of information, the type is discussed only in this section. Nevertheless two recent studies (Barrett 1993; Zack and Morgan 1994) as well as earlier observations (Arno 1980; Arno and Davis 1980; Marshall 1928) indicate that substantial areas of the type were also exposed to a mixed fire regime. Evidence of underburning is often found in valley bottoms, on gentle slopes on dry aspects, and on ridgetops. Surviving trees were primarily western white pine, western larch, and large western redcedars. Underburns may have been important locally in perpetuating a dominance by large trees of these species. Conditions associated with underburning have not been described, but underburning has occurred in recent wildland fires in these forests (Brown and others 1994).

The stand-replacement fires characteristic of the western white pine-cedar-hemlock type occurred at intervals of 130 to 300 years associated with severe drought, which commonly occurred for a short period during mid- or late-summer. Because of the productive growing conditions, ladder fuels can develop rapidly in young stands. Moreover, large woody fuels are created in abundance by fires and by insect and disease epidemics. Two or three decades after a stand-replacement fire, most of the dead trees have fallen and become heavy down fuels. This, coupled with a dense stand of small trees and tall shrubs, may constitute a fuelbed that allows severe burning in a second fire, known as a "double burn" (Wellner 1970). When fire occurs under conditions of extreme drought accompanied by strong winds, stand-replacement fires are likely in many natural fuel configurations. If large accumulations of down woody fuel (>25 tons/acre) are present, stand-replacement fires can occur under extreme drought without strong winds or steep slopes (Fischer 1981).

Past stand-replacement fires allowed seral fire-dependent species to dominate most pre-1900 stands. The major early seral tree dominants were western white pine, western larch, and lodgepole pine, but were accompanied by lesser amounts of interior Douglas-fir, grand fir, Engelmann spruce, paper birch, and ponderosa pine. Including the everpresent western redcedar and western hemlock, these disturbance communities were diverse. Luxuriant seral shrub and herbaceous plant growth added to this diversity, as described by Larsen (1929), Daubenmire and Daubenmire (1968), and Cooper and others (1991). Occasional fires allowed a variety of seral shrubs to thrive, including redstem and evergreen ceanothus, currants, red elderberry, Scouler willow, serviceberry, mountain maple, American mountain-ash, bittercherry, and chokecherry. After a double burn, shrubfields would persist for decades (Barrett 1982; Wellner 1970). Shiplett and Neuenschwander (1994) classify five common successional scenarios in these forests. These are (1) a relatively rapid succession to the redcedar-hemlock climax, (2) a prolonged domination by a mix of seral tree species as a result of disturbances, (3) a shrubfield resulting from multiple burns, (4) a sere influenced by scattered larch relicts that survived fires, and (5) lodgepole pine dominance throughout fire cycles on less productive sites with relatively frequent burns.

Post-1900 Succession and Management Considerations—Clearcut logging has to a considerable extent replaced fire as the principal stand-replacing disturbance in this forest type. Often the logging is followed by broadcast burning or dozer scarification and piling of large woody residues, which are then burned. Like stand-replacement fire, clearcutting favors establishment of early seral tree species. Unlike fire, clearcutting does not leave a snag forest to moderate the microclimate and provide large quantities of woody debris and future fuels. Recently, environmental concerns have been raised about cumulative impacts of road building and logging, including soil disturbance, erosion, loss of water quality, aesthetic impacts, loss of wildlife habitat, and smoke production from prescribed fire. These concerns have encouraged substitution of partial cutting or of no cutting at all accompanied by fire suppression. These approaches contrast strongly with natural fire in their effects on vegetation and may result in epidemic forest mortality resulting from root diseases and bark beetles (Byler and Zimmer-Grove 1991).

A high percentage of western white pine has been killed in the last 50 years as a result of white pine blister rust, an introduced disease. White pine has a low level of natural rust resistance, and resistant genotypes are available for use in reforestation. However, this species requires fire or logging with site preparation to make sites available for regeneration and successful establishment.

Relatively rapid change is characteristic of the vegetation in this highly productive forest type. In the past, fire was the principal agent initiating new cycles of change. Heavy logging and site preparation was to a limited extent a replacement for fire, but had some undesirable impacts. In revising management to reduce those impacts it is important to consider strategies and treatments that provide the beneficial effects associated with fire in these ecosystems (Smith and Fischer 1997).

Spruce-Fir-Whitebark Pine

Pre-1900 Succession—This type makes up the highest elevations of forest growth in the Rocky Mountains and other interior ranges of Western North America from central British Columbia to central Oregon and western Wyoming. Southward in the Rocky Mountains to New Mexico, beyond the range of whitebark pine, the ecologically similar limber pine is often associated with the spruce and fir. In northern British Columbia, the high-elevation spruce-fir forest merges with the white spruce and black spruce boreal types discussed in chapter 3.

In drier regions of the Interior West and locally on drier topographic sites is a mixed severity fire regime characterized by an abundance of whitebark pine (see the section "Mixed Fire Regimes" in this chapter). In contrast, the spruce-fir-whitebark pine type has variable amounts of whitebark pine or none at all, but is characterized by stand-replacement fires generally at intervals of 100 to 400 years. For example, in the most detailed study of this type so far, in the Bob Marshall Wilderness Complex in Montana, Keane and others (1994) found mainly stand-replacement fires at intervals of 54 to 400+ years in 110 sample stands distributed across a 1.5-million-acre (607,000 ha) area. In the Southern Rocky Mountains, spruce is often the dominant subalpine forest cover and other major disturbances—spruce beetle epidemics, extensive snow avalanches, and areas of wind-thrown forest—interact with stand-replacement fires in complex temporal and spatial patterns (Baker and Veblen 1990; Veblen and others 1994). In the wettest spruce-fir microsites, such as naturally subirrigated basins, fire occurs rarely and is not the prevalent factor controlling successional cycles that it is in most of Western North America.

Pre-1900 fires added structural and compositional diversity to the spruce-fir-whitebark pine forest. Burned areas often remained unforested for extended periods due to the harsh microclimate (Arno and Hammerly 1984). In extreme cases regenerating conifers take on a shrublike (krummholz) form for 50 years or longer. Often whitebark pine is able to become established first in a high-elevation burn due to its superior climatic hardiness and its advantage of having seeds planted in small caches by the Clark's Nutcracker (Arno and Hoff 1990).

Post-1900 Succession—Little is known about possible human-induced changes in successional patterns throughout this high-elevation type. Logging has occurred in some sizeable areas of the type and has to a limited extent been a substitute for stand-replacement fire. In other areas fire suppression may have effectively reduced the landscape component made up of young postfire communities. For example, Gruell (1980) published many photographs taken at subalpine sites in northwestern Wyoming in the late 1800s and early 1900s and compared them with modern retakes. Most of these comparisons show that mature forest is noticeably more extensive today. Presumably the slow postfire recovery period resulted in large areas being unforested at any given time.

Whitebark pine has suffered a major setback since the early 1900s due to heavy mortality from mountain pine beetle and blister rust (Keane and Arno 1993). The introduced blister rust particularly reduces the amount of whitebark pine seed source available for regeneration. In some areas large outbreaks of spruce bark beetle and root rot in subalpine fir have also resulted in heavy loadings of large woody fuels, which will support future stand-replacement fires (Veblen and others 1994).

Management Considerations—In smaller wilderness areas and parks it is hard to plan for stand-replacement fires from lightning ignitions due to the problem of confining fires within area boundaries. Data presented by Brown and others (1994) suggest that maintaining natural fire cycles in these high-elevation forests is difficult because the forests only burn when fire danger elsewhere is unacceptably high as a result of extreme drought. Traditional timber harvesting coupled with broadcast burning is now less likely to occur because of environmental concerns about road building, watershed impacts, and obtaining prompt tree regeneration. Concerns regarding fire management options are generally similar to those expressed in high-elevation lodgepole pine types. In some cases prescribed fires might be used to maintain natural fire cycles. Cutting to provide slash fuels could allow prescribed burning to be done when wildfire hazard is moderately low.

Aspen

For discussions about aspen, see chapter 3, "Fire in Northern Ecosystems."

Regimes Where Fire is Rare

In some forest types, which collectively cover only a small fraction of the forested land in Western North America, fire is so unusual or exceptionally infrequent that it exerts little selective influence on vegetation development. Certain stands and topographic situations within the subalpine spruce-fir type seldom burn and their successional cycles are initiated primarily by beetle epidemics or windthrow. In British Columbia, Hawkes (1997) reported fire cycles of 800 to 2,000 years on cool wet sites occupied by the spruce-fir type. The only major forest type of Western North America where fire is not a primary disturbance agent is Sitka spruce-western hemlock. This type is generally confined to the wettest lowland sites along the immediate Pacific Ocean coastal strip and in alluvial bottoms of coastal valleys from southern Oregon to southern British Columbia. Northward, in central British Columbia and southeastern and south-central Alaska, it expands to cover most of the narrow, low elevation coastal forest zone (Arno and Hammerly 1984). Fires of appreciable extent are unusual in this type due to the prevalence of moist conditions year-round (Agee 1993). Windfall is a more prevalent disturbance creating new stands. On the Alaskan coast most fires are human-caused and occur during rare droughts (Noste 1969). Lightning fires are rare; nevertheless, the historic role of fire is not fully resolved. For example, Harris and Farr (1974) report four episodes of extensive fires in southeastern Alaska between about 1660 and 1830.

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Wildland Fire in Ecosystems

Effects of Fire on Soil and Water

Abstract

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This state-of-knowledge review about the effects of fire on soils and water can assist land and fire managers with information on the physical, chemical, and biological effects of fire needed to successfully conduct ecosystem management, and effectively inform others about the role and impacts of wildland fire. Chapter topics include the soil resource, soil physical properties and fire, soil chemistry effects, soil biology responses, the hydrologic cycle and water resources, water quality, aquatic biology, fire effects on wetland and riparian systems, fire effects models, and watershed rehabilitation.

Keywords: ecosystem, fire effects, fire regime, fire severity, soil, water, watersheds, rehabilitation, soil properties, hydrology, hydrologic cycle, soil chemistry, soil biology, fire effects models

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Cover photo—Left photo: Wildfire encroaching on a riparian area, Montana, 2002. (Photo courtesy of the Bureau of Land Management, National Interagency Fire Center, Image Portal); Right photo: BAER team member, Norm Ambos, Tonto National Forest, testing for water repellancy, Coon Creek Fire 2002, Sierra Ancha Experimental Forest, Arizona.

Wildland Fire in Ecosystems

Effects of Fire on Soil and Water

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Preface

In 1978, a national workshop on fire effects in Denver, Colorado provided the impetus for the "Effects of Wildland Fire on Ecosystems" series. Recognizing that knowledge of fire was needed for land management planning, state-of-the-knowledge reviews were produced that became known as the "Rainbow Series." The series consisted of six publications, each with a different colored cover (frequently referred to as the Rainbow series), describing the effects of fire on soil (Wells and others 1979), water (Tiedemann and others 1979), air (Sandberg and others 1979), flora (Lotan and others 1981), fauna (Lyon and others 1978), and fuels (Martin and others 1979).

The Rainbow Series proved popular in providing fire effects information for professionals, students, and others. Printed supplies eventually ran out, but knowledge of fire effects continued to grow. To meet the continuing demand for summaries of fire effects knowledge, the interagency National Wildfire Coordinating Group asked Forest Service research leaders to update and revise the series. To fulfill this request, a meeting for organizing the revision was held January 46, 1993 in Scottsdale, AZ. The series name was then changed to "The Rainbow Series." The five-volume series covers air, soil and water, fauna, flora and fuels, and cultural resources.

The Rainbow Series emphasizes principles and processes rather than serving as a summary of all that is known. However, it does provide a lot of useful information and sources for more detailed study of fire effects. The five volumes, taken together, provide a wealth of information and examples to advance understanding of basic concepts regarding fire effects in the United States and Canada. While this volume focuses on the United States and Canada, there are references to information and examples from elsewhere in the world (e.g. Australia, South Africa, Spain, Zimbabwe, and others) to support the statements made. As conceptual background, they provide technical support to fire and resource managers for carrying out interdisciplinary planning, which is essential to managing wildlands in an ecosystem context. Planners and managers will find the series helpful in many aspects of ecosystem-based management, but they also have the responsibility to seek out and synthesize the detailed information needed to resolve specific management questions.

— The Authors

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Contents

Summary

Fire is a natural disturbance that occurs in most terrestrial ecosystems. It is also a tool that has been used by humans to manage a wide range of natural ecosystems worldwide. As such, it can produce a spectrum of effects on soils, water, riparian biota, and wetland components of ecosystems. Fire scientists, land managers, and fire suppression personnel need to evaluate fire effects on these components, and balance the overall benefits and costs associated with the use of fire in ecosystem management. This publication has been written to provide up-to-date information on fire effects on ecosystem resources that can be used as a basis for planning and implementing fire management activities. It is a companion publication to the recently published book, *Fire's Effects on Ecosystems* by DeBano and others (1998).

In the late 1970s, the USDA Forest Service published a series of state-of-knowledge papers about fire effects on vegetation, soils, water, wildlife, and other ecosystem resources. These papers, collectively called "The Rainbow Series" because of their covers, were widely used by forest fire personnel. This publication updates both the Tiedemann and others (1979) paper on fire's effects on water and the Wells and others (1979) paper on soils.

This publication is divided into three major parts (A, B, C) and an introductory chapter that provide discussions of fire regimes, fire severity and intensity, and fire related disturbances. Part A describes the nature of the soil resource, its importance, characteristics and the responses of soils to fire and the relationship of these features to ecosystem functioning and sustainability. Part A is divided into three main chapters (2, 3, and 4) that describe specific fire effects on the physical, chemical, and biological properties of the soil, respectively. Likewise, Part B discusses the basic hydrologic processes that are affected by fire, including the hydrologic cycle, water quality, and aquatic biology. It also contains three chapters which specifically discuss the effect of fire on the hydrologic cycle, water quality, and aquatic biology in chapters 5, 6, and 7, respectively. Part C has five chapters that cover a wide range of related topics. Chapter 8 analyzes the effects of fire on the hydrology and nutrient cycling of wetland ecosystems along with management concerns. The use of models to describe heat transfer throughout the ecosystem and erosional response models to fire are discussed in chapter 9. Chapter 10 deals with important aspects of watershed rehabilitation and implementation of the Federal Burned Area Emergency Rehabilitation (BAER) program. Chapter 11 directs the fire specialists and managers to important information sources including data bases, Web sites, textbooks, journals, and other sources of fire effects information. A summary of the important highlights of the book are provide in chapter 12. Last, a glossary of fire terms is included in the appendix. The material provided in each chapter has been prepared by individuals having specific expertise in a particular subject.

This publication has been written as an information source text for personnel involved in fire suppression and management, planners, decisionmakers, land managers, public relations personnel, and technicians who routinely and occasionally are involved in fire suppression and using fire as a tool in ecosystem management. Because of widespread international interest in the previous and current "Rainbow Series" publications, the International System of Units (Systeme International d'Unites, SI), informally called the metric systems (centimeters, cubic meters, grams), is used along with English units throughout the volume. In some instances one or the other units are used exclusively where conversions would be awkward or space does not allow presentation of both units.

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Chapter 1: Introduction

Background

At the request of public and private wildland fire managers, who recognize a need to assimilate current fire effects knowledge, the Rocky Mountain Research Station has produced a state-of-the-art integrated series of documents relative to management of ecosystems (Smith 2000, Brown and Smith 2000, Sandberg and others 2002, Neary and others this volume, and Jones and Ryan in preparation). The series covers our technical understanding of fire effects, an understanding that has grown considerably since the first version of this series, the "Rainbow Series," was published in 1979. Since that time our awareness has grown that fire is a fundamental process of ecosystems that must be understood and managed to meet resource and ecosystem management goals. The volumes in the current series are intended to be useful for land management planning, development of environmental assessments and environmental impact statements, training and education, informing others such as conservation groups and regulatory agencies, and accessing technical literature. Knowledge of fire effects has risen in importance to land managers because fire, as a disturbance process, is an integral part of the concept of ecosystem management and restoration ecology. Fire initiates changes in ecosystems that affect the composition, structure, and patterns of vegetation on the landscape. It also affects the soil and water resources of ecosystems that are critical to overall functions and processes.

Fire is a dynamic process, predictable but uncertain, that varies over time and landscape space. It has shaped plant communities for as long as vegetation and lightning have existed on earth (Pyne 1982). Recycling of carbon (C) and nutrients depends on biological decomposition and fire. In regions where decay is constrained either by dry or cold climates or saturated (in other words, anaerobic) conditions, fire plays a dominant role in recycling organic matter (DeBano and others 1998). In warmer, moist climates, decay plays the dominant role in organic matter recycling (Harvey 1994), except in soils that are predominantly saturated (in other words, hydric soils).

The purpose of this volume, *Effects of Fire on Soils and Water*, is to assist land managers with ecosystem restoration and fire management planning responsibilities in their efforts to inform others about the impacts of fire on these ecosystem resources. The geographic coverage in this volume is North America, but the principles and effects can be applied to any ecosystem in which fire is a major disturbance process.

This publication is divided into three major parts and an introductory chapter that provides discussions of fire regimes, fire severity and intensity, and fire related disturbances. Part A describes the nature of the soil resource, its importance, characteristics and the responses of soils to fire, and the relationship of these features to ecosystem functioning and sustainability. Part A begins with a general overview and then is divided into three chapters (2, 3, and 4). Likewise, part B begins with a general overview then is divided into three chapters (5, 6, and 7) that discuss the basic hydrologic processes that are affected by fire, including the hydrologic cycle, water quality, and aquatic biology. Part C has five chapters that cover a wide range of related topics. Chapter 8 analyzes the effects of fire on the hydrology and nutrient cycling of wetland ecosystems along with management concerns. The use of models to describe heat transfer throughout the ecosystem and erosional response models to fire are discussed in chapter 9. Chapter 10 deals with important aspects of watershed rehabilitation and implementation of the Federal Burned Area Emergency Rehabilitation (BAER) program. Chapter 11 directs the fire specialists and mangers to important information sources including databases, Web sites, textbooks, journals, and other sources of fire effects information. A summary of the important highlights of the book are provided in chapter 12. The book concludes with a list of references used in the volume, and a glossary of fire terms.

Importance of Fire to Soil and Water

Soil is the unconsolidated, variable-thickness layer of mineral and organic matter on the Earth's surface that forms the interface between the geosphere and the atmosphere. It has formed as a result of physical, chemical, and biological processes functioning simultaneously on geologic parent material over long periods (Jenny 1941, Singer and Munns 1996). Soil is formed where there is continual interaction between the soil system and the biotic (faunal and floral), climatic (atmospheric and hydrologic), and topographic components of the environment. Soil interrelates with other ecosystem resources in several ways. It supplies air, water, nutrients, and mechanical support for the sustenance of plants. Soil also receives and processes rainfall. By doing so, it partly determines how much becomes surface runoff, and how much is stored for delivery slowly from upstream slopes to channels where it becomes streamflow, and by how much is stored and used for soil processes (for example, transpiration, leaching, and so forth). When the infiltration capacity of the soil for rainfall is exceeded, organic and inorganic soil particles are eroded from the soil surface and become a major source of sediment, nutrients, and pollutants in streams that affect water quality. There is also an active and ongoing exchange of gases between the soil and the surrounding atmosphere. Soil also provides a repository for many cultural artifacts, which can remain in the soil for thousands of years without undergoing appreciable change.

Fire can produce a wide range of changes in landscape appearance (fig. 1.1ABC; DeBano and others

Figure 1.1—Fire produced a wide range of changes in the forest landscape of a ponderosa pine forest in Arizona where their appearance ranged from (A) unburned ponderosa pine to those burned at (B) low-to-moderate severity and those burned at (C) high severity. (Photos by Peter Ffolliott).

1998). The fire-related changes associated with different severities of burn produce diverse responses in the water, soil, floral, and faunal components of the burned ecosystems because of the interdependency between fire severity and ecosystem response. Both immediate and long-term responses to fire occur (fig. 1.2). Immediate effects also occur as a result of the release of chemicals in the ash created by combustion of biomass. The response of biological components (soil microorganisms and ecosystem vegetation) to these changes is both dramatic and rapid. Another immediate effect of fire is the release of gases and other air pollutants by the combustion of biomass and soil organic matter. Air quality in large-scale airsheds can be affected during and following fires (Hardy and others 1998, Sandberg and others 2002). The long-term fire effects on soils and water are usually subtle, can persist for years following the fire, or be permanent as occurs when cultural resources are damaged (DeBano and others 1998, Jones and Ryan in preparation). Other long-term fire effects arise from the relationships between fire, soils, hydrology, nutrient cycling, and site productivity (Neary and others 1999).

In the previous "Rainbow" series published after the 1978 National Fire Effects Workshop that reviewed the state-of-knowledge of the effects of fire, separate reports were published on soil (Wells and others 1979) and water (Tiedemann and others 1979). Because of the intricate linkage between soil and water effects, this volume combines both.

Scope

The scope of this publication covers fire and disturbances in forest, woodland, and shrubland, and grassland ecosystems of the United States and Canada. However, it is applicable to any area in the world having similar forest types and fire regimes. In some instances, research information from ecosystems outside of North America will be used to

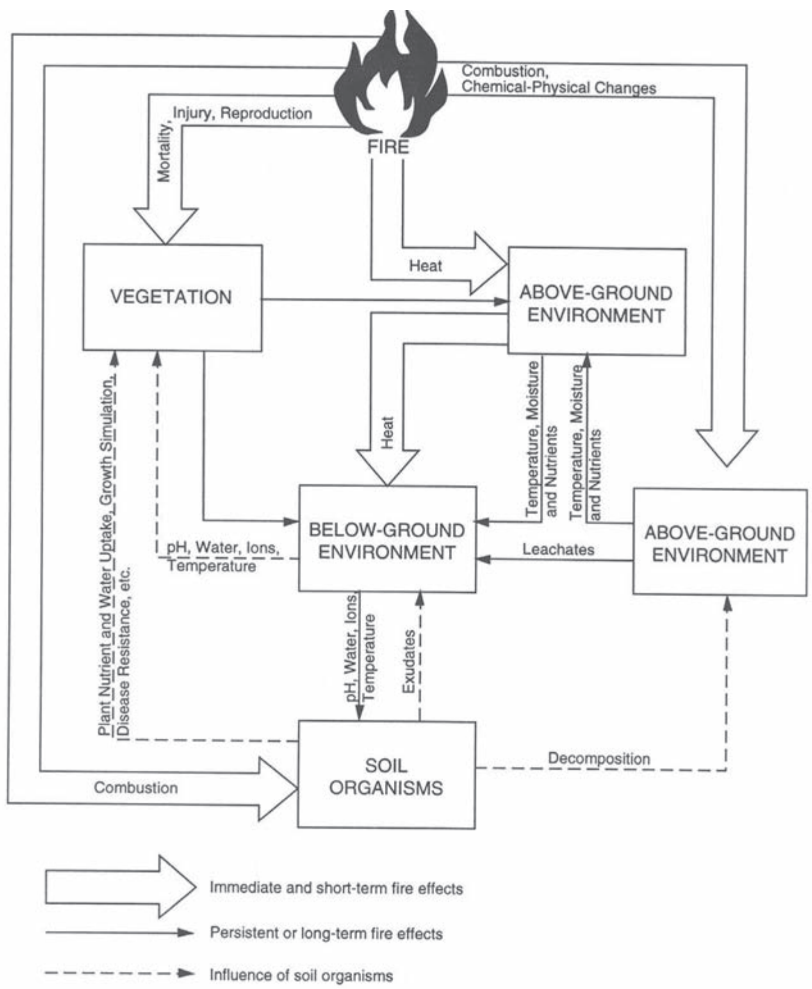

Figure 1.2—Immediate and long-term ecosystem responses to fire. (Adapted from Borchers and Perry 1990. In: *Natural and Prescribed Fire in Pacific Northwest Forests*, edited by J.D. Walstad, S.R. Radosevich, and D.V. Sandberg. Copyright © 1990 Oregon State University Press. Reproduced by permission).

elucidate specific fire effects on soils and water. Fire effects on ecosystems can be described at several spatial and temporal scales (Reinhardt and others 2001). In this chapter we will describe fire relationships suitable to each spatial and temporal scale.

The fire-related disturbances included in this review include wildland fires and prescribed fires, both in natural and management activity fuels. It also includes disturbances from fire suppression such as fire lines and roads, and fire retardant applications.

Fire Regimes

The general character of fire that occurs within a particular vegetation type or ecosystem across long successional time frames, typically centuries, is commonly defined as the characteristic fire regime. The fire regime describes the typical or modal fire severity that occurs. But it is recognized that, on occasion, fires of greater or lesser severity also occur within a vegetation type. For example, a stand-replacing crownfire is common in long fire-return-interval forests (fig. 1.3).

Figure 1.3—High severity, stand replacing wildfire. (Photo by USDA Forest Service).

The fire regime concept is useful for comparing the relative role of fire between ecosystems and for describing the degree of departure from historical conditions (Hardy and others 2001, Schmidt and others 2002). The fire regime classification used in this volume is the same as that used in the volume of this series (Brown 2000) on the effects of fire on flora. Brown (2000) contains a discussion of the development of fire regime classifications based on fire characteristics and effects (Agee 1993), combinations of factors including fire frequency, periodicity, intensity, size, pattern, season, and depth of burn (Heinselman 1978), severity (Kilgore 1981), and fire periodicity, season, frequency, and effects (Frost 1998). Hardy and others (1998, 2001) used modal severity and frequency to map fire regimes in the Western United States (table 1.1).

The fire regimes described in table 1.1 are defined as follows:

- **Understory Fire Regime:** Fires are generally nonlethal to the dominant vegetation and do not substantially change the structure of the dominant vegetation. Approximately 80 percent or more of the aboveground dominant vegetation survives fires. This fire regime applies to certain fire-resistant forest and woodland vegetation types.
- **Stand Replacement:** Fires are lethal to most of the dominant aboveground vegetation. Approximately 80 percent or more of the aboveground dominant vegetation is either consumed or dies as a result of fire, substantially changing the aboveground vegetative structure. This regime applies to fire-susceptible forests and woodlands, shrublands, and grasslands.
- **Mixed:** The severity of fires varies between nonlethal understory and lethal stand replacement fires with the variation occurring in space or time. First, spatial variability occurs when fire severity varies, producing a spectrum from

Table 1.1—Comparison of fire regime classifications according to Hardy and others (1998, 2001) and Brown (2000).

Hardy and others 1998, 2001			Brown 2000	
Fire regime group	Frequency (years)	Severity	Severity and effects	Fire regime
I	0-35	Low	Understory fire	1
II	0-35	Stand replacement	Stand replacement	2
III	35-100+	Mixed	Mixed	3
IV	35-100+	Stand replacement	Stand replacement	2
V	Greater than 200	Stand replacement	Stand replacement	2
			Non-fire regime	4

understory burning to stand replacement within an individual fire. This results from small-scale changes in the fire environment (fuels, terrain, or weather) and random changes in plume dynamics. Within a single fire, stand replacement can occur with the peak intensity at the head of the fire while a nonlethal fire occurs on the flanks. These changes create gaps in the canopy and small to medium sized openings. The result is a fine pattern of young, older, and multiple-aged vegetation patches. While this type of fire regime has not been explicitly described in previous classifications, it commonly occurs in some ecosystems because of fluctuations in the fire environment (DeBano and others 1998, Ryan 2002). For example, complex terrain favors mixed severity fires because fuel moisture and wind vary on small spatial scales. Secondly, temporal variation in fire severity occurs when individual fires alternate over time between infrequent low-intensity surface fires and long-interval stand replacement fires, resulting in a variable fire regime (Brown and Smith 2000, Ryan 2002). Temporal variability also occurs when periodic cool-moist climate cycles are followed by warm dry periods leading to cyclic (in other words, multiple decade-level) changes in the role of fire in ecosystem dynamics. For example in an upland forest, reduced fire occurrence during the cool-moist cycle leads to increased stand density and fuel build-up. Fires that occur during the transition between cool-moist and warm-dry periods can be expected to be more severe and have long-lasting effects on patch and stand dynamics (Kauffman and others 2003).

- **Nonfire Regime:** Fire is not likely to occur.

Subsequently, Schmidt and others (2002) used these criteria to map fire regimes and departure from historical fire regimes for the contiguous United States. This coarser-scale assessment was incorporated into the USDA Forest Service's Cohesive Strategy for protecting people and sustaining resources in fire adapted ecosystems (Laverty and Williams 2000). It is explicitly referred to in the United States as the 2003 Healthy Forest Restoration Act (HR 1904). The classification system used by Brown (2000) found in the *Effects of Fire on Flora* volume (Brown and Smith 2000) is also based on fire modal severity, emphasizes fire effects, but does not use frequency. Examples of vegetation types representative of each of the fire regime types are listed in tables 1.2a,b,c. These vegetation types are described in the Brown and Smith (2000) volume.

Fire Severity

At finer spatial and temporal scales the effects of a specific fire can be described at the stand and community level (Wells and others 1979, Rowe 1983, Turner and others 1994, DeBano and others 1998, Feller 1998, Ryan 2002). The commonly accepted term for describing the ecological effects of a specific fire is *fire severity*. Fire severity describes the magnitude of the disturbance and, therefore, reflects the degree of change in ecosystem components. Fire affects both the aboveground and belowground components of the

Table 1.2a—Examples of vegetation types associated with understory fire regimes in the United States and Canada. Some widely distributed vegetation types occur in more than one fire regime. Variations in fire regime result from regional differences in terrain and fire climate.

Communities	Source
Longleaf pine	Wade and others 2000
Slash pine	Wade and others 2000
Loblolly pine	Wade and others 2000
Shortleaf pine	Wade and others 2000
Pine flatwoods and pine rocklands	Myers 2000
Pondcypress wetlands	Myers 2000
Cabbage palmetto savannas and forests	Myers 2000
Oak-hickory forests	Wade and others 2000
Live oak forests	Myers 2000
Ponderosa pine	Arno 2000, Paysen and others 2000
Ponderosa pine-mixed conifer	Arno 2000
Jeffrey pine	Arno 2000
Redwood	Arno 2000
Oregon oak woodlands	Arno 2000

Table 1.2b—Examples of vegetation types associated with mixed severity fire regimes in the United States and Canada. Some widely distributed vegetation types occur in more than one fire regime. Variations in fire regime result from regional differences in terrain and fire climate.

Communities	Source
Aspen	Duchesne and Hawkes 2000
Eastern white pine	Duchesne and Hawkes 2000
Red pine	Duchesne and Hawkes 2000
Jack pine	Duchesne and Hawkes 2000
Virginia pine	Wade and others 2000
Pond pine	Wade and others 2000
Mixed mesophytic hardwoods	Wade and others 2000
Northern hardwoods	Wade and others 2000
Bottomland hardwoods	Wade and others 2000
Coast Douglas-fir and Douglas-fir/hardwoods	Arno 2000
Giant sequoia	Arno 2000
California red fir	Arno 2000
Sierra/Cascade lodgepole pine	Arno 2000
Rocky Mountain lodgepole pine	Arno 2000
Interior Douglas-fir	Arno 2000
Western larch	Arno 2000
Whitebark pine	Arno 2000
Ponderosa pine	Arno 2000
Pinyon-juniper	Paysen and others 2000
Texas savanna	Paysen and others 2000
Western oaks	Paysen and others 2000

Table 1.2c—Examples of vegetation types associated with stand replacement fire regimes in the United States and Canada. Some widely distributed vegetation types occur in more than one fire regime. Variations in fire regime result from regional differences in terrain and fire climate.

Communities	Source
Boreal spruce-fir	Duchesne and Hawkes 2000
Conifer bogs	Duchesne and Hawkes 2000
Tundra	Duchesne and Hawkes 2000
Wet grasslands	Wade and others 2000
Prairie	Wade and others 2000
Bay forests	Wade and others 2000
Sand pine	Wade and others 2000
Table mountain pine	Wade and others 2000
Eastern spruce-fir	Wade and others 2000
Atlantic white-cedar	Wade and others 2000
Salt and brackish marshes	Wade and others 2000
Fresh and oligohaline marshes and wet prairie	Myers 2000, Wade and others 2000
Florida coastal prairies	Myers 2000
Florida tropical hardwood forests	Myers 2000
Hawaiian forests and grasslands	Myers 2000
Forests of Puerto Rico and the Virgin Islands	Myers 2000
Coast Douglas-fir	Arno 2000
Coastal true fir/mountain hemlock	Arno 2000
Interior true fir-Douglas-fir-western larch	Arno 2000
Rocky Mountain lodgepole pine	Arno 2000
Western white pine-cedar-hemlock	Arno 2000
Western spruce-fir-whitebark pine	Arno 2000
Aspen	Arno 2000
Grasslands (annual and perennial)	Paysen and others 2000
Sagebrush	Paysen and others 2000
Desert shrublands	Paysen and others 2000
Southwestern shrubsteppe	Paysen and others 2000
Chaparral-mountain shrub	Paysen and others 2000

ecosystem. Thus severity integrates both the heat pulse above ground and the heat pulse transferred downward into the soil. It reflects the amount of energy (heat) that is released by a fire that ultimately affects resources and their functions. It can be used to describe the effects of fire on the soil and water system, ecosystem flora and fauna, the atmosphere, and society (Simard 1991). It reflects the amount of energy (heat) that is released by a fire that ultimately affects resource responses. Fire severity is largely dependent upon the nature of the fuels available for burning, and the combustion characteristics (in other words, flaming versus smoldering) that occur when these fuels are burned. This chapter emphasizes the relationship of fire severity to soil responses because the most is known about this relationship, and because soil responses (see chapters 2, 3, 4, and 10) are closely related to hydrologic responses (see chapters 5 and 6) and ecosystem productivity (see chapters 4 and 8).

Fire Intensity versus Fire Severity

Although the literature historically contains confusion between the terms *fire intensity* and *fire severity*, a fairly consistent distinction between the two terms has been emerging in recent years. Fire managers trained in the United States and Canada in fire behavior prediction systems use the term fire intensity in a strict thermodynamic sense to describe the rate of energy released (Deeming and others 1977, Stocks and others 1989). Fire intensity is concerned mainly with the rate of aboveground fuel consumption and, therefore the energy release rate (Albini 1976, Alexander 1982). The faster a given quantity of fuel burns, the greater the intensity and the shorter the duration (Byram 1959, McArthur and Cheney 1966, Albini 1976, Rothermel and Deeming 1980, Alexander 1982). Because the rate at which energy can be transmitted through the soil is limited by the soil's thermal properties, the duration of burning is critically important to the effects on soils (Frandsen and Ryan 1986, Campbell and others 1995). Fire intensity is not necessarily related to the total amount of energy produced during the burning process. Most energy released by flaming combustion of aboveground fuels is not transmitted downward (Packham and Pompe 1971, Frandsen and Ryan 1985). For example, Packham and Pompe (1971) found that only about 5 percent of the heat released by a surface fire was transmitted into the ground. Therefore, fire intensity is not necessarily a good measure of the amount of energy transmitted downward into the soil, or the associated changes that occur in physical, chemical, and biological properties of the soil. For example, it is possible that a high intensity and fast moving crown fire will consume little of the surface litter because only a small amount of the energy released during the combustion of fuels is transferred downward to the litter surface (Rowe 1983, VanWagner 1983, Ryan 2002). In this case the surface litter is blackened (charred) but not consumed. In the extreme, one author of this chapter has seen examples in Alaska and North Carolina where fast spreading crown fires did not even scorch all of the surface fuels. However, if the fire also consumes substantial surface and ground fuels, the residence time on a site is greater, and more energy is transmitted into the soil. In such cases, a "white ash" layer is often the only postfire material left on the soil surface (Wells and others 1979, Ryan and Noste 1985) (fig. 1.4).

Because one can rarely measure the actual energy release of a fire, the term fire intensity can have limited practical application when evaluating ecosystem responses to fire. Increasingly, the term fire severity is used to indicate the effects of fire on the different ecosystem components (Agee 1993, DeBano and others 1998, Ryan 2002). Fire severity has been used describe the magnitude of negative fire impacts on natural ecosystems in the past (Simard 1991), but a wider usage of the term to include all fire effects is proposed. In this context severity is a description of the magnitude of change resulting from a fire and does not necessarily imply negative consequences. Thus, a low severity fire may restore and maintain a variety of ecological attributes that are generally viewed as positive, as for example in a fire-adapted longleaf pine (*Pinus palustris*) or ponderosa pine (*P. ponderosa*) ecosystem. In contrast a high severity fire may be a dominant, albeit infrequent, disturbance in a non-fire-adapted ecosystem, for example, spruce (*Picea* spp.) whereas it is abnormal in a fire-adapted ecosystem.

Figure 1.4—Gray to white ash remaining after a pinyon-juniper slash pile was burned at high temperatures for a long duration, Apache-Sitgreaves National Forest, Arizona. (Photo by Steve Overby).

While all high severity fires may have significant negative social impacts, only in the latter case is the long-term functioning of the ecosystem significantly altered.

Fire Intensity Measures

Byram's (1959) definition of *fireline intensity* has become a standard quantifiable measure of intensity (Van Wagner 1983, Agee 1993, DeBano and others 1998). It is a measure of the rate of energy release in the flaming front of the spreading fire. It does not address the residual flaming behind the front nor subsequent smoldering combustion (Rothermel and Deeming 1980, Alexander 1982). Fireline intensity can be written as a simple equation:

$$I = Hwr$$

where

- I = fireline intensity (BTU/ft/sec or kW/m/sec)
- H = heat yield (BTU/lb or kW/kg of fuel)
- w = mass of available fuel burned (lb/ft^2 or kg/m^2)
- r = rate of spread (ft/sec or m/sec)

Fireline intensity is proportional to the flame length in a spreading fire and is a useful measure of the potential to cause damage to aboveground structures (Van Wagner 1973, Rothermel and Deeming 1980, Alexander 1982, Ryan and Noste 1985). The Canadian forest fire danger rating system calculates the intensity of surface fires and crown fires based on Byram's equation (Stocks and others 1989). Rothermel (1972) defined a somewhat different measure of fire intensity, heat per unit area, which is commonly used in fire behavior prediction in the United States (Albini 1976, Rothermel and Deeming 1980, Andrews 1986, Scott 1998, Scott and Reinhardt 2001). One problem with using current fire behavior prediction systems in ecological studies is that they focus on flaming combustion of fine fuels and do not predict all of the combustion and fuel consumned, or quantify all of the energy released, during a fire (Johnson and Miyanishi 2001). The intensity of residual combustion of large woody fuels is modeled in the BURNUP Model (Albini and Reinhardt 1995, Albini and others 1996) and the energy release rate from duff consumption is modeled in the First Order Fire Effects Model (FOFEM) v.5.0 (Reinhardt 2003).

Fires burn throughout a continuum of energy release rates (table 1.3) (Artsybashev 1983, Rowe 1983, Van Wagner 1983, Rothermel 1991). Ground fires burn in compact fermentation and humus layers and in organic muck and peat soils where they spread predominantly by smoldering (glowing) combustion and typically burn for hours to weeks (fig. 1.5). Forward rates of spread in ground fires range on the order of several inches (decimeters) to yards (meters) per day. Temperatures are commonly in excess of 572 °F (300 °C) for several hours (Frandsen and Ryan 1986, Ryan and Frandsen 1991, Hartford and Frandsen 1992, Agee 1993). The conditions necessary for ground fires are organic soil horizons greater than about 1.6 to 2.4 inches (4 to 6 cm) deep and extended drying (Brown and others 1985, Reinhardt and others 1997, Johnson and Miyanishi 2001). Surface fires spread by flaming combustion in loose litter, woody debris, herbaceous plants and shrubs and trees roughly less than 6 feet

Table 1.3–Representative ranges for fire behavior characteristics for ground, surface, and crown fires (From Ryan 2002).

Fire type	Dominant combustion phase	General description	Fire behavior characteristics: Rate of spread	Flame length	Fireline intensity
			(meters/minute)	*(meters)*	*(kW/meter)*
Ground	Smoldering	Creeping	0.1 to 0.2	0.0	<10
Surface	Flaming	Creeping	<1.1	0.1 to 0.5	1.7 to 15.8
		Active/Spreading	<1.1 to 8.3	0.5 to 1.5	15.8 to 46.6
		Intense/Running	8.3 to 42.9	1.5 to 3.0	46.6 to 56.2
Transition	Flaming	Passive Crowning (Intermittent Torching)	Variable[1]	3.0 to 10.0	Variable[1]
Crowning	Flaming	Active Crowning	4.1 to 7.4	5.0 to 15[2]	54.6 to 1,484.1
		Independent Crowning	Up to 14.8	Up to 70.0[2]	Up to 524.5

[1]Rates of spread, flame length, and fireline intensity vary widely in transitional fires. In subalpine and boreal fuels it is common for surface fires to creep slowly until they encounter conifer branches near the ground, then individual trees or clumps of trees torch sending embers ahead of the main fire. These embers start new fires, which creep until they encounter trees, which then torch. In contrast, as surface fires become more intense, torching commonly occurs prior to onset of active crowning.

[2]Flame lengths are highly variable in crown fires. They commonly range from 0.5 to 2 times canopy height. Fire managers commonly report much higher flames but these are difficult to verify or model. Such extreme fires are unlikely to result in additional fire effects within a stand but are commonly associated with large patches of continuous severe burning.

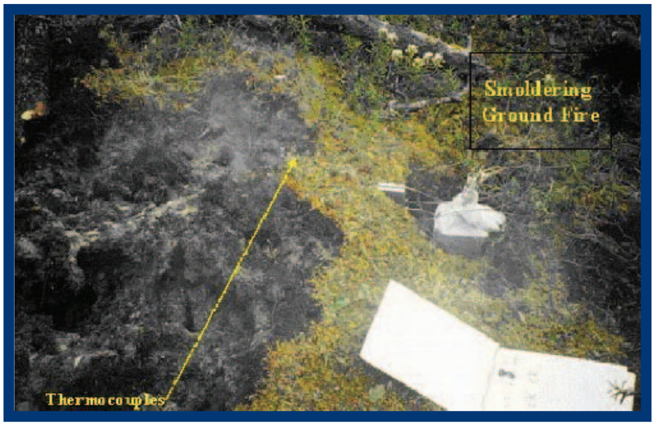

Figure 1.5—Smoldering ground fire. (Photo by Kevin Ryan).

(about 2 m) tall. Under marginal burning conditions surface fires creep along the ground at rates of 3 feet/hour (less than 1 m/hr) with flames less than 19 inches (less than 0.5 m) high (table 1.3). As fuel, weather, and terrain conditions become more favorable for burning, surface fires become progressively more active with spread rates ranging on the order of from tens of yards (meters) to miles (kilometers) per day. The duration of surface fires is on the order of 1 to a few minutes (Vasander and Lindholm 1985, Frandsen and Ryan 1986, Hartford and Frandsen 1992) except where extended residual burning occurs beneath logs or in concentrations of heavy woody debris. Here flaming combustion may last a few hours resulting in substantial soil heating (Hartford and Frandsen 1992). However, the surface area occupied by long-burning woody fuels is typically small, less than 10 percent and often much less (Albini 1976, Ryan and Noste 1985, Albini and Reinhardt 1995). If canopy fuels are plentiful and sufficiently dry, surface fires begin to transition into crown fires (Van Wagner 1977, Scott and Reinhardt 2001). Crown fires burn in the foliage, twigs, and epiphytes of the forest or shrub canopy located above the surface fuels. Such fires exhibit the maximum energy release rate but are typically of short duration, 30 to 80 seconds.

Fires burn in varying combinations of ground, surface, and crown fuels depending on the local conditions at the specific time a fire passes a given point. Ground fires burn independently from surface and crown fires and often occur some hours after passage of the flaming front (Artsybashev 1983, Rowe 1983, Van Wagner 1983, Hungerford and others 1995a, Hungerford and others 1995b). Changes in surface and ground fire behavior occur in response to subtle changes in the microenvironment, stand structure, and weather leading to a mosaic of fire treatments at multiple scales in the ground, surface, and canopy strata (Ryan 2002).

Depth of Burn Measures

The relationship of fire intensity to fire severity remains largely undefined because of difficulties encountered in relating resource responses to the burning process (Hungerford and others 1991, Hartford and Frandsen 1992, Ryan 2002). While quantitative relationships have been developed to describe changes in the thermal conductivity of soil, and changes in soil temperature and water content beneath surface and ground fires, these relationships have not been thoroughly extrapolated to field conditions (Campbell and others 1994, 1995). It is not always possible to estimate the effects of fire on soil, vegetation, and air when these effects are judged by only fire intensity measurements because other factors overwhelm fire behavior. The range of fire effects on soil resources can be expected to vary directly with the *depth of burn* as reflected in the amount of duff consumed and degree of large woody fuel consumption (Ryan 2002). Thus, for example, the depth of lethal heat (approximately 140 °F or 60 °C) penetration into the soil can be expected to increase with the increasing depth of surface duff that is burned (fig. 1.6).

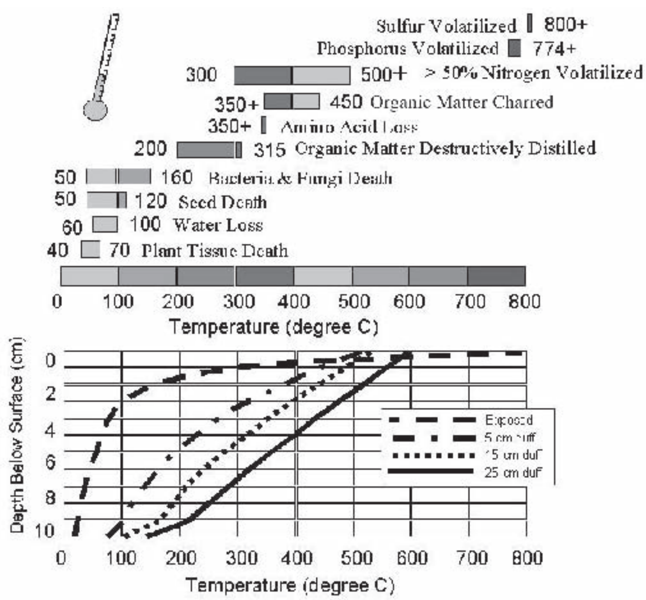

Figure 1.6—Temperature ranges associated with various fire effects (top) (from Hungerford and others 1991) compared to the depth of heat penetration into mineral soil (bottom) for a crown fire over exposed mineral soil (observed in jack pine *Pinus banksiana* in the Canadian Northwest Territories) or for ground fire burning in 5-, 15-, and 25-cm of duff (predicted via Campbell and others1994, 1995). Conditions are for coarse dry soil, which provides the best conduction (i.e., a worst-case scenario). (From Ryan 2002).

Numerous authors have used measures of the depth of burn into the organic soil horizons or visual observation of the degree of charring and consumption of plant materials to define fire severity for interpreting the effects of fire on soils, plants, and early succession (Conrad and Poulton 1966, Miller 1977, Viereck and Dyrness 1979, Viereck and Schandelmeier 1980, Dyrness and Norum 1983, Rowe 1983, Zasada and others 1983, Ryan and Noste 1985, Morgan and Neuenschwander 1988, Schimmel and Granström 1996, DeBano and others 1998, Feller 1998). Depth of burn is directly related to the duration of burning in woody fuels (Anderson 1969, Albini and Reinhardt 1995) and duff (Frandsen 1991a, 1991b, Johnson and Miyanishi 2001). In heterogeneous fuels, depth of burn can vary substantially over short distances (for example, beneath a shrub or tree canopy versus the inter-canopy area, or beneath a log versus not (Tunstall and others 1976, Ryan and Frandsen 1991). At the spatial scale of a sample plot within a given fire, depth of burn can be classified on the basis of visual observation of the degree of fuel consumption and charring on residual plant and soil surfaces (Ryan and Noste 1985, Ryan 2002).

Ryan and Noste (1985) summarized literature on the relationships between depth of burn and the charring of plant materials. An adaptation of their table 2, updated to reflect subsequent literature (Moreno and Oechel 1989, Pérez and Moreno 1998, DeBano and others 1998, and Feller 1998) and experience, particularly in peat and muck soils, is presented in table 1.4. This table can be used as a field guide to classifying depth of burn on small plots (for example, quadrats). A brief description of depth of burn characteristics is provided for clarification of subsequent discussion of fire effects:

- *Unburned:* Plant parts are green and unaltered, there is no direct effect from heat. The extent of unburned patches (mosaics) varies considerably within and between burns as the fire environment (fuels, weather, and terrain) varies. Unburned patches are important rufugia for many species and are a source of plants and animals for recolinization of adjacent burned areas.
- *Scorched:* Fire did not burn the area, but radiated or convected heat from adjacent burned areas caused visible damage. Mosses and leaves are brown or yellow but species characteristics are still identifiable. Soil heating is negligible. Scorched areas occur to varying degrees along the edges of more severely burned areas. As it occurs on edges, the area within the scorched class is typically small (Dyrness and Norum 1983) and effects are typically similar to those in light burned areas. The scorched class may, however, have utility in studies of microvariation of fire effects.
- *Light:* In forests the surface litter, mosses, and herbaceous plants are charred-to-consumed but the underlying forest duff or organic soil is unaltered. Fine dead twigs up to 0.25 inches (0.6 cm) are charred or consumed, but larger unburned branches remain. Logs may be blackened but are not deeply charred except where two logs cross. Leaves of understory shrubs and trees are charred or consumed, but fine twigs and branches remain. In nonforest vegetation, plants are similarly charred or consumed, herbaceous plant bases are not deeply burned and are still identifiable, and charring of the mineral soil is negligible. Light depth of burn is associated with short duration fires either because of light fuel loads (mass per unit area), high winds, moist fuels, or a combination of these three factors. Typical forest-floor moisture contents associated with light depth of burn are litter (O_i) 15 to 25 percent and duff (O_e+O_a) greater than 125 percent.
- *Moderate:* In forests the surface litter, mosses, and herbaceous plants are consumed. Shallow duff layers are completely consumed, and charring occurs in the top 0.5 inch (1.2 cm) of the mineral soil. Where deep duff layers or organic soils occur, they are deeply burned to completely consumed, resulting in deep char and ash deposits but the texture and structure of the underlying mineral soil are not visibly altered. Trees of late-successional, shallow-rooted species are often left on root pedestals or topple. Fine dead twigs are completely consumed, larger branches and rotten logs are mostly consumed, and logs are deeply charred. Burned-out stump holes and rodent middens are common. Leaves of understory shrubs and trees are completely consumed. Fine twigs and branches of shrubs are mostly consumed (this effect decreases with height above the ground), and only the larger stems remain. Stems of these plants frequently burn off at the base during the ground fire phase leaving residual aerial stems that were not consumed in the flaming phase lying on the ground. In nonforest vegetation, plants are similarly consumed, herbaceous plant bases are deeply burned and unidentifiable. In shrublands, average char-depth of the mineral soil is on the order of less than 0.4 inch (1 cm), but soil texture and structure are not noticeably altered. Charring may extend 0.8 to 1.2 inches (2.0 to 3.0 cm) beneath shrubs with deep leaf

Table 1.4—Visual characteristics of depth of burn in forests, shrublands, and grasslands from observations of ground surface characteristics, charring, and fuel consumption for unburned and light (part A), moderate (part B), and deep (part C) classes (Modified from Ryan and Noste 1985).

Table 1.4 Part A

Depth of burn class	Vegetation type: Forests	Shrublands	Grasslands
Unburned			
Surface:	Fire did not burn on the surface.	See Forests	See Forests
Fuels:	Some vegetation injury may occur from radiated or convected heat resulting in an increase in dead fuel mass.	See Forests	See Forests
Occurrence:	A wide range exists in the percent unburned in natural fuels. Under marginal surface fire conditions the area may be >50 percent. Under severe burning conditions <5 percent is unburned. Commonly 10 to 20 percent of the area in slash burns is unburned. Unburned patches provide refugia for flora and fauna.	See Forests	See Forests
Light			
Surface:	Leaf litter charred or consumed. Upper duff charred but full depth not altered. Gray ash soon becomes inconspicuous leaving a surface that appears lightly charred to black.	Leaf litter charred or consumed, but some leaf structure is discernable. Leaf mold beneath shrubs is scorched to lightly charred but not altered over its entire depth. Where leaf mold is lacking charring is limited to <0.2 cm into mineral soil. Some gray ash may be present but soon becomes inconspicuous leaving a blackened surface beneath shrubs.	Leaf litter is charred or consumed but some plant parts are discernable. Herbaceous stubble extends above the soil surface. Some plant parts may still be standing, bases not deeply burned, and still recognizable. Surface is black after fire but this soon becomes inconspicuous. Charring is limited to <0.2 cm into the soil.
Fuels:	Herbaceous plants and foliage and fine twigs of woody shrubs and trees are charred to consumed but twigs and branches >0.5 cm remain. Coarser branches and woody debris are scorched to lightly charred but not consumed. Logs are scorched to blackened but not deeply charred. Rotten wood scorched to partially burned.	Typically, some leaves and twigs remain on plants and <60 percent of brush canopy is consumed. Foliage is largely consumed whereas fine twigs and branches >0.5 cm remain.	Typically, 50 to 90 percent of herbaceous fuels are consumed and much of the remaining fuel is charred.
Occurrence:	Light depth of burn commonly occurs on 10 to 100 percent of the burned area in natural fuels and 45 to 75 percent in slash fuels. Low values are associated with marginal availability of fine fuels whereas high values are associated with continuous fine fuels or wind-driven fires.	In shrublands where fine fuels are continuous, light depth of burn occurs on 10 to 100 percent depending on fine fuel moisture and wind. Where fine fuels are limited, burns are irregular and spotty at low wind speeds. Moderate to high winds are required for continuous burns.	Burns are spotty to uniform, depending on grass continuity. Light depth of burn occurs in grasslands when soil moisture is high, fuels are sparse, or fires burn under high wind. This is the dominant type of burning in most upland grasslands.

(con.)

Table 1.4 Part B

Depth of burn class	Vegetation type		
	Forests	Shrublands	Grasslands
Moderate			
Surface:	In upland forests litter is consumed and duff deeply charred or consumed, mineral soil not visibly altered but soil organic matter has been partially pyrolized (charred) to a depth >1.0 cm. Gray or white ash persists until leached by rain or redistributed by rain or wind. In forests growing on organic soils moderate depth of burn fires partially burn the root-mat but not the underlying peat or muck.	In upland shrublands litter is consumed. Where present, leaf mold deeply charred or consumed. Charring 1 cm into mineral soil, otherwise soil not altered. Gray or white ash quickly disappears. In shrub-scrub wetlands growing on organic soils moderate depth of burn fires partially burn the root-mat but not the underlying peat or muck.	In upland grasslands litter is consumed. Charring extends to <0.5 cm into mineral soil, otherwise soil not altered. Gray or white ash quickly disappears. In grasslands, sedge meadows and prairies growing on organic soils moderate depth of burn fires partially burn the root-mat but not the underlying peat or muck.
Fuels:	Herbaceous plants, low woody shrubs, foliage and woody debris <2.5 cm diameter consumed. Branch-wood 2.5 to 7.5 cm 90+ percent consumed. Skeletons of larger shrubs persist. Logs are deeply charred. Shallow-rooted, late successional trees and woody shrubs are typically left on pedestals or topple. Burned-out stump holes are common.	Herbaceous plants are consumed to the ground-line. Foliage and branches of shrubs are mostly consumed. Stems <1 cm diameter are mostly consumed. Stems >1 cm mostly remain.	Herbaceous plants are consumed to the ground-line.
Occurrence:	Moderate depth of burn occurs on 0 to 100 percent of natural burned areas and typically 10 to 75 percent on slash burns. High variability is due to variability in distributions of duff depth and woody debris.	Moderate depth of burn varies with shrub cover, age, and dryness. It typically occurs beneath larger shrubs and increases with shrub cover. Typically burns are more uniform than in light depth of burn fires.	Moderate depth of burn tends to occur when soil moisture is low and fuels are continuous. Then burns tend to be uniform. In discontinuous fuels high winds are required for high coverage in moderate depth of burn.
Table 1.4 Part C			
Deep			
Surface:	In forests growing on mineral soil the litter and duff are completely consumed. The top layer of mineral soil visibly altered. Surface mineral soil structure and texture are altered and soil is oxidized (reddish to yellow depending on parent material). Below oxidized zone, >1 cm of mineral soil appears black due to charred or deposited organic material. Fusion of soil may occur under heavy woody fuel concentrations. In forests growing on organic soils deep depth of burn fires burn the root-mat and the underlying peat or muck to depths that vary with the water table.	In shrublands growing on mineral soil the litter is completely consumed leaving a fluffy white ash surface that soon disappears. Organic matter is consumed to depths of 2 to 3 cm. Colloidal structure of surface mineral soil is altered. In shrub-scrub wetlands growing on organic soils deep depth of burn fires burn the root-mat and the underlying peat or muck to depths that vary with the water table.	In grasslands growing on mineral soil the litter is completely consumed leaving a fluffy white ash surface that soon disappears. Charring to depth of 1 cm in mineral soil. Soil structure is slightly altered. In grasslands growing on organic soils deep depth of burn fires burn the root-mat and the underlying peat or muck to depths that vary with the water table.
Fuels:	In uplands twigs and small branches are completely consumed. Few large, deeply charred branches remain. Sound logs are deeply charred and rotten logs are completely consumed. In wetlands twigs, branches, and stems not burned in the surface fire may remain even after subsequent passage of a ground fire.	In uplands twigs and small branches are completely consumed. Large branches and stems are mostly consumed. In wetlands twigs, branches, and stems not burned in the surface fire may remain even after subsequent passage of a ground fire.	All above ground fuel is consumed to charcoal and ash.
Occurrence:	In uplands deep depth of burn occurs under logs, beneath piles, and around burned-out stump holes, and typically occupies <10 percent of the surface except under extreme situations (e.g., extensive blow-down). In forested wetlands deep depth of burn can occur over large areas when the water table is drawn down during drought.	In uplands deep depth of burn typically is limited to small areas beneath shrubs where concentrations of deadwood burn-out. In shrub-scrub wetlands – see forests	In uplands deep depth of burn is limited to areas beneath the occasional log or anthropogenic features (e.g., fences, corrals). In wetlands – see forests.

litter. Typical forest-floor moisture contents associated with moderate depth of burn are litter (O_i) 10 to 20 percent and duff (O_e+O_a) less than 75 percent.

- *Deep:* In forests growing on mineral soil the surface litter, mosses, herbaceous plants, shrubs, and woody branches are completely consumed. Sound logs are consumed or deeply charred. Rotten logs and stumps are consumed. The top layer of the mineral soil is visibly oxidized, reddish to yellow. Surface soil texture is altered and in extreme cases fusion of particles occurs. A black band of charred organic matter 0.4 to 0.8 inch (1 to 2 cm) thick occurs at variable depths below the surface. The depth of this band increases with the duration of extreme heating. The temperatures associated with oxidized mineral soil are typical of those associated with flaming (greater than 932 °F or 500 °C) rather than smoldering (less than 932 °F or 500 °C). Thus, deep depth of burn typically only occurs where woody fuels burn for extended duration such as beneath individual logs or in concentrations of woody debris and litter-filled burned out stump holes. Representative forest-floor moisture contents associated with deep depth of burn are litter (O_i) less than 15 percent and duff (O_e+O_a) less than 30 percent. In areas with deep organic soils deep depth-of-burn occurs when ground fires consume the root-mat or burn beneath the root-mat. Trees often topple in the direction from which the smoldering fire front approached (Artsybashev 1983, Wein 1983, Hungerford and others 1995a,b).

Depth of burn varies continuously and, as is typical of classifications, there is some ambiguity at the class boundaries. The moderate depth of burn class is a broad class. Some investigators have chosen to divide the class into two classes (Morgan and Neuenschwander 1988, Feller 1998). The most common criteria for splitting the moderate class are between areas with shallow versus deep duff. Partial consumption of a deep layer may be more severe than complete consumption of a shallow layer for some effects but not others. For example, consumption of 8 inches (20 cm) of a 12-inch (30-cm) duff layer represents greater fuel consumption, smoke production, energy release, and nutrient release than complete consumption of a 4-inch (10-cm) layer, but because organic matter (duff) is a good insulator, heat effects are limited to less than an inch (1 to 2 cm) below the duff-burn boundary. In contrast, complete consumption of the 4-inch (10-cm) layer can be expected to have similar thermal effects at three to five times greater depth (fig. 1.6). Duff consumption is a complex process (Johnson and Miyanishi 2001). Depth, bulk density, heat content, mineral content, moisture content, and wind speed all affect the energy release rate and soil heating. As these factors cannot be readily determined after a fire, it is difficult to describe postburn criteria that can be used to consistently split the class. While postfire examination of ground charring alone may not be adequate for classifying depth of burn, the actual depth can be inferred from the preponderance of the evidence, which includes reconstructing the prefire vegetative structure. Careful postfire observations of soil characteristics, fuel consumption, and the depth of charring of residual plant materials can be used to classify the depth of burn by using the descriptive characteristics provided in table 1.4.

Fire Severity Classification

Judging fire severity solely on ground-based processes ignores the aboveground dimension of severity implied in the ecological definition of the severity of a disturbance (White and Pickett 1985). This is especially important because soil heating is commonly shallow even when surface fires are intense (Wright and Bailey 1982, Vasander and Lindholm 1985, Frandsen and Ryan 1986, Hartford and Frandsen 1992, Ryan 2002). Ryan and Noste (1985) combined fire intensity classes with depth of burn (char) classes to develop a two-dimensional matrix approach to defining fire severity. Their system is based on two components of fire severity: (1) an aboveground heat pulse due to radiation and convection associated with flaming combustion, and (2) a belowground heat pulse due principally to conduction from smoldering combustion where duff is present or radiation from flaming combustion where duff is absent—in other words, bare mineral soil. Fire-intensity classes qualify the relative peak energy release rate for a fire, whereas depth of burn classes qualify the relative duration of burning. Their concept of severity focuses on the ecological work performed by fire both above ground and below ground. Ryan (2002) combined surface fire characteristic classes (table 1.3) and depth of burn classes (table 1.4) to revise the Ryan and Noste (1985) fire severity matrix (table 1.5). By this nomenclature two burned areas would be contrasted as having had, for example, an active spreading-light depth of burn fire versus an intense-moderate depth of burn fire. The matrix provides an approach to classifying the level of fire treatment or severity for ecological studies at the scale of the individual plant, sampling quadrat, and the community. The Ryan and Noste (1985) approach has been used to interpret differences in plant survival and regeneration (Willard and others 1995, Smith and

Table 1.5—Two-dimensional Fire Severity Matrix that relates fire intensity and depth of burn to One-dimensional, relative fire severity ratings (Modified from Ryan and Noste 1985 Figure 1) .

	Characteristic depth of burn			
Characteristic fire behavior	**Unburned/scorched**	**Light**	**Moderate**	**Deep**
Crowning	**Low** (common edge effect) – when radiation and convection from nearby burning scorch foliage but surface is unburned. **Moderate** (occurs) – when fire burns over snow or water (wetlands).	**Moderate** – when crown-fire occurs over wet duff/soil, or thin (<4 cm) duff.[1] **High** – when crownfire occurs over bare mineral soil.	**Moderate** – when residual duff (uplands) or root mat (wetlands) are present. **High** – when duff or root mat is completely consumed.	**High**
Intense/running and torching	See above	See above	See above	**High**
Active/Spreading	**Low** – See Above	**Low**	**Low** – when residual duff (uplands) or root mat (wetlands) are present. **Deep** – when duff or root mat is completely consumed.	**Moderate** – when forest canopy remains. **High** – in forests shrublands and grasslands where aboveground vegetation is consumed.
Creeping	**Low** – boundary condition of no practical significance, except as noted below.	**Low**	**Low** – when residual duff (uplands) or root mat (wetlands) are present. **Moderate** – when duff or root mat is completely consumed.	See Above
Unburned	**Refugia** – flora and fauna not directly affected by fire but microenvironment may be altered	NA	NA	NA

[1]Duff insulates the mineral soil from intense heat associated with flaming, and thin duff does not burn independently by smoldering combustion. As a result maximum temperatures and soil heat flux are reduced.

Fischer 1997, Feller 1998) and to field-validate satellite-based maps of burned areas (White and others 1996). The depth of burn characteristics are appropriate for quadrat-level descriptions in species response studies and for describing fire severity on small plots within a burned area.

In the literature there is common usage of a one-dimension rating of fire severity (Wells and others 1979, Morrison and Swanson 1990, Agee 1993, DeBano and others 1998, and many others). The single-adjective rating describes the overall severity of the fire and usually focuses primarily on the effects on the soil resource. The fire severity rating in table 1.5 provides guidance for making comparisons to the two-dimensional severity rating of Ryan and Noste (1985) and Ryan (2002), and for standardizing the use of the term. At the spatial scale of the stand or community, fire severity needs to be based on a sample of the distribution of fire severity classes. In the original Rainbow volume on the effects of fire on soils, Wells and others 1979 (see also

Ryan and Noste 1985, DeBano and others 1998) developed the following criteria to do this:

- Low severity burn—less than 2 percent of the area is severely burned, less than 15 percent moderately burned, and the remainder of the area burned at a low severity or unburned.
- Moderate severity burn—less than 10 percent of the area is severely burned, but more than 15 percent is burned moderately, and the remainder is burned at low severity or unburned.
- High severity burn—more than 10 percent of the area has spots that are burned at high severity, more than 80 percent moderately or severely burned, and the remainder is burned at a low severity.

The Wells and others (1979) criteria for defining the burn severity class boundaries are somewhat arbitrary but were selected on the basis of experience recognizing that even the most severe of fires has spatial variation due to random variation in the fire environment (fuels, weather, and terrain), and particularly localized fuel conditions. Recently, Key and Benson (2004) have developed a series of procedures for documenting fire severity in the context of field validation of satellite images of fire severity. Their procedures result in a continuous score, called the Composite Burn Index (CBI), which is based on visual observation of fuel consumption and depth of burn in several classes of fuels and vegetation.

In most situations, depth of burn is the primary factor of concern when assessing the impacts of fire on soil and water resources (fig. 1.7). Depth of burn relates directly to the amount of bare mineral soil exposed to rain-splash, the depth of lethal heat penetration, the depth at which a hydrophobic layer will form, the depth at which other chemical alterations occur, and the depth to which microbial populations will be affected. As such it affects many aspects of erodability and hydrologic recovery (Wright and Bailey 1982, DeBano and others 1998, Gresswell 1999, Pannkuk and others 2000). However, depth of burn is not the only controlling factor (Ryan 2002). For example, the surface microenvironment, shaded versus exposed, in surface fires versus crown fires can be expected to affect postfire species dynamics regardless of the depth of burn (Rowe 1983). In a surface fire, needles are killed by heat rising above the fire (Van Wagner 1973, Dickinson and Johnson 2001), thereby retaining their nutrients. Thus, litterfall of scorched needles versus no litterfall in crown fire areas can be expected to affect postfire nutrient cycling. Further, rainfall simulator experiments have shown that needle cast from underburned trees reduced erosion on sites where duff was completely consumed in contrast to crown fire areas with similar

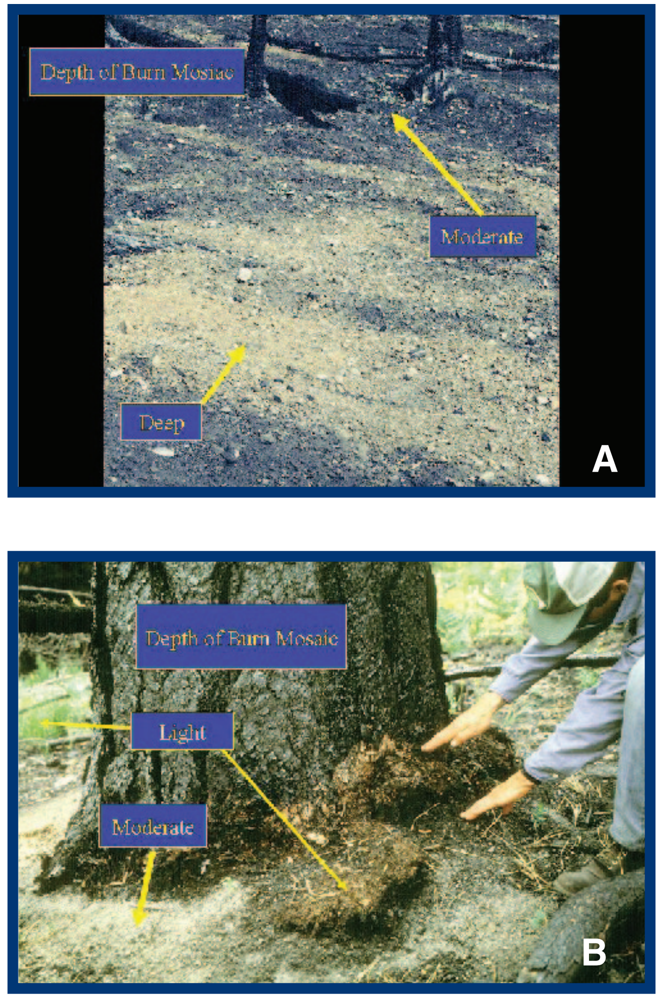

Figure 1.7—Depth of burn mosaics showing: (A) moderate to deep burn depths; and (B) light to moderate depths. (Photos by USDA Forest Service).

depth of burn (Pannkuk and others 2000). Thus, a lethal stand replacement crown fire (in other words, a fire that kills the dominant overstory; Brown 2000) represents a more severe fire treatment than a lethal stand replacement surface fire even when both have similar depth of burn (fig. 1.8). Thus for many ecological interpretations it is desirable to use the two-dimensional approach to rating fire severity to account for the effects of both the aboveground and belowground heat pulses.

A Conceptual Model

The previous discussion leads to development of a conceptual model to help planners, managers, and decisionmakers appreciate the spectrum of watershed responses to fire severity. The conceptual model describes fire severity as ranging from low water resource

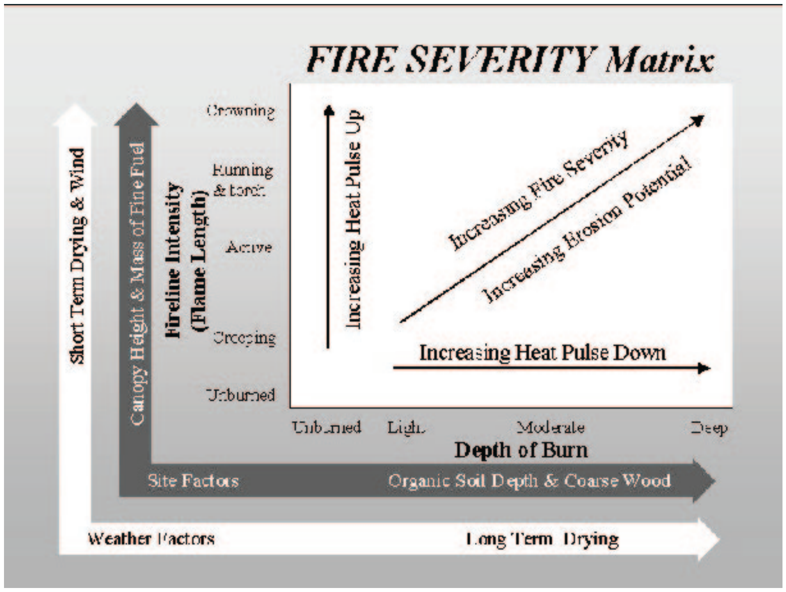

Figure 1.8—Site and weather factors associated with increasing fire severity and erosion potential.

responses likely to be experienced with a prescribed fire and no accompanying hydrologic events to high resource responses that could be expected from stand-replacing wildfire in forests and major storm events (fig. 1.9). Once again, fire severity is not directly related to fire intensity for the reasons discussed above.

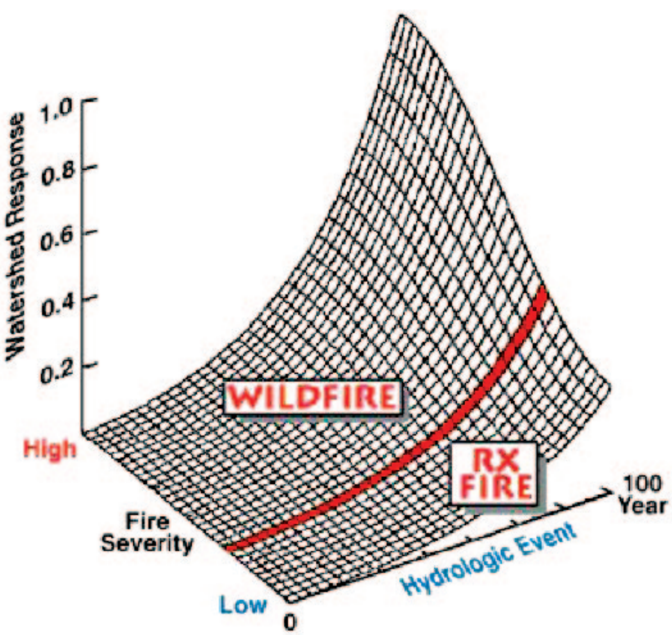

Figure 1.9—A conceptual model of watershed responses to fire severity.

Prescribed fire conditions (generally low fire severity) are depicted in figure 1.9 on the lower portion of the fire severity and resource response curve. These conditions are typically characterized by lower air temperature, higher relative humidity, and higher soil moisture burning conditions, where fuel loading is low and fuel moisture can be high. These conditions produce lower fire intensities and, as a consequence, lower fire severity leading to reduced potential for subsequent damage to soil and water resources. Prescribed fire, by its design, usually has minor impacts on these resources.

Fire at the other end of the severity spectrum (left side of fig. 1.9) more nearly represents conditions that are present during a wildfire, where temperatures, wind speeds, and fuel loadings are high, and humidity and fuel moisture are low. In contrast to prescribed burning, wildfire often has a major effect on soil and watershed processes, leading to increased sensitivity of the burned site to vegetative loss, increased runoff, erosion, reduced land stability, and adverse aquatic ecosystem impacts (Agee 1993, Pyne and others 1996, DeBano and others 1998).

Differences in watershed response along the spectrum between prescribed burning and wildfire or within an individual fire depend largely upon fire severity and the magnitude of hydrologic events following fire. Although significant soil physical, chemical, and biological impacts would occur with a stand-replacing wildfire, there would be no immediate watershed response in the absence of a hydrologic event. As indicated in figure 1.9, the magnitude of watershed response is keyed to the size (return period) of the hydrologic event or storm and the timing relative to fire. In addition to climate interactions with fire severity, topography also has a major influence on watershed response. Forested watersheds in mountainous regions of the West or East respond to storm events very differently than those in the Coastal Plain or Great Lakes where relief is at a minimum.

Fire-Related Disturbances

Fire

As discussed above, the primary disturbance to soil and water resources is a function of fire severity. Removal of vegetation alone is sufficient to produce significant soil and watershed responses (Anderson and others 1976, Swank and Crossley 1988, Neary and Hornbeck 1994). In areal extent and severity level, the disturbance produced by fire can be small or large, and uniform or a chaotic matrix. Beyond the initial vegetation and watershed condition impacts of the physical process of fire, suppression activities can add further levels of disturbance to both soils and water. These disturbances need to be evaluated along with those produced by fire.

Fire Suppression

Fire managers need to be aware of the soil and water impacts of suppression activities. The major soil and water disturbances associated with fire suppression are fire lines, roads, and fire retardants. Fire control lines created by hand or large equipment disturb the soil, alter infiltration, become sources of sediment, and can alter runoff patterns (see chapter 2). Suppression activities in boreal forests have been known to lead to decreasing permafrost depth resulting in downcutting and slope failures (Viereck 1982). Roads are already major sources of sediment in forest watersheds (Brown and Binkley 1994; Megahan 1984, Swift 1984). Temporary roads built during suppression operations increase the size of road networks and erosion hazard areas within watersheds. In addition, traffic from heavy equipment and trucks can deteriorate the surfaces of existing roads, making them more prone to erosion during rainfall. Although fire retardants are basically fertilizers, they can produce serious short-term water quality problems if dropped into perennial streams.

Type of Effects

The effects of fire on soil and water resources can be direct and indirect. They occur at many scales (microsite to ecosystem level), in different patterns, and over variable periods. In most watersheds of fire dependent or dominated ecosystems, fire impacts to soils and water are significant components and variable backgrounds of cumulative watershed effects. An understanding of these effects is important for land managers who deal with wildfire and use prescribed fire to accomplish ecosystem management objectives.

This Book's Objective

The objective of this volume in the Rainbow series is to provide an overview of the state-of-the-art understanding of the effects of fire on soils and water in wildland ecosystems. It is meant to be an information guide to assist land managers with fire management planning and public education, and a reference on fire effects processes, pertinent publications, and other information sources. Although it contains far more information and detailed site-specific effects of fire on soils and water than the original 1979 Rainbow volumes, it is not designed to be a comprehensive research-level treatise or compendium. That challenge is left to several textbooks (Chandler and others 1991, Agee 1993, Pyne and others 1996, DeBano and others 1998). The challenge in developing this volume was in providing a meaningful summary for North American fire effects on soils and water resources despite enormous variations produced by climate, topography, fuel loadings, and fire regimes.

Notes

Part A

Effects of Fire on Soil

Leonard F. DeBano
Daniel G. Neary

Part A—The Soil Resource: Its Importance, Characteristics, and General Responses to Fire

Introduction

Soil is a heterogeneous mixture of mineral particles and organic matter that is found in the uppermost layer of Earth's crust. The soil is formed as a product of the continual interactions among the biotic (faunal and floral), climatic (atmospheric and hydrologic), topographic, and geologic features of the environment over long periods (Jenny 1941, Singer and Munns 1996). Soils are important components of ecosystem sustainability because they supply air and water, nutrients, and mechanical support for the sustenance of plants. Soils also absorb water during infiltration. By doing so, they provide storage for water as well as acting as a conduit that delivers water slowly from upstream slopes to channels where it contributes to streamflow. There is also an active and ongoing exchange of gases between the soil and the surrounding atmosphere. When the infiltration capacity of the soil is exceeded, organic and inorganic soil materials are eroded and become major sources of sediment, nutrients, and pollutants in streams. These water and erosional processes are described in part B and chapters 5, 6, and 7 of this publication.

To fully evaluate the effects of fire on a soil, it is first necessary to quantitatively describe the soil and then to discuss the movement of heat through the soil during a fire (wildfires or prescribed burns). During the process of soil heating, significant changes can occur in the physical, chemical, and biological properties that are relevant to the future productivity and sustainability of sites supporting wildland ecosystems. This introductory part A presents a general discussion on the properties of soils and the heating processes occurring in soils during fires, and provides some general information on the physical, chemical, and biological responses to fire. This part is also intended as an extended executive summary for those interested in the general concepts concerning fire effects on soils. Readers who are interested in more detailed information on fire effects on soils are directed to indepth discussions of the individual physical, chemical, and biological properties and processes in soil that are affected by fire (see chapters 2, 3, and 4, respectively).

The General Nature of Soil

The features and the importance of soils are usually inconspicuous to the average person. Soil is simply the substrate that is walked on, that is used to grow trees or a garden, that creates a source of dust when the wind blows, that provides material that washes down the hillslope during runoff, or that is bared when the firefighter builds a fireline. However, closer examination shows that soil is a complex matrix made up of variable amounts of mineral particles, organic matter, air, and water. The inorganic constituents of soils contain a wide array of primary (for example, quartz) and secondary (for example, clays) minerals.

Organic matter is the organic portion of the soil and is made up of living and dead biomass that contains a wide range of plant nutrients. The living biomass in the soil consists of plant roots, microorganisms, invertebrates, and small and large vertebrate fauna that burrow in the soil. The nonliving organic matter is made up primarily of dead bark, large woody debris (dead trees, limbs, and so forth), litter, duff, and finely decomposed humus materials. Organic matter is broken down and decomposed by the actions of animals and microorganisms living in the soil. An important component of organic matter is humus, the colloidal soil organic matter (particles smaller than 3.9 to 20 x 10^{-6} inches [0.001 to 0.005 mm] in diameter) that decomposes slowly. Humus provides negative adsorption sites similar to clay minerals and also acts as an organic glue that helps to hold mineral soil particles together to form aggregates. This contributes to soil structure that creates pore space soil and provides passageways for the movement of air and water. The decomposition of organic matter also plays a central role in the cycling and availability of nutrients essential for plant growth. Organic matter also provides a source of energy necessary to support microbial populations in the soil.

Water and air occupy the empty spaces (pore space) created by the mineral-organic matter matrix in the soil. A delicate balance exists between the amount of pore space filled with water and that filled with air, which is essential for root respiration by living plants. Too much soil water can limit plant growth if the pore space is saturated with water (for example, waterlogged, anaerobic soils). In contrast, when too little water is available, plant growth can be limited by the lack of water necessary for transpiration and other physiological functions necessary for the growth of plants. Soil water also contains dissolved ions (cations and anions), and this is called the "soil solution." Many of the ions in the soil solution are absorbed by plant roots and used for plant growth.

A combination of the inorganic materials described above, along with variable amounts of finely divided and partially decomposed organic matter (humus), provides structure to the soil. Soil structure is the arrangement of the inorganic components of the soil into aggregates having distinctive patterns (for example, columnar, prismatic, blocky). These aggregates are stabilized by organic matter that provides an overall porous structure to the soil.

Soil Properties—Characteristics, Reactions, and Processes

The reader needs to be aware of some general definitions and terminology that are used in reference to soil properties when reading chapters 2, 3, and 4. The term "soil properties" is collectively used to include the characteristics, reactions, and processes that occur in soils. Traditionally, soils have been described in terms of physical, chemical, and biological properties. This classification is arbitrary, and in many cases the three classes of soil properties are not mutually exclusive but are so closely interconnected that it is impossible to clearly place a soil property in any one of the three categories. This interrelationship is particularly apparent in the discussions on organic matter and the different processes responsible for nutrient cycling. Because of this interdependency, we attempted to discuss the physical, chemical, and biological dimensions of nutrient cycling and organic matter separately, and then to cross-reference these discussions among the three general categories.

Soil Profile

Variable amounts and combinations of minerals, organic matter, air, and water produce a wide range of physical, chemical, and biological properties of a soil. However, these properties are not randomly distributed but occur in an orderly arrangement of horizontal layers called *soil horizons*. The arrangement of these layers extending from the surface litter downward to bedrock is referred to as the *soil profile*. A schematic profile is shown in figure A.1, and a real profile in figure A.2. Some profiles have distinct horizons as shown in figure A.2, but some soils have horizons that are not so distinct. The uppermost layers consist mainly of organic matter in various stages of decomposition. The surface litter layer (L-layer) is made up of undecomposed organic material that retains the features of the original plant material (leaves, stems, twigs, bark, and so forth). Immediately below the undecomposed layer is another organic layer that is in various stages of decomposition. It is called the fermentation layer (F-layer). In the F-layer, some of the original plant structure may still be discernable depending on the extent of decomposition. The lowermost surface organic matter layer is the humus layer (H-layer) that is completely

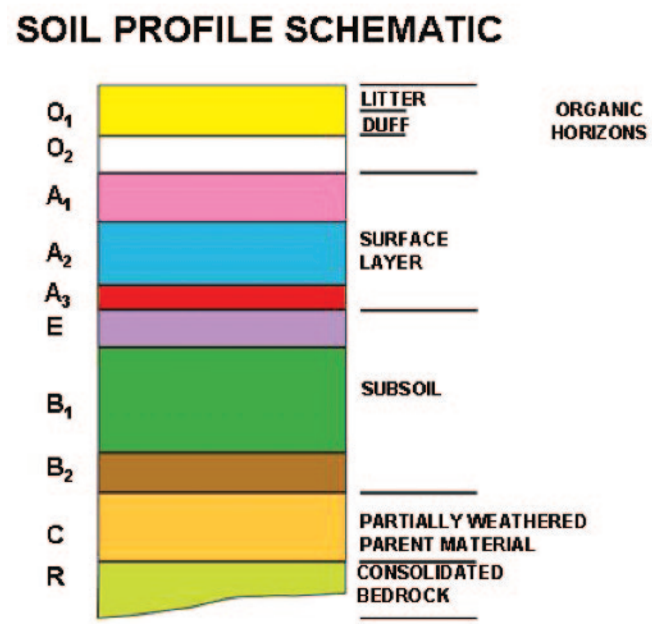

Figure A.1—A schematic of a well-developed (mature) soil profile showing a complete suite of the organic and inorganic soil horizons. (Figure courtesy of the USDA Forest Service, National Advanced Fire and Resource Institute, Tucson, AZ).

decomposed organic matter. The H-layer is an important site for nutrient availability and storage. The finely decomposed organic matter in the H-layer is also the source of aggregating substances that combine with the mineral soil particles in the upper inorganic horizons to produce soil structure. The original plant structure is no longer identifiable in the H-layer. The combined F-and H-layer is commonly referred to as the duff.

More recent designations have been developed for the L-, F-, and H-layers described above. Current taxonomic terminology refers to the organic horizon as the O horizon. The L-layer is referred to as the O_i or O_1 horizon. The F-layer is designated as the O_e, or part of the O_2 horizon and the H-layer is denser than the L- and F-layers and is designated as the O_a or O_2 horizon.

The mineral soil horizons begin with the uppermost part of the A-horizon and extend downward to bedrock. Depending upon the age and development of the soil profile, there can be several intermediate mineral horizons (for example, E-, B-, and C-horizon). The A-horizon is the top mineral layer, and the upper part of this horizon often contains large quantities of finely decomposed organic matter (humus). The mineral E-horizon is located immediately below the A-horizon. It is the site where substantial amounts of silicate, clay, iron, aluminum, carbonate, gypsum, or silicon are lost by weathering and leaching that occurs during soil development. Materials leached downward from the

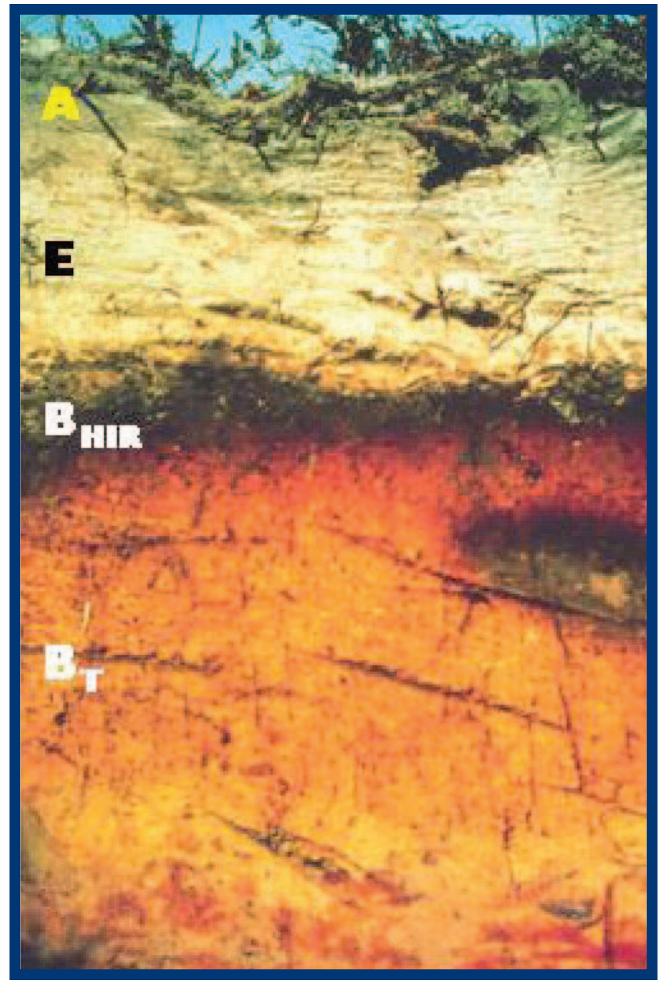

Figure A.2—Profile of a Pomona fine sand (Ultic Haplaquod, sandy, siliceous, hyperthermic family) from a slash pine (*Pinus elliottii*) stand in the flatwoods of northern Florida. (Photo by Daniel Neary).

E-horizon accumulate mainly in the B-horizon. In well-developed (mature) soil profiles, the original rock structure can no longer be recognized in the B-horizon. The C-horizon is unconsolidated parent rock material remaining above the R-horizon that is made up of hard consolidated bedrock.

Importance of Organic Matter

The effects of fire on soils cannot be fully evaluated unless the role of organic matter in the functioning and sustainability of soil ecosystems is understood. Organic matter is the most important soil constituent that is found in soils. Although it is concentrated on the soil surface where it makes up most of the L-, F-, and H-layers, it also plays an important role in the properties of the underlying mineral soil horizons. Ecologically, organic matter plays three major roles:

- Organic matter enhances the structure of soils. The most important physical function of organic matter is its role in creating and stabilizing soil aggregates. A porous well-structured soil is essential for the movement of water, air, and nutrients through soils and as a result organic matter contributes directly to the productivity and sustainability of wildland ecosystems.
- Chemically, organic matter maintains and regulates the biogeochemical cycling of nutrients by providing an active medium for sustaining numerous chemical and biological transformations. As such, it plays a key role in the productivity of plant ecosystems. Its specific roles in nutrient cycling include: providing a storage reservoir for all plant nutrients, maintaining a balanced supply of available nutrients, creating a large cation exchange capacity for storing available nutrients in soils, and functioning as a chelating agent for essential plant micronutrients (for example, iron).
- Soil provides a habitat for plant and animal organisms that range in size from bacteria and viruses to small mammals. The microbiological populations in the soil are usually inconspicuous to most observers although the soil can be teeming with hundreds of millions of microorganisms in each handful of forest soil. Their activity and diversity far exceeds that of the other biological components of forest ecosystems (for example, vegetation, insects, wildlife, and so forth). Soil organic matter is of particular importance to soil microorganisms because it provides the main source of energy for sustaining soil microorganisms. Soil microorganisms are involved in nearly all of the processes responsible for the cycling and availability of nutrients such as decomposition, mineralization, and nitrogen (N) fixation.

Fire Effects—General Concepts and Relationships

The effects of fire on soil properties must be evaluated within the concept of a complex organic and inorganic matrix of the soil profile described above. The magnitude of change occurring during a fire depends largely upon the level of fire severity, combustion and heat transfer, magnitude and depth of soil heating, proximity of the soil property to the soil surface, and the threshold temperatures at which the different soil properties change.

Severity and Fire Intensity

When discussing the effects of fire on the soil resource it is important to differentiate between fire intensity and fire severity because frequently they are not the same (Hartford and Frandsen 1992). Fire intensity is a term that is used to describe the rate at which a fire produces thermal energy (Brown and Davis 1973, Chandler and others 1991). *Fire intensity* is most frequently quantified in terms of fireline intensity because this measure is related to flame length, which is easily measured (DeBano and others 1998). *Fire severity*, on the other hand, is a more qualitative term that is used to describe ecosystem responses to fire and is particularly useful for describing the effects of fire on the soil and water system (Simard 1991). Severity reflects the amount of energy (heat) that is released by a fire and the degree that it affects the soil and water resources. It is classified according to postfire criteria on the site burned and has been classified into low, moderate, and high fire severity. Detailed descriptions of the appearances of these three severity levels for timber, shrublands, and grasslands are presented in table 1.4, Parts A,B,C respectively. (Note: this table and the parts can be found on pages 11 and 12 in Chapter 1).

The level of fire severity depends upon:

- Length of time fuel accumulates between fires and the amount of these accumulated fuels that are combusted during a fire (Wells and others 1979).
- Properties of the fuels (size, flammability, moisture content, mineral content, and so forth) that are available for burning
- The effect of fuels on fire behavior during the ignition and combustion of these fuels.
- Heat transfer in the soil during the combustion of aboveground fuels and surface organic layers.

High intensity fires can produce high severity changes in the soil, but this is not always the case. For example, low intensity smoldering fires in roots or duff can cause extensive soil heating and produce large changes in the nearby mineral soil. In contrast, high severity crown fires may not cause substantial heating at the soil surface because they sweep so rapidly over a landscape that not much of the heat generated during combustion is transferred downward to the soil surface.

Combustion

Energy generated as heat during the combustion of aboveground and surface fuels provides the driving force that causes a wide range of changes in soil properties during a fire (DeBano and others 1998). *Combustion* is the rapid physical-chemical destruction of organic matter that releases the large amounts of energy stored in fuels as heat. These fuels consist of dead and live standing biomass, fallen logs, surface litter (including bark, leaves, stems, and twigs), humus, and sometimes roots. During the combustion process heat and a mixture of gaseous and particulate byproducts are released. Flames are the most visual characteristic of the combustion process.

Three components are necessary in order for a fire to ignite and initiate the combustion process (Countryman 1975). First, burnable fuel must be available. Second, sufficient heat must be applied to the fuel to raise its temperature to the ignition point. And last, sufficient oxygen (O_2) is needed to be present to keep the combustion process going and to maintain the heat supply necessary for ignition of unburned fuel. These three components are familiar to fire managers as the fire triangle.

A common sequence of physical processes occurs in all these fuels before the energy contained in them is released and transferred upward, laterally, or downward where it heats the underlying soil and other ecosystem components. There are five physical phases during the course of a fire, namely: preignition, ignition, flaming, smoldering, and glowing (DeBano and others 1998). These different phases have been described in more detail by several authors (Ryan and McMahon 1976, Sandberg and others 1979, Pyne and others 1996). Preignition is the first phase when the fuel is heated sufficiently to cause dehydration and start the initial thermal decomposition of the fuels (pyrolysis). After ignition, the three phases of combustion that occur are flaming, smoldering, and glowing combustion. When active flaming begins to diminish, smoldering increases, and combustion diminishes to the glowing phase, which finally leads to extinction of the fire.

Heat Transfer

Heat produced during the combustion of aboveground fuels (for example, dead and live vegetation, litter, duff) is transferred to the soil surface and downward through the soil by several heat transfer processes (radiation, convection, conduction, vaporization, and condensation). *Radiation* is the transfer of heat from one body to another, not in contact with it, by electromagnetic wave motion; it increases the molecular activity of the absorbing substance and causes the temperature to rise (Countryman 1976b). *Conduction* is the transfer of heat by molecular activity from one part of a substance to another, or between substances in contact, without appreciable movement or displacement of the substance as a whole (Countryman 1976a). *Convection* is a process whereby heat is transferred from one point to another by the mixing of one portion of a fluid with another fluid (Chandler and others 1991). Vaporization and condensation are important in fire behavior and serve as a coupled reaction facilitating more rapid transfer of heat through soils. *Vaporization* of water occurs when it is heated to a temperature at which it changes from a liquid to a gas. *Condensation* occurs when water changes from a gas to a liquid with the simultaneous release of heat. The coupled reaction of vaporization and condensation provides a mechanism for the transfer of both water and organic materials through the soil during fires (DeBano and others 1998).

The mechanisms for the transfer of heat through different ecosystems components vary widely (table A.1). Although heat is transferred in all directions, large amounts of the heat generated during a fire are lost into the atmosphere (along with smoke, gases, and particulate matter generated by fire) by radiation, convection, and mass transfer (DeBano and others 1998). It has been estimated that only about 10 to 15 percent of the heat energy released during combustion of aboveground fuels is absorbed and transmitted directly downward to the litter and duff, or mineral soil if surface organic layers are absent. This occurs mainly by radiation (DeBano 1974, Raison and others 1986). Within the fuels themselves most of the heat transfer is by radiation, convection, and mass transfer. Meanwhile in the soil, convection and vaporization and condensation are the most important mechanisms for heat transfer in a dry soil. In a wet or moist soil, conduction can contribute significantly to heat transfer. The heat transfer processes occurring in the soil are described in more detail in chapter 2 of this publication. The transfer of heat through mineral soil is important because it causes soil heating and produces changes in the soil physical, chemical, and biological properties described in chapters 2, 3, and 4, respectively.

Depth and Magnitude of Soil Heating

As heat is transferred downward into and through the soil, it raises the temperature of the soil. The greatest increase in temperature occurs at, or near, the soil surface. Within short distances downward in the soil, however, the temperature increases quickly diminish so that within 2.0 to 3.9 inches (5 to 10 cm) of the soil surface the temperatures are scarcely above ambient temperature. A diagram of the heat increases with depth is called a temperature profile and is useful for determining the amount of change that occurs in a soil during a fire as the result of heating. The magnitude of these temperature increases depends on the severity of the fire as described above. Residence time of the fire (the duration of heating) is a particularly important feature of fires, affecting the depth and magnitude of soil heating. Detailed information on temperature profiles that can develop during grassland, shrubland, and forest fires is presented in chapter 2.

Table A.1—Importance of different heat transfer mechanisms in the transfer of heat within different ecosystem components.

Heat transfer mechanism	Ecosystem component	Importance to heat transfer
Radiation	Air	Medium
	Fuel	High
	Soil	Low
Conduction	Air	Medium
	Fuel	Low
	Soil	Low (dry), high (wet)
Convection	Air	High
	Fuel	Medium
	Soil	Low
Mass transfer	Air	High
	Fuel	Low
	Soil	Low
Vaporization/condensation	Air	Low
	Fuel	Medium
	Soil	High

Temperature Thresholds of Soil Properties

An important feature when assessing the effect of fire on soil properties is the temperature at which nutrients are volatilized or that irreversible damage occurs to a particular soil property. This temperature is called the threshold temperature (DeBano and others 1998). Temperature thresholds have been identified for numerous physical, chemical, and biological properties. The ranges of temperatures over which some common soil properties change in response to soil heating are displayed in figure 1.6. These temperature thresholds have been classified into three general classes, namely:

- Relatively insensitive soil properties that do not change until temperatures have reached over about 842 °F (450 °C). This class includes clays, cations (calcium, magnesium, potassium) and other minerals such as manganese.
- Moderately sensitive soil properties that are changed at temperatures between 212 and 752 °F (100 and 400 °C). Materials belonging to this class include sulfur, organic matter, and soil properties dependent upon organic matter.
- Sensitive soil properties are those that are changed at temperatures less than 212 °F (100 °C). Examples of sensitive materials are living microorganisms (for example, bacteria, fungi, mychorrizae), plant roots, and seeds. This class also includes many of the biologically mediated nutrient cycling processes in soils.

Threshold values for specific physical, chemical, and biological soil properties are described in greater detail in chapters 2, 3, and 4, respectively.

Location of Soil Properties

The natural differentiation of the soil profile into horizons creates a stratification of the physical, chemical, and biological soil properties discussed above. Understanding this stratified arrangement is necessary in order to accurately assess the effects of fire and soil heating on the different soil properties. The soil properties near or on the soil surface are the most directly exposed to heat that is radiated downward during a fire. Soil heating generally decreases rapidly with soil depth in a dry soil because dry soil is a poor conductor of heat.

The organic horizons that make up the forest floor are particularly important when discussing fire effects, because they are directly subjected to heat produced by burning of surface fuel, and they contain a large proportion of the organic matter found in soil profiles (DeBano and others 1998). Although some of the individual nutrients contained in the organic matter may not be volatilized, others such as N are vaporized in direct proportion to the amount of organic material lost. Most of the fire effects produced during surface fires occur in the upper organic horizons, or in the top part of the A-horizon. Heating of the B-horizon and deeper in the soil profile occurs only when roots are ignited and create localized subsurface heating.

Assessing Fire Effects on Soils

The above general information on soils along with the detailed information given in chapters 2, 3, and 4 on physical, chemical, and biological soil properties, respectively, can be used to assess fire effects on soils. This assessment requires being able to quantify the effect of fire and associated soil heating on soil properties and includes three main steps:

- First, the amount of energy radiated downward during combustion of fuels must be estimated. This energy is the driving force responsible for producing changes in soil properties. In general, the magnitude of change in individual soil properties is largely dependent upon the amount of energy radiated onto the soil surface, and subsequently transferred downward into the underlying duff and mineral soil. This radiated heat increases the temperature and causes changes in organic matter and other soil properties. Therefore, the postfire appearance of vegetation, litter, duff, and upper soil horizons can be used to estimate the amounts of surface heating and used to classify fire severity as low, moderate, or high. The basic assumption used in this technique is that as the amount of heat radiated downward increases, the severity increases from low to moderate to high (in other words, the magnitude of change in the soil property increases). An earlier discussion describing fire severity provides the necessary framework for establishing the severity of the fire in different ecosystems.
- After the fire severity has been established it can be used to estimate soil temperatures that develop when different ecosystems are burned (for example, grassland, shrubland, forests). Representative soil temperatures for different severities of burning for different ecosystems are discussed in chapter 2.
- Finally, once the approximate soil temperatures have been established, the changes in specific soil properties can be estimated using temperature threshold information. The percentage loss of different nutrients can be used along with estimates of the quantities of nutrients affected by fire to estimate the total nutrient losses, or gains, which occurred on a specific site during a fire. Specific information on the temperatures at which different physical, chemical, and biological soil properties changes are give in chapters 2, 3, and 4, respectively.

Management Implications

The condition of the soil is a key factor in the productivity of forest ecosystems and the hydrologic functioning of watersheds. Cumulative impacts that occur in soils as a result of fire can manifest themselves in significant changes in soil physical, chemical, or biological properties. These include breakdown in soil structure, reduced moisture retention and capacity, development of water repellency, changes in nutrient pools cycling rates, atmospheric losses of elements, offsite erosion losses, combustion of the forest floor, reduction or loss of soil organic matter, alterations or loss of microbial species and population dynamics, reduction or loss of invertebrates, and partial elimination (through decomposition) of plant roots. Although the most serious and widespread impacts on soils occur with stand-replacing wildfires, prescribed fires sometimes produce localized problems. Managers need to be aware of the impacts that fire can have on soil systems, and that these impacts can lead to undesired changes in site productivity, sustainability, biological diversity, and watershed hydrologic response.

Land managers need to be aware that some changes in soil systems after fire are quite obvious (for example, erosion, loss of organic matter), but others are subtle and can have equal consequences to the productivity of a landscape. For example, carbon and N are the key nutrients affected by burning. The significance of these changes is directly tied to the productivity of a given ecosystem. With a given change in N capital, the productivity of a nutrient-rich soil system might not significantly change following burning. A similar loss in N capital in a nutrient stressed system could result in a much greater change in productivity. Recovery of soil nutrient levels after fires can be fairly slow in some ecosystems, particularly those with limited N; and in semiarid regions such as the Southwestern United States and Northern Mexico nutrient fixation and turnover rates are slow.

Summary

This introductory section to the chapters in part A has provided information on the general nature of soil systems, some of the important soil properties, the character of soil profiles, and important constituents such as organic matter. It also introduces key concepts of heat transfer to soils and thresholds for important soil properties. The three chapters in part A address in greater detail fire effects on individual physical, chemical, and biological properties and processes in soil systems (see chapters 2, 3, and 4, respectively).

Notes

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Chapter 2: Soil Physical Properties

Introduction

Soil physical properties are those characteristics, processes, or reactions of a soil that are caused by physical forces that can be described by, or expressed in, physical terms or equations (Soil Science Society of America 2001). These physical properties (including processes) influence the mineral component of the soil and how it interacts with the other two components (chemical and biological). Plants depend on the physical characteristics of soils to provide the medium for growth and reproduction. Fire can produce significant changes in the soil that profoundly affect the ecology of plants (Whelan 1995). The effect of fire on individual soil physical properties depends on the inherent stability of the soil property affected and the temperatures to which a soil is heated during a fire. The physical mechanisms responsible for heat transfer into soils are also discussed in this chapter along with the temperatures that develop during different severities of burning in several wildland ecosystems. The relationships between soil physical properties affected by fire and erosional processes are also reviewed.

Soil Physical Characteristics

Important physical characteristics in soil that are affected by soil heating include: texture, clay content, soil structure, bulk density, and porosity (amount and size). The threshold temperatures for these soil physical characteristics are given in table 2.1. Physical properties such as wettability and structure are affected at relatively low temperatures, while quartz sand content, which contributes to texture, is affected least and only at the most extreme soil temperatures.

Soil Texture and Mineralogy

Soil texture is based on the relative proportion of different-sized inorganic constituents that are found in the 0.08 inch (less than 2 mm) mineral fraction of the mineral soil (DeBano and others 1998). Several soil textural classes have been specified according to the relative proportions of sand (0.05 to 2 mm in diameter), silt (0.002 to 0.05 mm in diameter), and clay (less than 0.002 mm in diameter) particles in the soil. Various proportions of the sand, silt, and clay

Table 2.1—Temperature thresholds for several physical characteristics of soil.

Soil characteristic	Threshold temperature		Source
	°F	*°C*	
Soil wettability	482	250	DeBano and Krammes 1966
Soil structure	572	300	DeBano 1990
Calcite formation	572-932	300-500	Iglesias and others 1997
Clay	860-1,796	460-980	DeBano 1990
Sand (quartz)	2,577	1,414	Lide 2001

fractions are used as the basis for identifying 12 textural classes (for example, sand, sandy loam, clay loam, silt loam). Clays are small-diameter silicate minerals having complex molecular structures that contribute to both the physical and chemical properties of a soil.

The components of soil texture (sand, silt, and clay) have high temperature thresholds and are not usually affected by fire unless they are subjected to high temperatures at the mineral soil surface (A-horizon). The most sensitive textural fraction is clay, which begins changing at soil temperatures of about 752 °F (400 °C) when clay hydration and clay lattice structure begin to collapse. At temperatures of 1,292 to 1,472 °F (700 to 800 °C), the complete destruction of internal clay structure can occur. However, sand and silt are primarily quartz particles that have a melting point of 2,577 °F (1,414 °C; Lide 2001). Only under extreme heating do quartz materials at the soil surface become fused. When fusion does occur, soil texture becomes more coarse and erodible. As a result, temperatures are rarely high enough to alter clays beyond a couple centimeters below the mineral soil surface. The effect of soil heating on the stability of clays is further mitigated by the concentration of clays during soil development in the B-horizons. These horizons are usually far removed from heating at the soil surface and rarely increase above ambient surface temperatures unless heated by smoldering roots.

The effect of soil heating on soil minerals other than clays has been studied to a limited extent. For example, a study on the effect of burning logs and slash piles on soil indicated that substantial changes can occur in the mineralogy of the underlying soil during severe heating while burning in juniper (*Juniperus* spp.) and oak (*Quercus* spp.) woodlands (Iglesias and others 1997). Although changes in minerals occurred in the juniper stands, they did not occur in the soils under oak. Followup laboratory burning experiments were done on calcite formation and the alteration of vermiculite in the soils collected from the juniper and oak woodland sites. Temperatures required for calcite formation in oak soils in the laboratory were found to be 932 °F (500 °C) compared to 572 °F (300 °C) in juniper soils.

Soil Structure

Soil structure has long been recognized as an important soil characteristic that can enhance productivity and water relations in both agricultural and wildland soils (DeBano and others 1998). Improving soil structure facilitates the infiltration into and the percolation of water through the soil profile, thereby reducing surface runoff and erosion (see chapter 5). The interaction of organic matter with mineral soil particles that create soil structure also increases the cation adsorption capacities of a soil and nutrient-supplying capabilities of the soil (see chapter 3).

Soil structure is the arrangement of primary soil particles into aggregates having distinctive patterns (columnar, prismatic, blocky). Humus is an important component of soil structure because it acts as a glue that helps hold mineral soil particles together to form aggregates and thus contributes to soil structure, particularly in the upper part of the mineral soil at the duff-upper A-horizon interface (see fig. A.1). However, further downward in the soil profile (in the B-horizon), soil structure is more dependent on clay minerals and the composition of the cations found in the soil solution.

Soil structure created as a result of organic matter in the soil can easily be affected by fire for two reasons. First, the organic matter in a soil profile is concentrated at, or near, the soil surface where it is directly exposed to heating by radiation produced during the combustion of aboveground fuels. Second, the threshold value for irreversible changes in organic matter is low. Living organisms can be killed by temperatures as low as 122 to 140 °F (50 to 60 °C). Nonliving organic matter begins changing at 224 °F (200 °C) and is completely lost at temperatures of 752 °F (400 °C) (DeBano 1990). The loss of soil structure reduces both the amount and size of soil pore space, as is described below.

Soil structure can also be changed by physical processes other than fire, such as deformation and compression by freezing and thawing, as well as by wetting and drying. The abundance of cations in saline and alkali soils can provide an aggregating effect, leading to a strong prismlike structure. Hydrophobic substances discussed later in this chapter also tend to improve the stability of soil aggregates by increasing their resistance to disintegration (slaking) when wetted (Giovannini and Lucchesi 1983, Giovannini and others 1983).

Bulk Density and Porosity

Bulk density is the mass of dry soil per unit bulk volume (expressed in g/cm^3) and is related to *porosity*, which is the volume of pores in a soil sample (nonsolid volume) divided by the bulk volume of the sample. Pore space in soils controls the rates of water (soil solution) and air movement through the soil. Well-aggregated soils contain a balance of macropores, which are greater than 0.02 inch (greater than 0.6 mm) in diameter, and micropores, which are less than 0.02 inch (less than 0.6 mm) in diameter (Singer and Munns 1996). This balance in pore sizes allows a soil to transmit both water and air rapidly through macropores and retain water by capillarity in micropores. Macropores in the surface soil horizons are especially important pathways for infiltration of water into the soil and its subsequent percolation downward through the soil profile.

Soil aggregation improves soil structure, creates macropore space, and improves aeration, and as a result decreases bulk density. Pore space not only influences the infiltration and percolation of water through the soil, but the presence of large pores also facilitates heat transfer by convection, and vaporization and condensation.

Fire and associated soil heating can destroy soil structure, affecting both total porosity and pore size distribution in the surface horizons of a soil (DeBano and others 1998). These changes in organic matter decrease both total porosity and pore size. Loss of macropores in the surface soil reduces infiltration rates and produces overland flow. Alteration of organic matter can also lead to a water repellent soil condition that further decreases infiltration rates. The scenario occurring during the destruction of soil structure by fire is:

- The soil structure collapses and increases the density of the soil because the organic matter that served as a binding agent has been destroyed.
- The collapse in soil structure reduces soil porosity (mainly macropores).
- The soil surface is further compacted by raindrops when surface soil particles and ash are displaced, and surface soil pores become partially or totally sealed.
- Finally, the impenetrable soil surface reduces infiltration rates into the soil and produces rapid runoff and hillslope erosion.

Physical Processes

The soil matrix provides the environment that controls several physical processes concerned with heat flow in soils during a fire. The results of heat transfer are manifested in the resulting soil temperatures that develop in the soil profile during a fire (Hartford and Frandsen 1992). Other soil physical processes affected by fire are infiltration rates and the heat transfer of organic substances responsible for water repellency.

Heat Transfer in Soils

The energy generated during the ignition and combustion of fuels provides the driving force that is responsible for the changes that occur in the physical, chemical, and biological properties of soils during a fire (Countryman 1975). Mechanisms responsible for heat transfer in soils include radiation, conduction, convection, mass transport, and vaporization and condensation (table A.1).

Radiation is defined as the transfer of heat from one body to another, not in contact with it, by electromagnetic wave motion (Countryman 1976b). Radiated energy flows outward in all directions from the emitting substance until it encounters a material capable of absorbing it. The absorbed radiation energy increases the molecular activity of the absorbing substance, thereby increasing its temperature.

Conduction is the transfer of heat by molecular activity from one part of a substance to another part, or between substances in contact, without appreciable movement or displacement of the substance as a whole (Countryman 1976a). Metals are generally good conductors in contrast to dry mineral soil, wood, and air that conduct heat slowly. Water as a liquid is a good conductor of heat up to the boiling point, and has an especially high capacity for storing heat until it evaporates.

Convection is a process whereby heat is transferred from one point to another by the mixing of one portion of a fluid with another fluid (Chandler and others 1991). Heat transfer by convection plays an important role the rate of fire spread through aboveground fuels. In soils, however, the complicated air spaces and interconnections between them provide little opportunity for the movement of heat through the soil by convection.

Vaporization and condensation are important coupled heat transfer mechanisms that facilitate the rapid transfer of heat through dry soils. Vaporization is the process of adding heat to water until it changes phase from a liquid to a gas. Condensation occurs when a gas is changed into a liquid with heat being released during this process. Both water and organic materials can be moved through the soil by vaporization and condensation.

Heat Transfer Pathways and Models—The heat that is generated by the combustion of surface and aboveground fuels is transferred to the mineral soil surface where it is transferred downward into the underlying soil by a series of complex pathways (fig. 2.1). Quantifying these different pathways for heat flow requires the mathematical modeling of fire behavior, duff ignition and combustion, and the transfer of heat downward to and through moist and dry mineral soil (Dimitrakopoulos and others 1994).

The heat radiated downward during the combustion of aboveground fuels is transferred either to the surface of the forest floor (path A), or directly to the surface of mineral soil if organic surface layers are absent (path B). In most forest ecosystems, heat is usually transferred to an organic layer of litter and duff (path A). When duff is ignited it can produce additional heat that is subsequently transferred to the underlying mineral soil (path D). More details concerning the influence of smoldering and burning duff

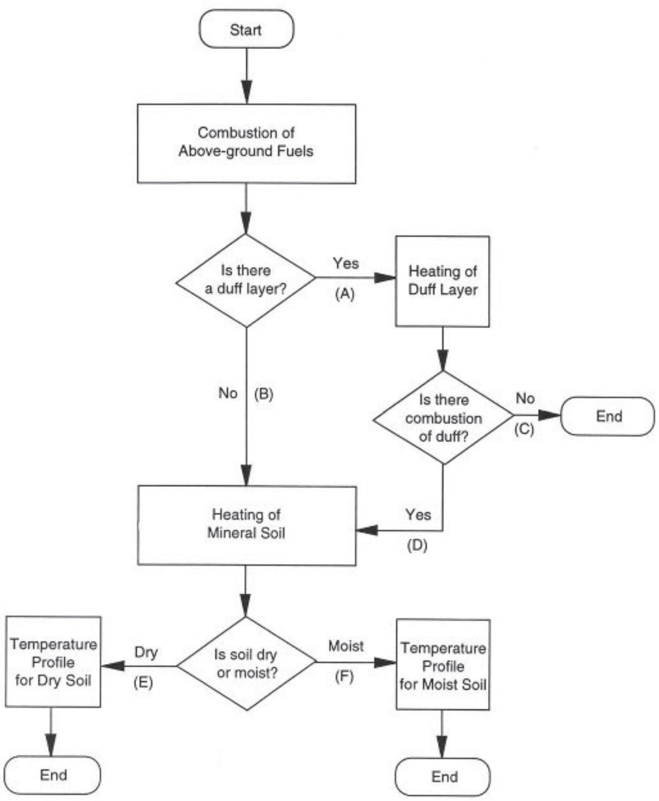


Figure 2.1—A conceptual model of heat flow pathways from combusting fuels downward through the litter into the underlying mineral soil. (Adapted from Dimitrakopoulos and others 1994. A simulation model of soil heating during wildfires. In Sala, M., and J. L. Rubio, editors. Soil Erosion as a Consequence of Forest Fires. Geoforma Edicones, Logrono, Spain, pp. 199-206. Copyright © 1994 Geoforma Edicones, Longro, Spain, ISBN 84-87779-14-X, DL 7353-1994).

on soil heating is presented below. If duff does not ignite, it does not heat the underlaying mineral soil (path C). The heat reaching the mineral soil is either transferred through a dry soil (path E) or a moist soil (path F). Dry soils are common during wildfires, whereas prescribed fires can be planned so as to burn over wet or dry soils. The temperature profiles that develop during heat transfer into moist and dry mineral soil vary widely and as a result affect the physical, chemical, and biological soil properties differently (DeBano and others 1998).

The temperature in moist soils does not rise much above 203 °F (95 °C) until all the water in a given soil layer has been vaporized. As a result, most of the chemical and physical properties of soil are not greatly affected by heating until the soil becomes dry. However, irreversible damage to living organisms near the soil surface is likely to occur because soil temperatures can easily be elevated above lethal temperatures 140 °F (60 °C) for seeds and microorganisms (see part A). Also, the lethal temperatures for microorganisms are lower for moist soils than for those that are dry.

The depth that heat penetrates a moist soil depends on the water content of the soil, and on the magnitude and duration of the surface heating during the combustion of aboveground fuels, litter, and duff (Frandsen 1987). During long-duration heating, such as that occurring under a smoldering duff fire or when burning slash piles, substantial heating can occur 40 to 50 cm downward in the soil. This prolonged heating produces temperatures that are lethal to soil organisms and plant roots. Increased thermal conductivity of moist soil may also create lethal temperatures at much deeper soil depths than if the soil was dry, due to increased thermal conductivity.

Organic-Rich Soils—Organic matter-rich soils are created when the primary productivity exceeds decomposition. Organic matter accumulations can vary from thick surface duff layers located mainly on the soil surface to deep deposits of peat that have been accumulating for thousands of years. The ignition and combustion of these organic-rich soils is of global concern because of the magnitude and duration of these fires and because of the severity of soil heating that occurs during these types of fires (fig. 2.2). The primary combustion process during these fires is by smoldering. The role of fires in wetlands is discussed further in chapter 8 of this publication.

General fire relationships: Although peatland soils are usually saturated, they can dry out during drought periods and become highly combustible. The fires that occur in peatland soils can be extremely long lasting and cover extensive areas where contiguous deposits of peat are present. Such was the case for one of the largest and longest burning fires in the world that occurred in Kalimantan, Indonesia (Kilmaskossu 1988,

Figure 2.2—Burn out of surface organic matter in the Seney National Wildlife Refuge of Michigan. (Photo by Roger Hungerford)

DeBano and others 1998). This fire was started during drought season when the farmers were clearing an area of logging slash by burning before planting agricultural crops. The fire started in 1983 and burned unchecked more-or-less continuously until the later part of the 1990s. The entire burn covered an area of over 8.4 million acres (3.4 million ha).

Although a thick accumulation of organic matter is commonly associated with tropical and semitropical ecosystems, the largest areas of organic soils are actually found in the boreal regions of the world. Although conditions are usually cool and moist in boreal forests, fires can occur periodically in underlying wetland soils during low rainfall years, at which time these fires mostly burn only the drier surface layers (Wein 1983). The combustion of peatland soils in the boreal forests during wildfires that are started by lightning have been identified as a major source of CO^2 that is released into the atmosphere. It has been estimated that greater than 20 percent of the atmospheric emissions linked to global warming are caused by these fires in the boreal forests worldwide (Conard and Ivanova 1997). During 1980s alone, more than 138.3 million acres (56 million ha) of boreal forest were estimated to have burned globally (Stocks 1991). Research is currently under way to develop better methods for quantifying the amounts of organic matter lost as the result of wildland fires in peatlands (Turetsky and Wieder 2001).

Soil heating pathways: When dealing with the combustion of organic soils it is important to understand the processes that sustain combustion. The heat produced by the combustion of aboveground fuels can be transferred to the duff (path A, fig. 2.1). Duff (or thick organic layers) can act as an insulating layer when it does not ignite (path C), or a heat source when it ignites, combusts, and continues smoldering (path D). Therefore, the amount of heat transferred into the underlying mineral soil depends on whether the duff burns and whether the smoldering duff acts as a long-term source of heat. The ignition and combustion of the duff is complex, and attempts to correlate it with heat produced during slash burning have been largely unsuccessful (Albini 1975). Although the duff complicates the heat transfer from the burning aboveground fuels into the underlying mineral soil, some features controlling duff ignition and combustion are known. Important variables needed to describe heat production in duff include depth, total amount, density of packing, the amount of inorganic constituents present, and the moisture content.

If duff does not burn, it provides a barrier to heat flow because the thermal conductivity of organic matter is low (path C, fig. 2.1). The probability of ignition in organic soils (including duff) depends on both inorganic constituents and moisture content (Hungerford and others 1995b). Organic soils are not necessarily completely organic matter, but instead they may contain variable amounts of mineral soil as a result of mixing by surface disturbance. The chances of ignition in organic soils decrease as mineral content increases at any given moisture content. Likewise, the chances of ignition of organic soil decrease as moisture content increases at any given mineral content. In general, duff burns more efficiently when the moisture content is below 30 percent. Varying amounts will burn at moisture contents from 30 to 150 percent, and it is too wet to burn when moisture contents exceed 150 percent (Brown and others 1985). Moisture affects both the thermal conductivity and the heat capacity of the duff, which in turn affect its ignitability (Hungerford 1990).

Combustion of the duff and thick organic deposits such as peat soils involves a smoldering reaction that is initiated by the ignition of a spot or several spots by fire brands, hot ash material, or radiated and conducted heat from the fire front (Pyne and others 1996). After duff ignites, it can transfer large amounts of heat into the underlying soil by convection, conduction, and radiation, and can raise the mineral soil above 350 °C for several hours. Therefore, it becomes difficult to quantify this combined heat flow into the underlying mineral soil. When thick layers of organic materials ignite, glowing combustion can also create an ash layer on the surface of the glowing duff. This ash layer retards heat dissipation upward, thereby causing more heat to penetrate into the soil (Sackett and Haase 1992). As a result, organic layers can transfer 40 to 73 percent of the heat generated during the smoldering process into the underlying mineral soil (Hungerford and Ryan 1996). The ignition, smoldering, and combustion of thick duff layers can continue for hours, thereby allowing substantial time for heat to be transferred deeply into the soil.

A duff burnout model: Combustion in duff (duff burnout) and organic soils was summarized by Hungerford and others (1995a,b). Ignition is initiated at a single point or several locations on the surface duff (fig. 2.3). Ignition can also occur in cracks or depressions in duff, or be caused by woody material that burns downward through the duff (fig. 2.3A). Fire burns both laterally and vertically after ignition (fig. 2.3B). Fire will burn laterally until it encounters incombustible conditions (moist organic matter, rocks, or the absence of duff). It burns vertically until it reaches mineral soil or moisture conditions that will not support combustion. During the smoldering of the fire, a hole develops in the burned-out organic layer. Horizontal spread of the fire can leave a thin unburned top crust (Pyne and others 1996). As the smoldering zone moves laterally and vertically, it creates a drying zone caused by the heat from the glowing zone, which allows the glowing front to advance until it reaches incombustible conditions (fig. 2.3C).

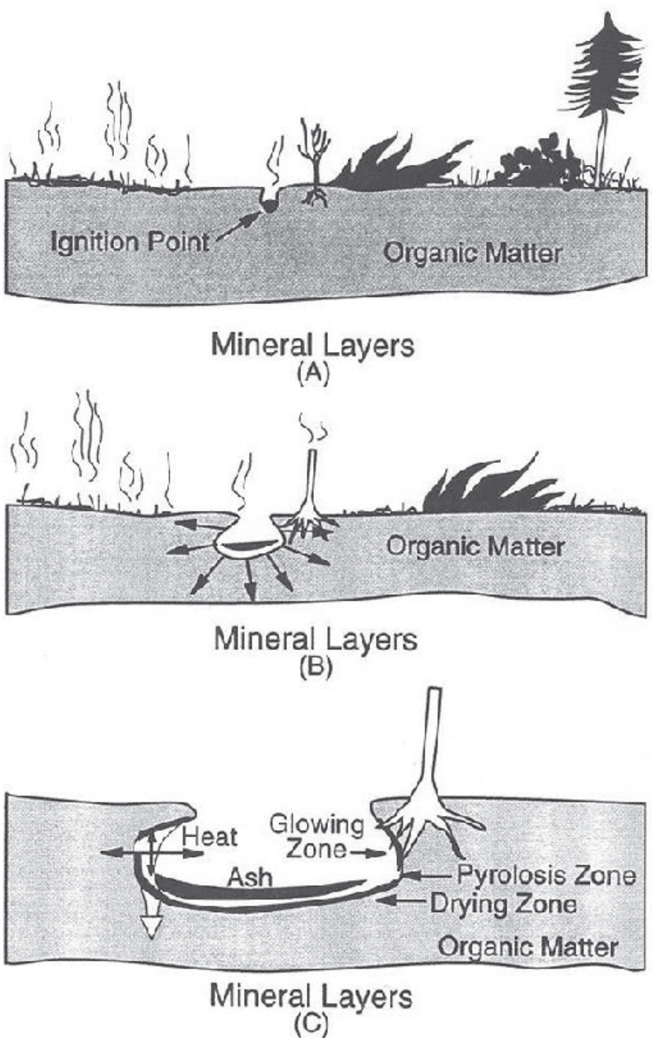


Figure 2.3—A schematic diagram of the smoldering process. The initial ignition point is created by the passing fire front. The fire spreads concentrically from the ignition point (A), develops concentric burned areas (B), and finally develops into a large burned out area (C). (Adapted from Hungerford and others 1995b).

Soil Temperature Profiles—Heat absorption and transfer in soils produce elevated temperatures throughout the soil. Temperature increases near the surface are greatest, and they are the least downward in the soil. These temperature regimes are called temperature profiles and can be highly variable depending mainly on the amount of soil water present. Dry soils are poor conductors of heat and thereby do not heat substantially below about 2 inches (5 cm) unless heavy long-burning fuels are combusted. In contrast, wet soils conduct heat rapidly via the soil water although temperatures remain at the boiling point of water until most of the water has been lost. The final soil temperatures reached vary considerably between fires (different fires may produce similar soil temperatures, and conversely, similar fires can produce widely different soil temperatures) and within fires because of heterogeneous surface temperatures.

Numerous reports describing soil temperatures during fire under a wide range of vegetation types and fuel arrangements are present in the literature. As a point of reference, some typical soil temperature profiles are presented for different severities of fire in grass, chaparral, and forests.

Soil temperature increases generated during a cool-burning prescribed fire in mixed conifer forests are low and of short duration (fig. 2.4). This type of fire would

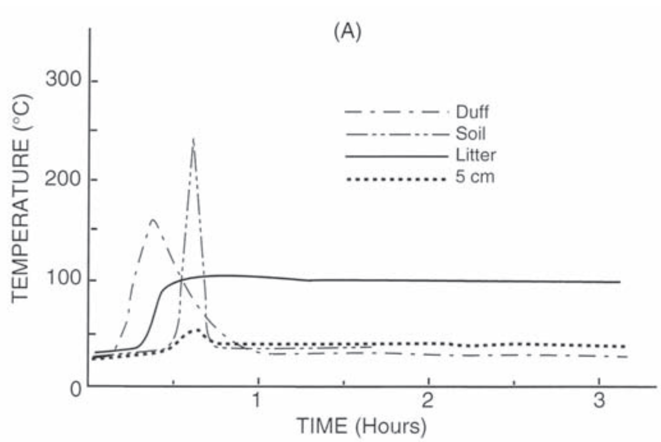


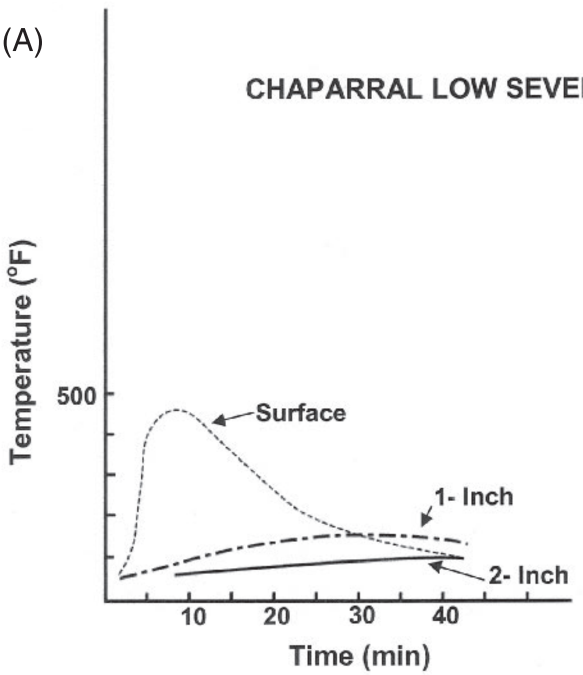
Figure 2.4—Surface and soil temperatures recorded under a cool-burning prescribed fire in mixed conifer forest. (Adapted from Agee 1973, University of California, Water Resources Center, Contribution 143).

be carried by the surface litter and would probably not consume much standing vegetation, although it might affect some smaller seedlings.

Fire behavior during brush fires, however, differs widely from that occurring during prescribed burning in forests. Both wildfires and prescribed fires in brush fields need to be carried through the plant canopy. The difference between wild and prescribed fires is mainly the amount and rate at which the plant canopy is consumed. During wildfires, the entire plant canopy can be consumed within a matter of seconds, and large amounts of heat that are generated by the combustion of the aboveground fuels are transmitted to the soil surface and into the underlying soil. In contrast, brush can be prescribe-burned under cooler burning conditions (for example, higher fuel moisture contents, lower wind speeds, higher humidity, lower ambient temperatures, using northerly aspects) such that fire behavior is less explosive. Under these cooler burning conditions the shrub canopy may be not be entirely consumed, and in some cases a mosaic burn pattern may be created (particularly on north-facing slopes). The soil temperature profiles that were measured during low, medium, and high severity fires in chaparral vegetation in southern California are presented in figure 2.5A, B, and C.

The highest soil temperatures are reached when concentrated fuels such as slash piles and thick layers of duff burn for long periods (fig. 2.6A and B). The soil temperatures under a pile of burning eucalyptus logs (fig. 2.6A) reached lethal temperatures for most living biota at a depth of almost 22 cm in the mineral soil (after Roberts 1965). In must be kept in mind, however, that this extreme soil heating occurred on only a small fraction of the area, although the visual effects on plant growth were observed for several years. An

(A)



(B)

(C)

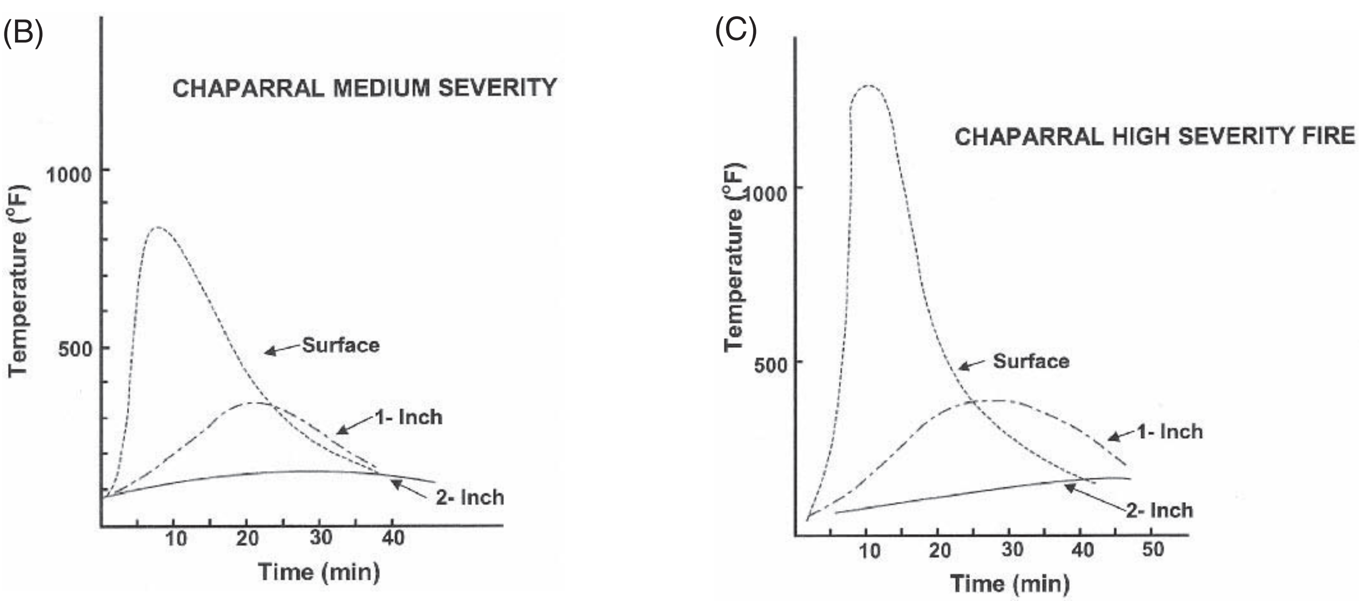
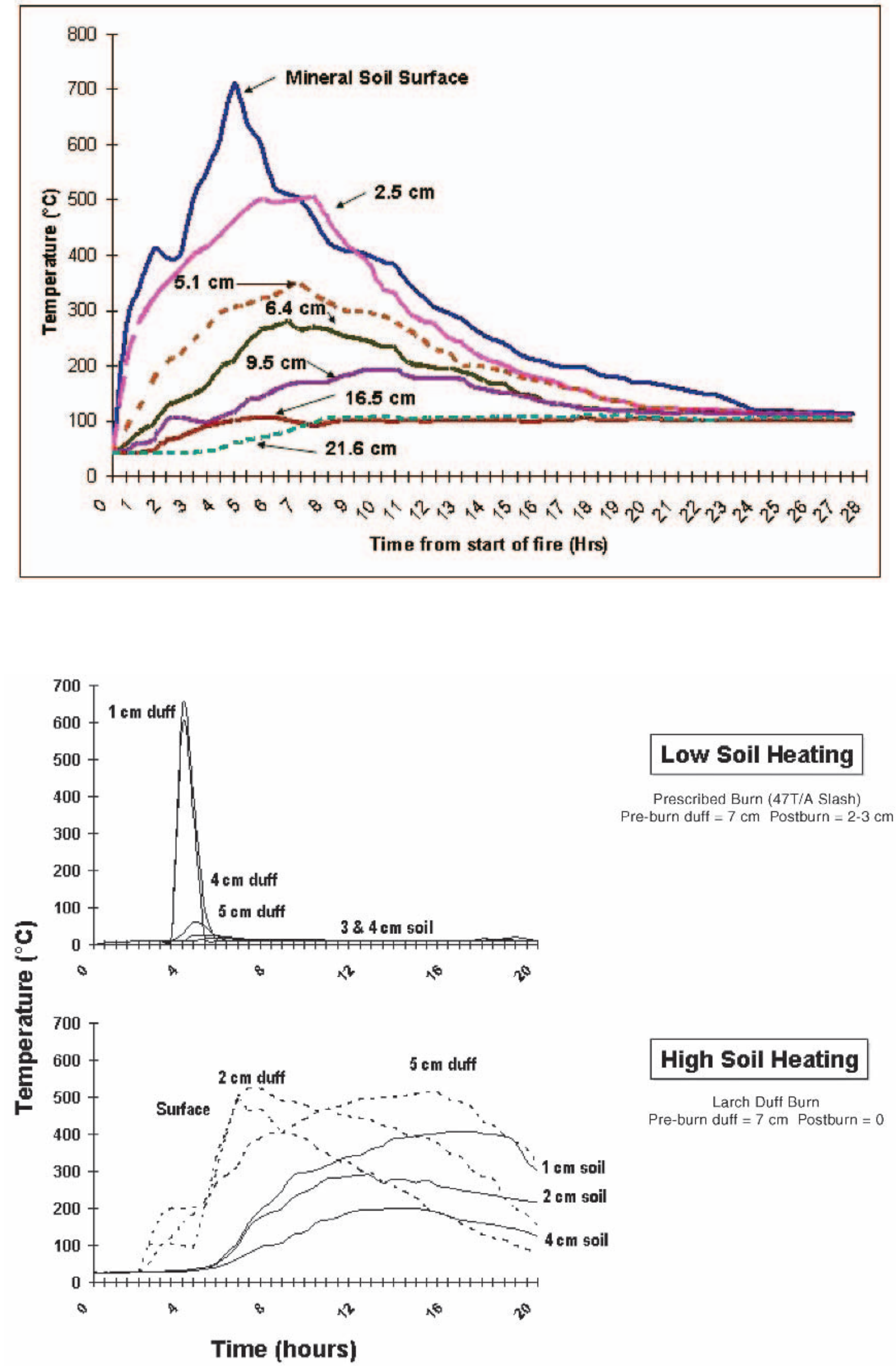


Figure 2.5—Soil temperature profiles during a low severity chaparral fire (A), a medium severity chaparral fire (B), and high severity chaparral fire (C). (After DeBano and others 1979)

(A)



(B)

Figure 2.6—Soil temperatures profiles under (A) windrowed logs (After W.B. Roberts. 1965. Soil temperatures under a pile of burning eucalyptus logs. Australian Forest Research 1(3):21-25), and (B) under a 7-cm (18 inch) duff layer in a larch forest. (After Hungerford 1990).

example of extensive soil heating that can occur during the burning of areas having large accumulations of duff and humus (fig. 2.6B) was reported during the complete combustion of 2.8 inches (7 cm) of a duff layer found under larch (Hungerford 1990).

Based on the information available on the relationship between soil heating and type of fire, the following generalities can be made:

- Crown fires are fast-moving, wind-driven, large, impressive, and usually uncontrollable, and they have a deep flame front (fig. 1.3). Usually little soil heating results when a fire front passes mainly through the tree crowns.
- Surface fires, compared to crown fires, are slower moving, smaller, patchy, and are more controllable, and they may also have a deep flame front (fig. 2.7A). These fires usually ignite and combust a large portion of the surface fuels in forests and brushlands that can produce substantial soil heating.
- Grass fires are fast-moving and wind-driven, may be large, and have a narrow flame front (fig. 2.7B). The amount of fuel available for burning in grasslands is usually much less than that contained in brushlands and forests, and as a result, soil heating is substantially less than occurs during surface or smoldering fires.
- Smoldering fires do not have flames, are slow-moving and unimpressive, but frequently have long burnout times. They generally are controllable although they may have a deep burning front. Soil heating during this long duration smoldering process may be substantial. Temperatures within smoldering duff often are between 932 and 1,112 °F (500 and 600 °C). The duration of burning may last from 18 to 36 hours, producing high temperatures in the underlying mineral soil.

Water Repellency

The creation of *water repellency* in soils involves both physical and chemical processes. It is discussed within the context of physical properties because of its importance in modifying physical processes such as infiltration and water movement in soils. Although hydrophobic soils had been observed since the early 1900s (DeBano 2000a,b), fire-induced water repellency was first identified on burned chaparral watersheds in southern California in the early1960s. Watershed scientists were aware of it earlier, but it had been referred to simply as the "tin roof" effect because of its effect on infiltration (fig. 2.8A, B, and C). In southern California both the production of a fire-induced water repellency and the loss of protective vegetative cover play a major role in the postfire runoff and erosion, and the area is particularly important because of the large centers of

Figure 2.7—Surface fire in (A) an uneven aged ponderosa pine forest, Mogollon Rim, Arizona, and (B) Alaskan grasslands. (Photos by USDA Forest Service).

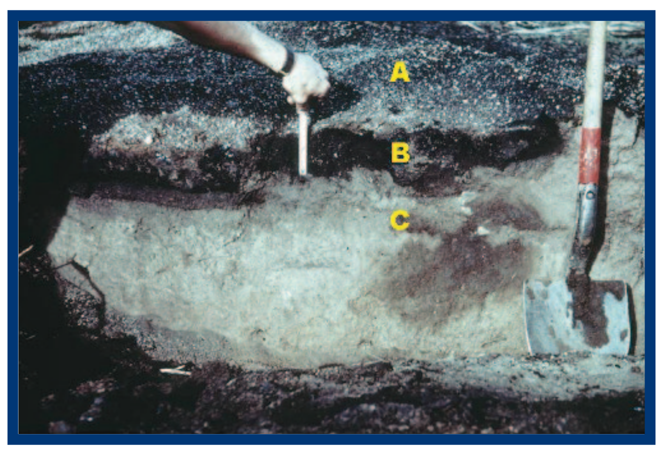


Figure 2.8—The "tin roof" effect on burned chaparral watersheds as described by earlier watershed researchers include (A) the wettable ash and carbon surface layer, (B) the discontinuous water repellent layer, and (C) the wettable subsoil. (After DeBano 1969).

populations located immediately below steep, unstable chaparral watersheds.

Nature of Water Repellency in Soils—Normally, dry soils have an affinity for adsorbing liquid and vapor water because there is strong attraction between the mineral soil particles and water. In water-repellent soils, however, the water droplet "beads up" on the soil surface where it can remain for long periods and in some cases will evaporate before being absorbed by the soil. Water, however, will not penetrate some soils because the mineral particles are coated with hydrophobic substances that repel water. Water repellency has been characterized by measuring the contact angle between the water droplet and the water-repellent soil surface. Wettable dry soils have a liquid-solid contact angle of nearly zero degrees. In contrast, water-repellent soils have liquid-solid contact angles around 90 degrees (fig. 2.9).

Causes of Water Repellency—Water repellency is produced by soil organic matter and can be found in both fire and nonfire environments (DeBano 2000a,b). Water repellency can result from the following processes involving organic matter:

- An irreversible drying of the organic matter (for example, rewetting dried peat).
- The coating of mineral soil particles with leachates from organic materials (for example, coarse-grained materials treated with plant extracts).
- The coating of soil particles with hydrophobic microbial byproducts (for example, fungal mycelium).
- The intermixing of dry mineral soil particles and dry organic matter.

Figure 2.9—Appearance of water droplets that are "balled up" on a water-repellent soil. (After DeBano 1981).

- The vaporization of organic matter and condensation of hydrophobic substances on mineral soil particles during fire (for example, heat-induced water repellency).

Formation of Fire-Induced Water-Repellent Soils— A hypothesis by DeBano (1981) describes how a water-repellent layer is formed beneath the soil surface during a fire, noting that organic matter accumulates on the soil surface under vegetation canopies during the intervals between fires. During fire-free intervals, water repellency occurs mainly in the organic-rich surface layers, particularly when they are proliferated with fungal mycelium (fig. 2.10A). Heat produced during the combustion of litter and aboveground fuels vaporizes organic substances, which are then moved downward into the underlying mineral soil where they condense in the cooler underlying soil layers (fig. 2.10B) The layer where these vaporized hydrophobic substances condense forms a distinct water-repellent layer below and parallel to the soil surface (fig. 2.10C).

The magnitude of fire-induced water repellency depends upon several parameters, including:

- The severity of the fire. The more severe the fire, the deeper the layer, unless the fire is so hot it destroys the surface organic matter.
- Type and amount of organic matter present. Most vegetation and fungal mycelium contain hydrophobic compounds that induce water repellency.
- Temperature gradients in the upper mineral soil. Steep temperature gradients in dry soil enhance the downward movement of volatilized hydrophobic substances.
- Texture of the soil. Early studies in California chaparral showed that sandy and coarse-textured soils were the most susceptible to fire-induced water repellency (DeBano 1981). However, more recent studies indicate that water repellency frequently occurs in soils other than coarse-textured ones (Doerr and others 2000).
- Water content of the soil. Soil water affects the translocation of hydrophobic substances during a fire because it affects heat transfer and the development of steep temperature gradients.

Effect of Water Repellency on Postfire Erosion—Fire affects water entering the soil in two ways. First, the burned soil surface is unprotected from raindrop impact that loosens and disperses fine soil and ash particles that can seal the soil surface. Second, soil heating during a fire produces a water-repellent layer at or near the soil surface that further impedes infiltration into the soil. The severity of the water repellency in the surface soil layer, however, decreases

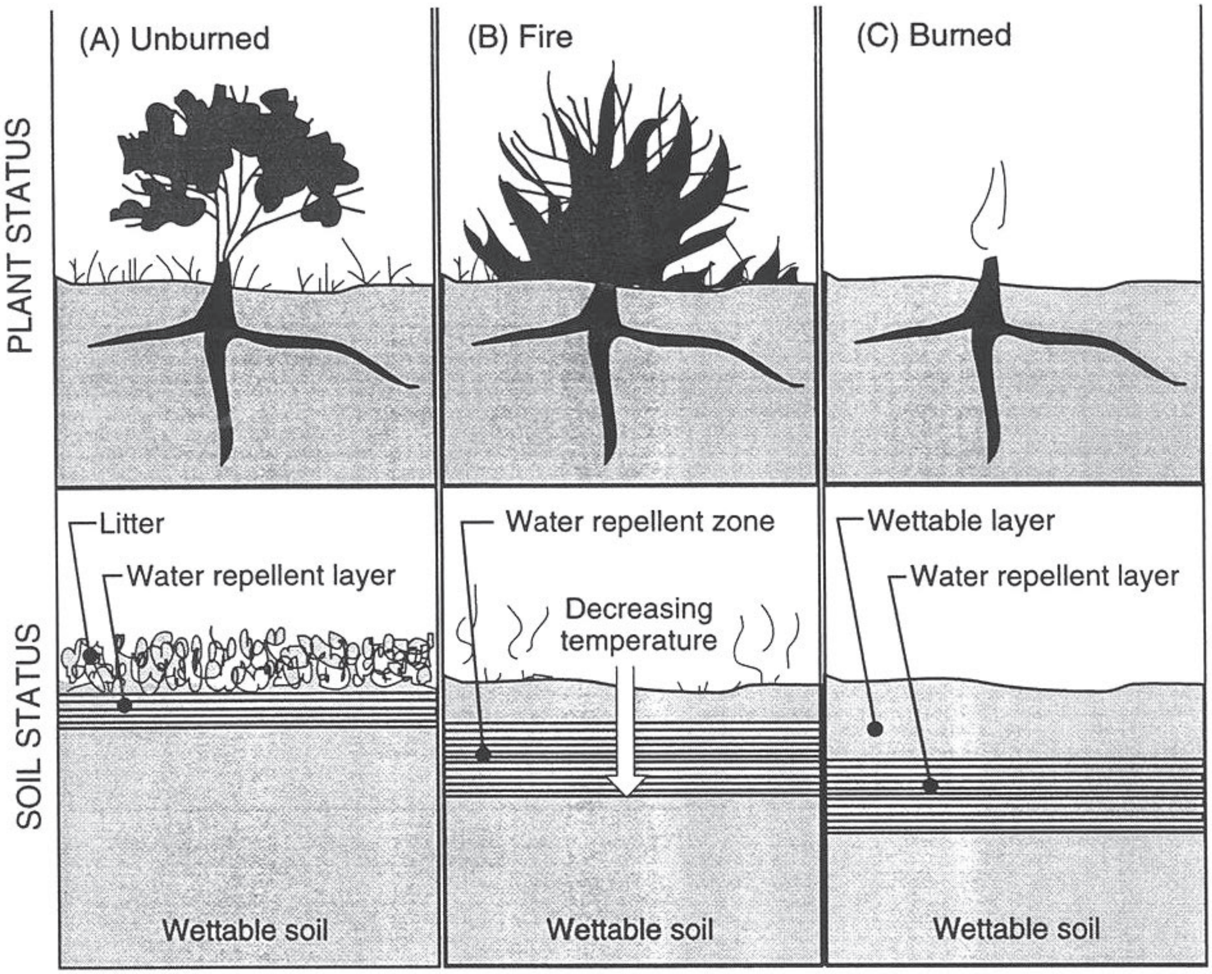


Figure 2.10—Formation of fire-induced water repellency. Water repellency before (A), during (B), and following (C) fire. (After DeBano 1981).

over time as it is exposed to moisture; in many cases, it does not substantially affect infiltration beyond the first year. More detailed effects of water repellency on the infiltration process are discussed in chapter 6. Water repellency has a particularly important effect on two postfire erosion processes, that of raindrop splash and rill formation.

Raindrop splash: When a water-repellent layer is formed at the soil surface, the hydrophobic particles are more sensitive to raindrop splash than those present on a wettable soil surface (Terry and Shakesby 1993). Consequently, raindrops falling on a hydrophobic surface produce fewer, slower moving ejection droplets that carry more sediment a shorter distance than in the case of a wettable soil. Further, the wettable surfaces have an affinity for water and thereby become sealed and compacted during rainfall, which makes them increasingly resistant to splash detachment. Conversely, the hydrophobic soil remains dry and noncohesive; particles are easily displaced by splash when the raindrop breaks the surrounding water film.

Rill formation: A reduction in infiltration caused by a water-repellent layer quickly causes highly visible rainfall-runoff-erosion patterns to develop on the steep slopes of burned watersheds. The increased surface runoff resulting from a water-repellent layer quickly entrains loose particles of soil and organic debris, and produces surface runoff that rapidly becomes concentrated into well-defined rills. As a result, extensive rill networks develop when rainfall exceeds the slow infiltration rates that are characteristic of water-repellent soils.

The sequence of rill formation as a result of fire-induced water repellency has been documented to follow several well-defined stages (Wells 1987). First, the wettable soil surface layer, if present, is saturated during initial infiltration (fig. 2.11A). Water infiltrates rapidly into the wettable surface ash layer until it is impeded by a water-repellent layer. This process occurs uniformly over the landscape so that when the wetting front reaches the water-repellent layer, it can neither drain downward or laterally (fig. 2.11A). If the water-repellent soil layer is on the soil surface, runoff begins immediately after rain droplets reach the soil surface. As rainfall continues, water fills all available pores until the wettable soil layer becomes saturated. Because of the underlying water-repellent layer, the saturated pores cannot drain, which creates a positive pore pressure above the water-repellent layer. This increased pore pressure decreases the shear strength of the soil mass and produces a failure zone located at the boundary between the wettable and water-repellent layers where pore pressures are greatest (fig. 2.11B). As the water flows down this initial failure zone, turbulent flow develops, which accelerates erosion and entrains particles from both the wettable ash layer if present and the water-repellent layer (fig. 2.11C and 2.11D). The downward erosion of the water-repellent rill continues until the water-repellent layer is eroded away and water begins infiltrating into the underlying wettable soil. Flow then diminishes, turbulence is reduced, and down-cutting ceases. The final result is a rill that has stabilized immediately below the water-repellent layer (fig.2.11E). On a watershed basis these individual rills develop into a well-defined network that can extend throughout a small watershed (fig. 2.11F).

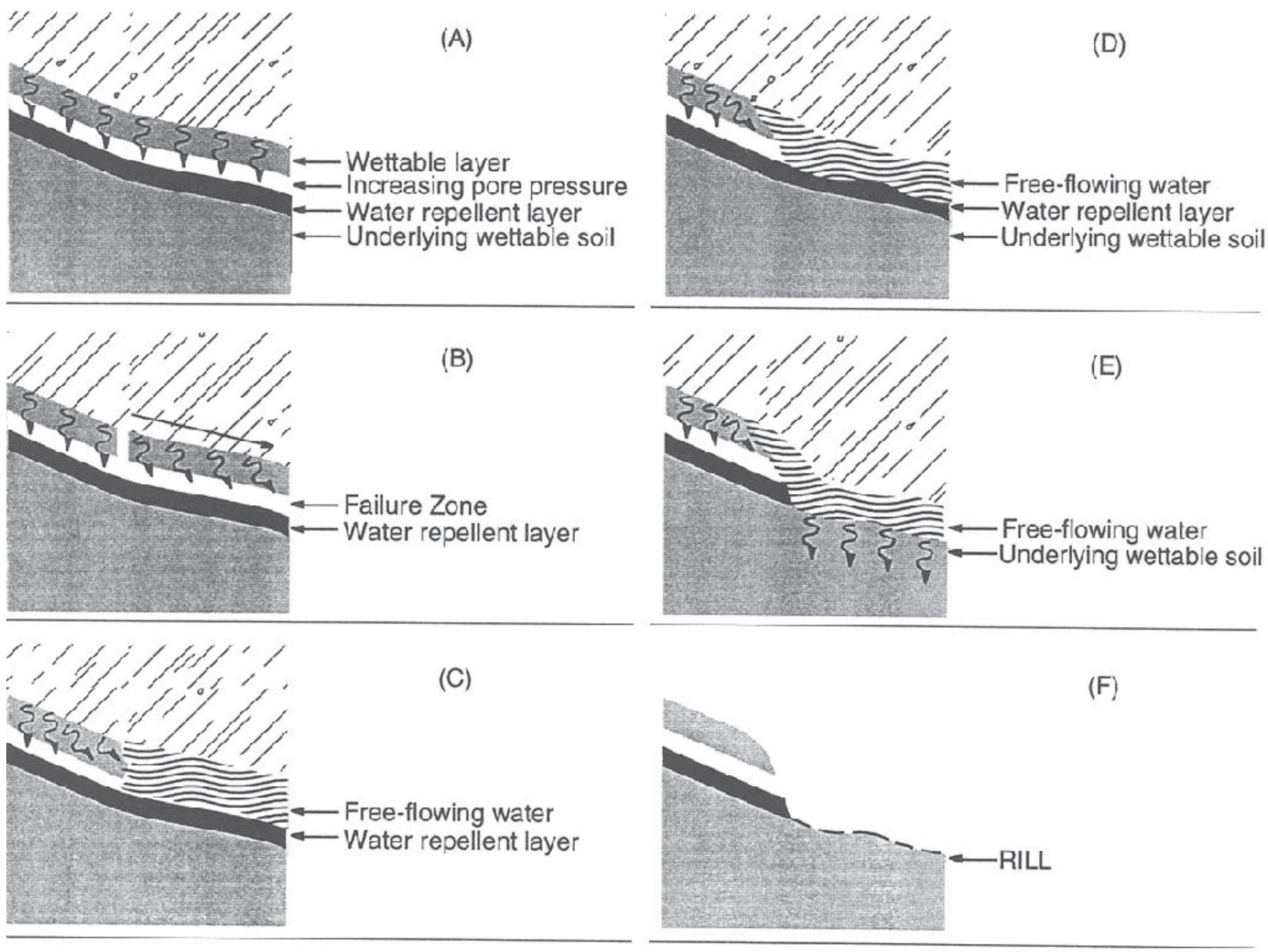


Figure 2.11—Sequence of rill formation on a burned slope with a water-repellent layer includes (A) saturation of wettable surface area, (B) development of a failure zone in wettable surface layer, (C) free flowing water over the water-repellent layer, (D) erosion of the water-repellent layer, (E) removal of the water repellent layer and infiltration into underlying wettable soil, and (F) resultant rill. (From Wells 1987. The effects of fire on the generation of debris flows in southern California. Reviews in Engineering Geology. 7:105-114. Modified with permission of the publisher, the Geological Society of America, Boulder, Colorado U.S.A. Copyright © 1987 Geological Society of America.)

Soil Erosion

Processes and Mechanics

The erosion process involves three separate components that are a function of sediment size and transport medium (water or air) velocity. These are: (1) detachment, (2) transport, and (3) deposition. Erosion occurs when sediments are exposed to water or air and velocities are sufficient to detach and transport the sediments. Table 2.2 gives a generalized breakdown of sediment classes and detachment/transport/deposition velocities in water.

Erosion is a natural process occurring on landscapes at different rates and scales depending on geology, topography, vegetation, and climate. Natural rates of erosion are shown in table 2.3. Geologic erosion rates were calculated on large basins so they are higher than those listed for forests. The data from forests come from much smaller watershed experiments. Natural erosion rates increase as annual precipitation increases, peaking in semiarid ecoregions when moving from desert to wet forest (Hudson 1981). This occurs because there is sufficient rainfall to cause natural erosion from the sparser desert and semiarid grassland covers. As precipitation continues to increase, the landscapes start supporting dry and eventually wet forests, which produce increasingly dense plant and litter covers that decrease natural erosion. However, if the landscapes are denuded by disturbance (for example, fire, grazing, timber harvesting, and so forth), then the rate of erosion continues to increase with increasing precipitation (fig. 2.12). Surface conditions after fire are important for determining where water moves and how much erosion is produced (table 2.4).

Table 2.2—Sediment size classes and detachment/deposition velocities.

Sediment type	Size class		Detachment velocity		Deposition velocity	
	in	*cm*	*in/sec*	*cm/sec*	*in/sec*	*cm/sec*
	39.37	100.00				
Boulders						
	10.08	25.60	7.480	19.00	4.724	12.00
Cobble						
	2.520	6.40	5.906	15.00	3.150	8.00
Gravel						
	0.078	0.19	1.378	3.50	0.591	1.48
Sand						
	0.007	0.02	0.669	1.70	0.059	0.15
Silt						
	<0.001	<0.01	9.843	25.00	0.004	0.01
Clay						
	<0.001	<0.01				

Table 2.3—Natural sediment losses in the United States.

Location	Watershed condition	Sediment loss		Reference
		tons/ac	*Mg/ha*	
United States	Geologic erosion:			
	Natural, lower limit	0.26	0.58	Schumm and Harvey 1982
	Natural, upper limit	6.69	15.00	Schumm and Harvey 1982
Eastern U.S. Forests	Lower baseline	0.05	0.10	Patric 1976
	Upper baseline	0.11	0.22	
Western U.S.	Lower baseline	<0.01	<0.01	Biswell and Schultz 1965
	Upper baseline	2.47	5.53	DeByle and Packer 1972

Figure 2.12—Erosion from a clearcut and burned *Pinus rigida* stand planted on degraded farmland, Southern Appalachian Mountains, Georgia. (Photo by Daniel Neary).

Table 2.4—Soil surface conditions affect infiltration, runoff, and erosion.

Soil surface condition	Infiltration	Runoff	Erosion
Litter charred	High	Low	Low
Litter consumed	Medium	Medium	Medium
Bare soil	Low	High	High
Water repellent layers	Very low	Very high	Severe

Erosion is certainly the most visible and dramatic impact of fire apart from the consumption of vegetation. Fire management activities (wildfire suppression, prescribed fire, and postfire watershed rehabilitation) can affect erosion processes in wildland ecosystems. Wildfire, fireline construction, temporary roads, and permanent, unpaved roads receiving heavy vehicle traffic will increase erosion. Increased stormflows after wildfires will also increase erosion rates. Burned Area Emergency Rehabilitation (BAER) work on watersheds will decrease potential postfire erosion to varying degrees depending on the timing and intensity of rainfall (see chapter 10; Robichaud and others 2000).

Sheet, Rill, and Gully Erosion: Progressive Erosion—In sheet erosion, slope surfaces erode uniformly. This type proceeds to rill erosion in which small, linear, rectangular channels cut into the surface of a slope. Further redevelopment of rills leads to the formation of deep, large, rectangular to v-shaped channels (gullys) cut into a slope (fig. 2.13).

Some special erosion conditions can be encountered. For instance, in ecoregions with permafrost, the progression of erosion from sheet to rill to gully interacts with the depth of permafrost thaw. Until thaw occurs, erosion is essentially frozen. Fire and fire control activities such as fireline construction will affect thaw depth after wildfires, and subsequent erosion of firelines can be substantial (fig. 2.14).

Dry Ravel—*Dry ravel* is the gravity-induced downslope surface movement of soil grains, aggregates, and rock material, and is a ubiquitous process in semiarid steepland ecosystems (Anderson and others 1959). Triggered by animal activity, earthquakes, wind, freeze-thaw cycles, and thermal grain expansion during soil heating and cooling, dry ravel may best be

Figure 2.13—Incised gulley after postwildfire runoff on the White Springs Fire, 1996, White Mountain Apache Nation, Arizona. (Photo by Daniel Neary).

described as a type of dry grain flow (Wells 1981). Fires greatly alter the physical characteristics of hillside slopes, stripping them of their protective cover of vegetation and organic litter and removing barriers that were trapping sediment. Consequently, during and immediately following fires, large quantities of surface material are liberated and move downslope as dry ravel (Krammes 1960, Rice 1974). Dry ravel can equal or exceed rainfall-induced hillslope erosion after fire in chaparral ecosystems (Krammes 1960, Wohlgemuth and others 1998).

In the Oregon Coast Range, Bennett (1982) found that prescribed fires in heavy slash after clearcutting produced noncohesive soils that were less resistant to the force of gravity. Dry ravel on steep slopes (greater than 60 percent) that were prescribed burned produced 118 yard^3/acre (224 m^3/ha) of surface erosion compared to 15 yard^3/acre (29 m^3/ha) on moderate slopes with burning, and 9 yard^3/acre (17 m^3/ha) where burning was not done after clearcutting. Sixty-four percent of the erosion, as dry ravel, occurred within the first 24 hours after burning.

Mass Failures—This term includes slope creep, falls, topples, rotational and translational slides, lateral spreads, debris flows, and complex movements (Varnes 1978). *Slope creep* is a slow process that does not deliver large amounts of sediment to stream channels in the periods normally considered in natural resources management. The most important for forest management considerations are rotational and translational slides, flows, and complex movement (Ice 1985). *Slump-earthflows* and *debris avalanches* are more likely with increased water in the soil because of the decreased tension between soil particles, increased

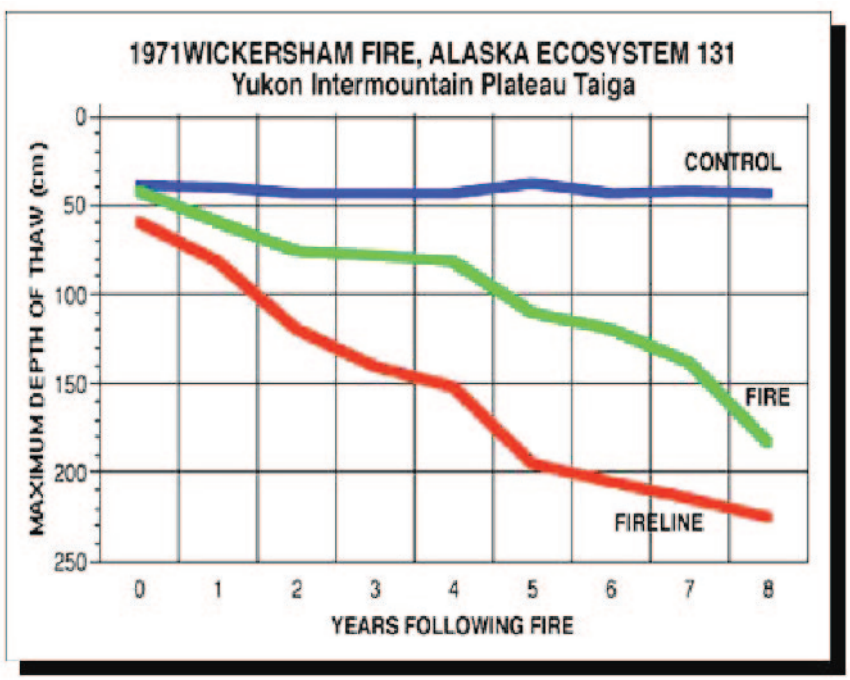


Figure 2.14—Postwildfire erosion following permafrost thaw in Alaska. (After Viereck 1982, Figure courtesy of the USDA Forest Service, National Advanced Fire and Resource Institute, Tucson, AZ).

loading on slopes produced by excess soil water, and a buoyant effect created by soil water along a failure plane. These types of failures are most often associated with clearcutting, tree death due to disease or insects, and road network construction. Slope failures associated with fires are the result of the loss of the forest floor, surface sealing, and the development of water repellency. These processes produce a diversion of rainfall from infiltration to surface runoff. The result is a dramatic increase in debris torrents in channels that greatly increase sediment delivery to channels and flooding (see chapter 5).

Debris avalanches are the largest, most dramatic, and main form of mass wasting that delivers sediment to streams (Benda and Cundy 1990). They can range from slow moving earth flows to rapid avalanches of soil, rock, and woody debris. Debris avalanches occur when the mass of soil material and soil water exceed the sheer strength needed to maintain the mass in place. The loss of root strength (for example, soil strength, anchoring, soil mass cohesion, and so forth) due to removal of trees or tree death caused by insects or disease aggravates the situation. Steep slopes, logging, road construction, and heavy rainfall aggravate debris avalanching potential (table 2.5).

Most fire-associated mass failures are debris flows associated with development of water repellency in soils (DeBano and others 1998). Chaparral occupying steep slopes in southern California has a high potential for mass failures, particularly when deep-rooted chaparral species are replaced with shallower-rooted grass species (Rice 1974). These mass failures are a large source of sediment delivered to stream channels (can be 50 percent of the total postfire sediment yield in some ecoregions; fig. 2.15). Wells (1981) reported that wildfire in chaparral vegetation in coastal southern California can increase average debris avalanche sediment delivery in large watersheds from 18 to 4,845 yard^3/mile^2/year (7 to 1,910 m^3/km^2/year). However, individual storm events in smaller basins can trigger much greater sediment yields (Gartner and others 2004; table 2.6). Rates as high as 221,026 yard^3/mile^2 (65,238 m^3/km^2) have been measured after single storms in California chaparral. Other ecoregions in the Western United States have postfire debris flows that have been larger (for example, 304,761 yard^3/mile^{2}or 89,953 m^3/km^2 in ecoregion M331, Colorado) but not with the same frequency (table 2.6). The situation could change with the increasing severity, frequency, and distribution of forest wildfires that have characterized the past decade.

Table 2.5—Annual sediment yields from debris avalanches in undisturbed forests and those affected by clearcutting, roads, and wildfire. (From Ice 1985, Neary and Hornbeck 1994).

Ecoregion-location	Treatment/condition	Sediment yield	
		yd^3/mi^2/yr	*m^3/km^2/yr*
M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE[1]			
Siuslaw National Forest,	Uncut	95	28
Oregon	Clearcut	376	111
	Roads	11,858	3,500
H.J. Andrews, Oregon	Uncut	122	36
	Clearcut	447	132
	Roads	3,964	1,170
Northwest Washington	Uncut	244	72
	Roads	39,978	11,800
British Columbia	Uncut	37	11
	Clearcut	81	24
	Roads	955	282
Entiat Experimental Forest,	Wildfire – Fox Basin	10,164	3,600
Washington	Wildfire – Burns	420	124
	BasinWildfire – McCree Basin	12,197	3,000
M262 CALIFORNIA COASTAL RANGE WOODLAND-SHRUB-CONIFER PROVINCE			
Southern California	Uncut	24	7
	Wildfire	6,461	1,907

[1]Bailey's (1995) descriptions of ecoregions of the United States.

Figure 2.15—Deeply incised gully in the Chiricahua Mountains after the Rattlesnake Fire, 1996, Coronado National Forest, Arizona. (Photo courtesy of the Tree Ring Laboratory, University of Arizona).

Cannon (2001) describes several types of debris flow initiation mechanisms after wildfires in the Southwestern United States. Of these, surface runoff, which increases sediment entrainment, was the dominant triggering mechanism. Campbell and others (1977) reported a 416-fold increase in sediment yield after wildfire in southwestern ponderosa pine.

Channel Stability—Fire-related sediment yields vary, depending on fire frequency, climate, vegetation, and geomorphic factors such as topography, geology, and soils (Swanson 1981). In some regions, more than 60 percent of the total landscape sediment production over the long term is fire-related. Much of that sediment loss can occur the first year after a wildfire (Rice 1974, Agee 1993, DeBano and others 1996, DeBano and others 1998, Wohlgemuth and others 1998, Robichaud and Brown 1999).

A stable stream channel reflects a dynamic equilibrium between incoming and outgoing sediment and streamflow (Rosgen 1996). Increased sideslope erosion after fires can alter this equilibrium by transporting additional sediment into channels (aggradation). Here it is stored until increased peakflows (see chapter 5) produced after fires erode the channel (degradation) and move the stored material downstream (Heede and others 1988). Sediment transported from burned areas as a result of increased peakflows can adversely affect aquatic habitat, recreation areas, roads, buildings, bridges, and culverts. Deposition of sediments alters habitat and can fill in lakes and reservoirs (Reid 1993, Rinne 1996).

Table 2.6—Event-based sediment yields from debris flows due to wildfires. (From Gartner and others 2004).

Ecoregion-location	Treatment/condition Burn area	Rainfall	Sediment yield per event	
	%	mm	yd^3/mi^2	m^3/km^2
M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE [1]				
Entiat Valley, WA, 1972	100	335	1,355	400
M262 CALIFORNIA COASTAL RANGE WOODLAND-SHRUB-CONIFER PROVINCE				
Los Angeles County, CA, 1914	80	Unknown	60,069	17,730
Los Angeles County, CA, 1928	100	36	45,680	13,483
Los Angeles County, CA, 1933	100	356	67,943	20,054
San Dimas, W. Fork, CA, 1961	100	40	54,906	16,206
Glendora, Glencoe, CA, 1969	80	1,143	203,280	60,000
Glendora, Rainbow, CA, 1969	80	1,143	221,026	65,238
Big Sur, Pfiefer, CA, 1972	100	21	22,588	6,667
Sierra Madre, CA, 1978	100	38	7,650	2,258
San Bernardino, CA, 1980	NA	Unknown	160,432	47,353
Laguna Canyon, CA, 1993	85	51	73,303	21,636
Hidden Springs, CA, 1978	100	250	84,700	25,000
Sierra Madre, CA, 1978	100	38	7,650	2,258
Topanga, CA, 1994	100	66	783	231
Ventura, Slide Creek, CA, 1986	100	122	871	257
M331 ROCKY MOUNTAIN STEPPE-OPEN WOODLAND-CONIFEROUS FOREST				
Glenwood Springs, CO, 1994	97	17	41,537	12,260
Glenwood Springs, CO, 1994	58	17	7,247	2,139
M341 NV-UT SEMI-DESERT-CONIFEROUS FOREST-ALPINE MEADOW				
Santaquin, UT, 2001	29	12	304,761	89,953
Santaquin, UT, 2001	28	12	31,657	9,344
M313 AZ-NM MOUNTAINS SEMIDESERT-WOODLAND-CONIFER PROVINCE				
Huachuca Mountains, AZ, 1988	80	8	56,468	16,667

[1]Bailey's (1995) descriptions of ecoregions of the United States.

Postfire Sediment Yields

Baseline Yields—Some reference sediment yield baselines are presented in tables 2.3, 2.5, and 2.6. Natural erosion rates for undisturbed forests in the Western United States of less than 0.01 to 2.47 tons/acre/year (less than 0.01 to 5.53 Mg/ha/year) are higher than Eastern United States yields of 0.05 to 0.10 tons/acre/year (0.1 to 0.2 Mg/ha/year) but don't approach the upper limit of geologic erosion (Maxwell and Neary 1991). These differences are due to natural site factors such as soil and geologic erosivity, rates of geologic uplift, tectonic activity, slope, rainfall amount and intensity, vegetation density and percent cover, and fire frequency. Landscape-disturbing activities such as mechanical site preparation (6.7 tons/acre; 15 Mg/ha/year; Neary and Hornbeck 1994), agriculture (249.8 tons/acre; 560 Mg/ha/year; Larson and others 1983), and road construction (62.4 tons/acre; 140 Mg/ha/year; Swift 1984) produce the most sediment loss and can match or exceed the upper limit of natural geologic erosion.

Yields from Fires—Fire-related sediment yields vary considerably, depending on fire frequency, climate, vegetation, and geomorphic factors such as topography, geology, and soils (Anderson and others 1976, Swanson 1981). In some regions, over 60 percent of the total landscape sediment production over the long term is fire-related. Much of that sediment loss can occur the first year after a wildfire (Rice 1974, Agee 1993, DeBano and others 1996, DeBano and others 1998, Wohlgemuth and others 1998, Robichaud and Brown 1999). An example is the large amount of sediment that filled in a 10 acre (4 ha) lake on the Coronado National Forest after the Rattlesnake Fire of 1996 (fig. 2.16). Tables 2.7, 2.8, 2.9, and 2.10 show the range of sediment yield increases from the first

Figure 2.16—Rucker Lake, Coronado National Forest, filled in by erosion off of the Rattlesnake Fire, Chiricahua Mountains, Arizona. (Photo by Daniel Neary).

Table 2.7—Sediment losses produced by different land management activities.

Location	Management activity	1st year sediment loss		Reference
		tons/ac	*Mg/ha*[1]	
CONTINENTAL UNITED STATES				
USA	Cropland			Larson and others 1983
	Maximum tolerance	5.00	11.20	
	Maximum loss	249.98	560.50	
Eastern U.S.A.	Forest roadbuilding	62.44	140.00	Swift 1984
M212 ADIRONDACK-NEW ENGLAND MIXED FOREST PROVINCE[2]				
New Hampshire	Clearcut	0.16	0.37	
231 SOUTHEASTERN MIXED FOREST PROVINCE				
North Carolina	Cut, shear, disk	4.18	9.37	Douglass and Godwin 1980
Virginia	Cut, shear, disk	7.14	15.00	Fox and others 1983
Mississippi	Cut, disk, bed	6.36	14.25	Beasley 1979
232 COASTAL PLAIN MIXED FOREST PROVINCE				
Florida	Clearcut, windrow	0.02	0.04	Riekerk 1983
M231 OUCACHITA MIXED FOREST-MEADOW PROVINCE				
Arkansas	Clearcut, shear	0.11	0.54	Beasley and others 1986
M313 AZ-NM MOUNTAINS SEMIDESERT-WOODLAND-CONIFER PROVINCE				
Arizona	Clearcut, roading	0.06	0.14	Neary and Hornbeck 1994
M332 MIDDLE ROCKY MOUNTAIN STEPPE-CONIFER-MEADOW PROVINCE				
Montana	Clearcut	0.08	0.18	DeByle and Packer 1972
M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE				
Oregon	Clearcut, roading	0.65	0 .46	Neary and Hornbeck 1994

[1]Mg/ha is metric tons per hectare.
[2]Bailey's (1995) descriptions of ecoregions of the United States.

Table 2.8—Sediment losses the first year after prescribed (Rx) fire and wildfires, part 1.

Location	Management activity	1st year sediment loss		Reference
		tons/ac	*Mg/ha*[1]	
231 SOUTHEASTERN MIXED FOREST PROVINCE[2]				
South Carolina	Loblolly pine			Van Lear and others 1985
	Control	0.012	0.027	
	Understory burn (R=2)	0.019	0.042	
	Burn, cut (R=2)	0.067	0.151	
North Carolina	Southern hardwoods			Copley and others 1944
	Control	0.002	0.004	
	Semi-annual Rx burn	3.077	6.899	
Mississippi	Scrub oak			Ursic 1970
	Control	0.210	0.470	
	Rx burn (R=3 yr)	0.509	1.142	
Mississippi	Scrub oak			Meginnis 1935
	Control	0.025	0.056	
	Rx burn	0.330	0.739	
Texas	Loblolly pine			Pope and others 1946
	Control	0.050	0.112	
	Annual Rx burning	0.359	0.806	
Texas	Loblolly pine			Ferguson 1957
	Control	0.100	0.224	
	Single Rx burn	0.210	0.470	
M231 OUACHITA MIXED FOREST-MEADOW PROVINCE				
Arkansas	Shortleaf pine			Miller and others 1988
	Control	0.016	0.036	
	Cut, slash Rx burn	0.106	0.237	
M222 OZARK BROADLEAF FOREST PROVINCE				
Oklahoma	Mixed hardwoods			Daniel and others 1943
	Control	0.010	0.022	
	Annual Rx burning	0.110	0.246	

[1]Mg/ha is metric tons per hectare.
[2]Bailey's (1995) descriptions of ecoregions of the United States.

year after prescribed burns and wildfires. Sediment yields 1 year after prescribed burns and wildfires range from very low, in flat terrain and in the absence of major rainfall events, to extreme, in steep terrain affected by high intensity thunderstorms. Erosion on burned areas typically declines in subsequent years as the site stabilizes, but the rate of recovery varies depending on burn or fire severity and vegetation recovery.

Soil erosion following fires can vary from under 0.1 tons/acre/year (0.1 Mg/ha/year) to 6.7 tons/acre/year (15 Mg/ha/year) in prescribed burns, and from less than 0.1 tons/acre (less than 0.1 Mg/ha/year) in low-severity wildfire, to more than 164.6 tons/acre/year (369 Mg/ha/year) in high-severity wildfires on steep slopes (Hendricks and Johnson 1944, Megahan and Molitor 1975, Neary and Hornbeck 1994, Robichaud and Brown 1999). For example, Radek (1996) observed erosion of 0.13 tons/acre/year (0.3 Mg/ha/year) to 0.76 tons/acre/year (1.7 Mg/ha/year) from several large wildfires that covered areas ranging from 494 to 4,370 acres (200 to 1,770 ha) in the northern Cascades Mountains. Three years after these fires, large erosional events occurred from spring rainstorms, not from snowmelt. Most of the sediment produced did not leave the burned area. Sartz (1953) reported an average soil loss of 1.5 inch (37 mm) after a wildfire on a north-facing slope in the Oregon Cascades. Raindrop splash and sheet erosion accounted for the measured soil loss. Annual precipitation was 42.1 inches (1,070 mm), with a maximum intensity of 3.54 inch/hour (90 mm/hour). Vegetation covered the site within 1 year after the burn. Robichaud and Brown (1999) reported first-year erosion rates after a wildfire from 0.5 to 1.1 tons/acre (1.1 to 2.5 Mg/ha/year), decreasing by an order of magnitude by the second year, and to no

Table 2.9—Sediment losses the first year after prescribed (Rx) fire and wildfires, part 2.

Location	Management activity	1st year sediment loss		Reference
		tons/ac	*Mg/ha*[1]	
M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE[2]				
Washington	Mixed conifer			Helvey 1980
	Control	0.012	0.028	
	Wildfire	1.049	2.353	
M261 SIERRAN STEPPE-MIXED FOREST-CONIFER FOREST PROVINCE				
California	Ponderosa pine			Biswell and Schultz 1965
	Control	<0.001	<0.001	
	Understory Rx burn	<0.001	<0.001	
261 CALIFORNIA COASTAL CHAPARRAL				
California	Chaparral			Wells 1981
	Control	0.019	0.043	
	Wildfire (R=3 yr)	12.758	28.605	
California	Chaparral			Krammes 1960
	Control	2.466	5.530	
	Wildfire	24.664	55.300	
California	Chaparral			Wohlgemuth 2001
	Control			
	Rx burn	0.708	1.587	
	Reburn	0.389	0.872	
	Wildfire	9.058	20.309	
313 COLORADO PLATEAU SEMI-DESERT PROVINCE				
Arizona	Chaparral			Pase and Lindenmuth 1971
	Control	<0.001	<0.001	
	Prescribed fire	1.685	3.778	
Arizona	Chaparral			Pase and Ingebo 1965
	Control	0.043	0.096	
	Wildfire	12.798	28.694	
Arizona	Chaparral			Glendening and others 1961
	Control	0.078	0.175	
	Wildfire	90.984	204.000	

[1]Mg/ha is metric tons per hectare.
[2]Bailey's (1995) descriptions of ecoregions of the United States.

sediment by the fourth, in an unmanaged forest stand in eastern Oregon. DeBano and others (1996) found that following a wildfire in ponderosa pine, sediment yields from a low severity fire recovered to normal levels after 3 years, but moderate and severely burned watersheds took 7 and 14 years, respectively. Nearly all fires increase sediment yield, but wildfires in steep terrain produce the greatest amounts, 12.5 to164.8 tons/acre/year (28 to more than 369 Mg/ha/year). Noble and Lundeen (1971) reported an average annual sediment production rate of 2.5 tons/acre (5.7 Mg/ha) from a 902 acre (365 ha) burn on steep river breaklands in the South Fork of the Salmon River, Idaho. This rate was approximately seven times greater than hillslope sediment yields from similar, unburned lands in the vicinity.

Sediment yields usually are the highest the first year after a fire and then decline in subsequent years. However, if precipitation is below normal, the peak sediment delivery year might be delayed until years 2 or 3. In semiarid areas like the Southwest, postfire sediment transport is episodic in nature, and the delay may be longer. All fires increase sediment yield, but it is wildfire that produces the largest amounts. Slope is a major factor in determining the amount of sediment yielded during periods of rainfall following fire (see table 2.10). There is growing evidence that short-duration, high-intensity rainfall (greater than 50 mm/hour in 10- to-15 minute bursts) over areas of about 1 km^2 (247 acres) often produce the flood flows that result in large amounts of sediment transport (Neary and

Table 2.10—Sediment losses the first year after prescribed (Rx) fire and wildfires, part 3

Location	Management activity	1st year sediment loss		Reference
		tons/ac	*Mg/ha*[1]	
M313 AZ-NM MOUNTAINS SEMIDESERT-WOODLAND-CONIFER PROVINCE[2]				
Arizona	Ponderosa Pine			Campbell and others 1977
	Control	0.001	0.003	DeBano and others 1996
	Wildfire low severity	0.036	0.080	
	Wildfire moderate severity	0.134	0.300	
	Wildfire high severity	0.559	1.254	
Arizona	Mixed conifer			Hendricks and Johnson 1944
	Control	<0.001	<0.001	
	Wildfire, 43% slope	31.969	71.680	
	Wildfire, 66% slope	89.914	201.600	
	Wildfire, 78% slope	164.842	369.600	
315 SOUTHWEST PLATEAU AND PLAINS STEPPE AND SHRUB PROVINCE				
Texas	Juniper and grass			Wright and others 1982
	Control	0.027	0.060	
	Rx burn (3 yr)	6.690	15.000	
	Rx burn, seed (1 yr)	1.338	3.000	
Texas	Juniper and grass			Wright and others 1976
	Control: level	0.011	0.025	
	Rx burn: level	0.013	0.029	
	Control: 15-20%	0.034	0.076	
	Rx: 15-20% slope	0.836	1.874	
	Control: 43-54%	0.006	0.013	
	Rx: 43-54% slope	3.766	8.443	
M332 MIDDLE ROCKY MTN STEPPE-CONIFER FOREST-MEADOW PROVINCE				
Montana	Larch, Douglas-fir			DeByle 1981
	Control	<0.001	<0.001	
	Slash Rx burn	0.067	0.150	
342 IINTERMOUNTAIN SEMIDESERT				
Idaho	Sagebrush, grass, forb			Pierson and others 2001b
	Control, interspace	<0.013	<0.030	
	Moderate severity fire	0.056	0.125	
	High severity fire	0.686	1.538	

[1]Mg/ha is metric tons per hectare.
[2]Bailey's (1995) descriptions of ecoregions of the United States.

Gottfried 2002, Gottfried and others 2003, Gartner and others 2004; see also chapter 5).

Best Management Practices certainly have value in reducing sediment losses from prescribed fires. O'Loughlin and others (1980) reported that a 66 foot (20 m) buffer strip in a steep watershed reduced sediment loss after prescribed fire from 800 percent of the control watershed to 142 percent. Mitigative techniques for reducing sediment losses after wildfires often are used as part of burned area emergency watershed rehabilitation, but they have their limitations (see chapter 10).

After fires, turbidity can increase due to the suspension of ash and silt-to-clay-sized soil particles in streamflow. Turbidity is an important water quality parameter because high turbidity reduces municipal water quality and can adversely affect fish and other aquatic organisms (see chapter 7). It is often the most easily visible water quality effect of fires (DeBano and others 1998). Less is known about turbidity than sedimentation in general because it is difficult to measure, highly transient, and extremely variable.

Extra coarse sediments (sand, gravel, boulders) transported off of burned areas or as a result of increased storm peakflows can adversely affect aquatic habitat, recreation areas, and reservoirs. Deposition of coarse sediments destroys aquatic and riparian habitat and fills in lakes or reservoirs (Reid 1993, Rinne 1996).

Management Implications

Resource managers need to be aware of the changes in the physical properties that occur in the soil during a fire. The most important physical process functioning during a fire is the transfer of heat into the soil. Soil heating not only affects soil physical properties but also changes many of the chemical and biological properties in soils described in chapters 3 and 4, respectively.

Wildfires present their own unique concerns. Although little can be done to modify soil heating during a wildfire, managers need to be aware of the susceptibility of severely burned areas to postfire runoff and erosion. Excessive heating of the underlying soils during these uncontrollable fires can change soil structure to such an extent that water infiltration is impeded, creating excessive runoff and erosion following fire. Formation of water repellent soils during these wildfires may present special concerns with erosion following wildfires and needs to be addressed when initiating postfire treatments. Postfire rehabilitation is an important activity on areas burned by wildfire where it is necessary to reestablish plant cover as soon as possible to protect the bare soil surface from raindrop impact. However, discretion is needed in order to apply the most effective and practical postfire rehabilitation treatments. Numerous revegetation techniques that are available for use on burned watersheds are discussed in chapter 10 of this publication and elsewhere (Robichaud and others 2000).

The use of prescribed fire, however, presents the manager with alternatives for minimizing the damage done to the soil. The least amount of damage occurs during cool-burning, low-severity fires. These fires do not heat the soil substantially, and the changes in most soil properties are only minor and are of short duration. However, the burning of concentrated fuels (for example, slash, large woody debris) can cause substantial damage to the soil resource, although these long-term effects are limited to only a small proportion of the landscape where the fuels are piled. These types of fire use should be avoided whenever possible. The burning of organic soils is also a special case where extensive damage can occur unless burning prescriptions are carefully planned.

Summary

The physical processes occurring during fires are complex and include both heat transfer and the associated change in soil physical characteristics. The most important soil physical characteristic affected by fire is soil structure because the organic matter component can be lost at relatively low temperatures. The loss of soil structure increases the bulk density of the soil and reduces its porosity, thereby reducing soil productivity and making the soil more vulnerable to postfire runoff and erosion. Although heat is transferred in the soil by several mechanisms, its movement by vaporization and condensation is the most important. The result of heat transfer in the soil is an increase in soil temperature that affects the physical, chemical, and biological properties of the soil. When organic substances are moved downward in the soil by vaporization and condensation they can cause a water-repellent soil condition that further accentuates postfire runoff and erosion. Water repellency accelerates postfire runoff, which in turn creates extensive networks of surface rill erosion. Water repellency also increases erosion by raindrop splash. The magnitude of change in soil physical properties depends on the temperature threshold of the soil properties and the severity of the fire. The greatest change in soil physical properties occurs when smoldering fires burn for long periods.

Notes

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Chapter 3: Soil Chemistry

Introduction

The chemical properties of the soil that are affected by fire include individual chemical characteristics, chemical reactions, and chemical processes (DeBano and others 1998). The soil chemical characteristics most commonly affected by fire are organic matter, carbon (C), nitrogen (N), phosphorus (P), sulfur (S), cations, cation exchange capacity, pH, and buffer power. Some purely chemical reactions occur in soils. These include the exchange of cations adsorbed on the surface of mineral soil particles and humus with their surrounding solutions. Another predominately chemical reaction is the chemical weathering of rocks and their eventual transformation into secondary clay minerals during soil formation. During the chemical decomposition of rock material, the soil and its surrounding solution become enriched with several cations. Associated with the chemical interactions during weathering and soil formation are physical forces (freezing and thawing, wetting and drying) and biological activities (production of organic acids during the decomposition of humus) that also accelerate soil development. The most common chemical processes occurring in soils that are affected by fire, however, are those mechanisms that are involved in nutrient availability and the losses and additions of nutrients to the soil.

Soil Chemical Characteristics

The chemical characteristics of soils range from the inorganic *cations*—for example, calcium (Ca), sodium (Na), magnesium (Mg), potassium (K), and so forth—that are adsorbed on the surface of clay materials to those contained mainly within the organic matrix of the soil—for example, organic matter, C, N, P, S. All chemical characteristics are affected by fire, although the temperatures at which changes occur can vary widely. The best estimates available in the literature for the threshold temperatures of individual soil chemical characteristics are given in table 3.1.

The published information describing the effects of fire on changes in individual chemical constituents of soils and organic matter are contradictory and have often led to differing conclusions about the magnitude and importance of the chemical changes that actually occur during a fire. Different studies have concluded that soil chemical constituents increase, decrease, or remain the same (DeBano and others 1998). This has been particularly true for studies reporting changes in N and other nutrients that can volatilize readily during a fire (for example, organic matter, sulfur, and phosphorus). Differing conclusions arise primarily because of the method used for

Table 3.1—Temperature thresholds for several soil chemical characteristics of soil.

Soil characteristic	Threshold temperature		Source
	°F	*°C*	
Organic matter	212	100	Hosking 1938
Nitrogen	414	200	White and others 1973
Sulfur	707	375	Tiedemann 1987
Phosphorus and potassium	1,425	774	Raison and others 1985a,b
Magnesium	2,025	1,107	DeBano 1991
Calcium	2,703	1,484	Raison and others 1985a,b
Manganese	3,564	1,962	Raison and others 1985a,b

calculating the chemical constituents. Chemical changes can be expressed in either percentages (or some other expression of concentrations, for example, ppm or mg/kg) or be based on the actual changes in total amounts of the constituent (for example, pounds/acre or kg/ha). Before fire, the percent of a given chemical constituent is usually based on the amount contained in a prefire sample that can contain variable amounts of organic matter. In contrast, following the fire the percent of the same chemical constituent is based on the weight of a burned sample that contains varying amounts of ash along with charred and unburned organic matter. Thus, the confusion in nutrient changes arises because different bases are used for calculating the change in a particular chemical constituent (that is, based on mainly organic matter before combustion as compared to ashy and unburned materials following a fire). This confusion between percentages and total amounts was first reported in a study on the effect of fire on N loss during heating (Knight 1966). This study indicated that the differences between percent N and total amount of N started at 212 °F (100 °C) and became greater until about 932 °F (500 °C). Because of these difficulties in interpreting concentration and percentage data, the following discussion on the fire-related changes in chemical constituents will first focus on the more fundamental changes in chemical constituents in wildland ecosystems.

As a general rule, the total amounts of chemical elements are never increased by fire. The total amounts of different chemical elements on a particular burned site most likely decrease, although in some cases may remain the same (for example, elements with high temperature thresholds such as Mg, Ca, and others listed in table 3.1). The fire, however, does change the form of different elements and in many cases makes them more available for plants and other biological organisms. A classic example of this is total N contained in the ecosystem organic matter (table 3.2). When organic matter is combusted, total N on the site is always decreased, although increases in the available forms of N are likely to occur as is discussed in a later section, "Nitrogen." Therefore, managers must be alert when interpreting the significance of the sometimes contradictory changes in different nutrients during a fire that are reported in the literature. The following sections focus on describing these changes in terms of the underlying chemical processes and to indicate the management implications of these changes in terms of soil and ecosystem productivity and postfire management.

Table 3.2—Effect of burning at low, medium, and high severities on organic matter and mineralizable N in forest soils in northern Idaho. (Adapted from Niehoff 1985, and Harvey and others 1981).

Treatment depths =	Organic matter Mineral soil 2.5-7.5 cm		Mineralizable N Mineral 2.5-7.5 cm	Mineralizable N Organic 0-2.5 cm	Total N
	Percent	*Change (%)*	*ppm (mg/kg)*	*ppm (mg/kg)*	*Change (%)*
Undisturbed	3.6	0	9.4	68	0
Clearcut					
No burn	3.9	+8	9.7	97	+22
Low	4.1	+12	9.5	75	+8
Medium	2.8	–22	9.3	5	–82
High	0.6	–83	0.7	0	–99

Organic Matter and Carbon

Many chemical properties and processes occurring in soils depend upon the presence of organic matter. Not only does it play a key role in the chemistry of the soil, but it also affects the physical properties (see chapter 2) and the biological properties (see chapter 4) of soils as well. Soil organic matter is particularly important for nutrient supply, cation exchange capacity, and water retention. However, burning consumes aboveground organic material (future soil organic matter, including large logs), and soil heating can consume soil organic matter (fig. 3.1). The purpose of the following discussion is to focus as much as possible on the purely chemical properties of organic matter and on changes that occur as the result of soil heating. Because organic C is one of the major constituents of organic matter, the changes in organic matter and organic C during soil heating are considered to be similar for all practical purposes.

Location of Organic Matter in Different Ecosystems—Organic compounds are found in both aboveground and belowground biomass where they make up the standing dead and live plants and dead organic debris (that is, leaves, stems, twigs, and logs) that accumulate on the soil surface and throughout the soil profile (DeBano and others 1998). Organic matter found in the soil consists of at least seven components, namely:

- The L-layer, (oi) which is made up of readily identifiable plant materials.
- The F-layer, (oe) which contains partially decomposed organic matter but can still be identified as different plant parts (needles, leaves, stems, twigs, bark, and so forth).
- The H-layer, (oa) which is made up of completely decayed and disintegrated organic materials, some of which is usually mixed with the upper mineral soil layers.
- Coarse woody debris that is eventually decayed but can remain on the soil surface or buried in the mineral soil for long periods.
- Charcoal or other charred materials that become mixed with the forest floor and uppermost layers of the mineral soil.
- The uppermost part of the A-horizon, which is composed mainly of a mixture of humus and mineral soil particles.
- A mixture of mineral soil, plant roots, and biomass (live and dead) that is concentrated primarily in the A-horizon but may extend downward into the B-horizon or deeper depending upon the type of vegetation growing on the site.

The amount of aboveground and belowground organic matter varies widely between different vegetation types depending upon on the temperature and moisture conditions prevailing in a particular area (DeBano and others 1998). In almost all ecosystems throughout the world, greater quantities of C (a measure of organic matter) are found belowground than aboveground (fig. 3.2). In grasslands, savannas, and tundra-covered areas, much greater quantities of organic C are found in the underground plant parts than aboveground (less than 10 percent of the total C in these herbaceous vegetation ecosystems is found

Figure 3.1—Large logs combusting in a prescribed fire in a ponderosa pine stand. (Photo by Daniel Neary).

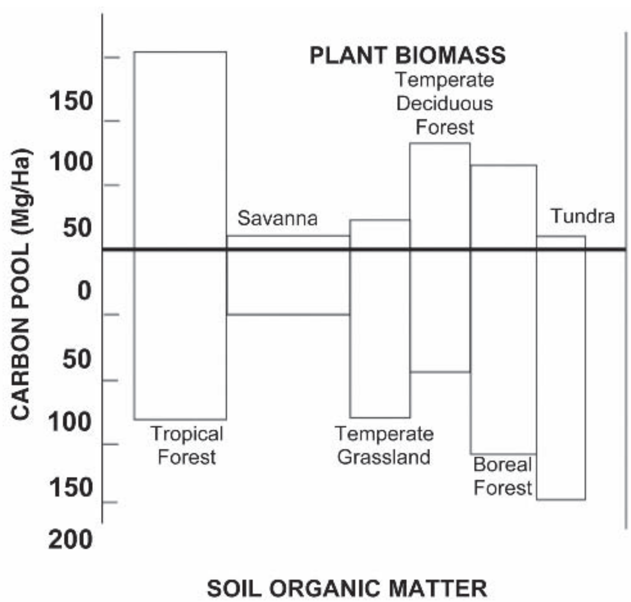

Figure 3.2—Distribution of C and soil organic matter (including litter) in major ecosystem types of the world. (Adapted from J. M. Anderson, 1991. The effects of climate change on decomposition processes in grassland and coniferous forest. Ecological Applications. 1: 326-347. Copyright © 1991 Ecological Society of America.)

aboveground). Tundra ecosystems are unique in that large amounts of organic matter accumulate on the soil surface because the low year-long temperatures severely limit decomposition. In forest ecosystems, C is more evenly distributed aboveground and belowground (for example, temperate deciduous and boreal forests). In general, soils with larger proportions of organic matter in the aboveground biomass and on their forest floors are more prone to disturbances (including fire) in their nutrient and C regimes than those in which most of the C in the ecosystem is located belowground.

Dynamics of Organic Matter Accumulation— In forests, the dynamics of the forest floor are responsible for the accumulation of organic matter, and the forest floor provides a major storage reservoir for nutrients that are cycled within natural ecosystems (fig. 3.3). An aggrading forest ecosystem sequesters nutrients and C aboveground in both the biomass and the forest floor (Knoepp and Swank 1994). Over many years this material forms the forest floor or the organic soil horizons (designated as Oi, Oe, and Oa horizons in part A, fig A.1). Depending on the soil type, organic matter may be concentrated in the forest floor or spread in decreasing amounts downward through the soil profile (fig. 3.4 and A.2).

The forest floor increases during forest development and aggradation when the rate of addition is greater than the rate of decomposition. For example, Knoepp and Swank (1994) found that forest floor mass increased by 28 and 45 percent over a 10-year period in an aggrading mixed oak (*Quercus* spp.) forest and white pine (*Pinus strobus*) plantation, respectively. In Arizona, Covington and Sackett (1992) examined forest floor accumulation in a ponderosa pine (*P. ponderosa*) stand having several different age "substands" within it and found several significant differences in forest floor mass among the substands. In general, the younger sapling areas had the smallest accumulations of forest floor materials compared to old growth pine substands that had the greatest. Over long periods the inputs and outputs of forest floor materials, coarse woody debris, fine woody debris, and leaf litter eventually reach a dynamic equilibrium depending upon the stand type (for example, coniferous versus hardwood) and forest management practices used (for example, uncut versus cut, log only versus whole-tree harvest).

Figure 3.4—Burn out of high organic matter soil in the Barca Slough, Vandenburg Air Force Base, California. (Photo by U.S. Air Force).

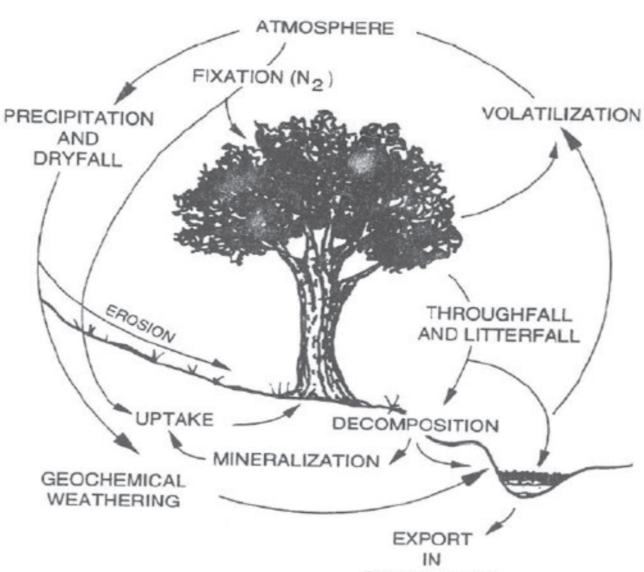

Figure 3.3—Nutrient cycling in natural environments. (Adapted from Brown 1980.)

Coarse Woody Debris—*Coarse woody debris* (including slash piles) is an important component of the organic matter pools found in forested ecosystems (fig. 3.5). In many cases it is partially or totally covered by soil and humus layers, and it has been found to comprise more than 50 percent of the total surface organic matter (this can amount to 16.5 to 22.3 tons/acre or 37 to 50 Mg/ha) in old growth forests in the Inland Northwest (Page-Dumroese and others 1991, Jurgensen and others 1997). Coarse woody debris, along with smaller organic matter, enhances the physical, chemical, and biological properties of the soil and thereby contributes directly to site productivity (Brooks and others 2003). It also provides a favorable microenvironment for the establishment

Figure 3.5—Coarse woody debris: (A) slash pile from pinyon-juniper thinning operation, Apache-Sitgreaves National Forest, Arizona; (B) logging slash in a Douglas-fir and mixed conifer, Mt. Baker-Snoqualmie National Forest, Cascade Mountains, Washington. (Photos by Malchus Baker and Kevin Ryan, respectively).

of seedlings and plant growth. The microbiological role of coarse woody debris is discussed further in chapter 4.

Fire Effects—Fire not only affects the organic matter by directly affecting its chemical composition but it also indirectly affects the subsequent decomposition rates. The magnitude of changes is related to the severity of the burn (for example, low, moderate, or high).

Chemical Changes: The low temperature threshold of organic matter makes it especially sensitive to soil heating during fire (table 3.1). Also, a major portion of the organic matter is located in the uppermost part of the soil profile (surface litter and humus layers) where it is exposed directly to the heat radiated downward during a fire (see part A). The changes in organic matter during the course of heating have been of interest to scientists for more than 60 years (Hosking 1938), and the changes reported by these earlier studies showed that:

- Losses of organic matter can occur at temperatures below 212 °F (100 °C).
- Volatile constituents in organic matter are lost at temperatures up to 392 °F (200 °C).
- Destructive distillation destroys about 85 percent of the soil organic matter at temperatures between 392 and 572 °F (200 and 300 °C).
- Above 572 °F (300 °C), the greater part of the residual organic matter consists of carbonaceous material, which is finally lost upon ignition.
- Heating the soil to 842 °F (450 °C) for 2 hours, or to 932 °F (500 °C) for 1/2 hour, destroys about 99 percent of the organic matter.

Recent studies on the detailed heat-induced changes in organic matter have improved our understanding of the specific chemical changes that occur in organic matter during the course of heating (DeBano and others 1998). At low temperatures, the changes in organic matter affect the more sensitive functional groups; at higher temperatures the thermal decomposition of nuclei occurs (Schnitzer and Hoffman 1964). At lower temperatures between 482 and 752 °F (250 and 400 °C) both phenolic OH and carboxyl (COOH) groups are lost, although phenolic groups are the more stable of the two. Another study of the thermal changes occurring in the H-layer under an evergreen oak forest in Spain showed that oxygen-containing functional groups found in humic and fulvic acids were altered (Almendros and others 1990). Humic acids were converted into alkali-insoluble substances that contributed to soil humus, while fulvic acids were transformed into acid-insoluble polymers. Biomass that was not completely burned contained both alkali-soluble lignin materials and brown products formed by dehydration of carbohydrates. The lignin compounds formed were more resistant to further chemical and biological change.

In a separate study, Ross and others (1997) found a decrease in total soil C and potassium sulfate extractable C at 1.5 and 2.5 years after burning. Sands (1983) found that 24 years after an intense site preparation burn on sandy soils, soil C was still lower than on adjacent unburned sites. This was generally the case for all soil C components he examined, including total organic C, extractable C, water-soluble C, humic acids, and carbohydrates.

Decomposition rates: In unburned ecosystems, natural decomposition processes (most biologically mediated) slowly release nutrients to tree and plant roots growing within the forest floor and to the mineral soil below. These biological processes add organic matter and nutrients to soils in forest environments under moderate temperatures through the activity of insects and microbes (DeBano 1991). Burning, however, acts as an instantaneous physical decomposition process that not only volatilizes nutrients, such as N, from the site but also alters the remaining organic materials (St. John and Rundel 1976).

Burning not only rapidly accelerates the rates of organic matter decomposition during the fire itself but can also indirectly affect postfire decomposition rates. For example, Schoch and Binkley (1986) and Raison and others (1990) studied loblolly pine (*P. taeda*) and radiata pine (*P. radiata*) plantations following fire. They found that decomposition rates of the remaining forest floor increased after burning, releasing ammonium (NH_4) and other nutrients. Observed changes in nutrient release and availability may be due to the alteration of organic matter solubility as a result of soil heating during a fire. The change in nutrient release and availability along with increased soil temperature and moisture content may also increase biological activity. This response may be short lived, however, because this readily available organic matter often diminishes rapidly and decomposition rates decrease (Raison and others 1990). Conversely, some studies have noted decreases in the rates of organic matter decomposition following burning. Monleon and Cromack (1996) measured decreased rates of decomposition in ponderosa pine forests immediately following burning and for up to 12 years. They concluded that the lower decomposition rates may be due to the combination of increased temperature and decreased moisture in the postfire forest floor. Springett (1976) also measured slower decomposition in Australian plantations and native forests caused by the changes in soil temperature and moisture as well as a decrease in the diversity and density of soil fauna following burning. This suggested that burning on a frequent rotation could simplify the litter fauna and flora, but these changes may permanently alter patterns of organic matter decomposition and nutrient release.

Both fire severity and frequency of burning affect the amount of organic matter that is lost as a result of burning. These are two characteristics of fire effects that fire management specialists can alter within the context of a prescribed burning program.

Fire severity: The effect of severity of burning on the amount of organic matter burned was reported for a 350 acre (142 ha) wildfire that burned a table mountain pine stand in the Shenandoah Valley (Groeschl and others 1990, 1992, 1993). This wildfire left a mosaic pattern of areas that were burned at different severities. They reported that on areas burned by a low-severity fire, the forest floor Oi and Oe layers were completely combusted, but the Oa layer remained. High-severity burning also consumed the Oa layer. Of the 10.1 tons/acre (22.6 Mg/ha) of C present in the forest floor in the unburned areas, no C remained in the high-severity burned areas compared to 9.3 tons/acre (20.8 Mg/ha) C that was left on the burned areas at low severities.

The effect of prescribed burning on the C and N content of the forest floor on an area supporting a mixed pine-oak overstory with a ericaceous shrub layer was studied in the Southern Appalachian Mountains (Vose and Swank 1993, Clinton and others 1996). The study areas were treated with felling-and-burning of existing trees to stimulate pine regeneration. Carbon and N content of the forest floor in these systems was examined after felling, prior to burning, and it was found that the forest floor contained between 15 and 22 percent of the total aboveground C on the site and 44 to 55 percent of the total aboveground N. Prescribed burning of these sites consumed the entire Oi layer, but 75 to 116 percent of the total forest floor C and 65 to 97 percent of the N remained in the combined Oe and Oa layers. On these sites, about half of the total aboveground N pool was contained in the forest floor. Total aboveground N losses ranged from 0.086 tons/acre (0.193 Mg/ha) for the lowest severity fire to 0.214 tons/acre (0.480 Mg/ha) on the most severe burn. Under prescribed burning conditions it is frequently possible to select appropriate weather conditions prior to burning (for example, time since last rain, humidity, temperature) to minimize the effects of fire on the consumption of organic matter. Therefore, it has been recommended that weather conditions prior to burning (for example, time since rainfall) could be used as a predictor of forest floor consumption (Fyles and others 1991).

Responses of total C and N are variable and depend on site conditions and fire characteristics. For example, Grove and others (1986) found no change in organic C in the surface 0 to 1.2 inch (0-3 cm) of soil immediately following burning; percent total N, however, increased. Knoepp and Swank (1993a,b) found no consistent response in total N in the upper soil layer, but did find increases in NH_4 concentrations and N mineralization on areas where a burning treatment followed felling.

Fire frequency: As would be expected, frequency of burning can affect C accumulations. A study was carried out on tropical savanna sites in Africa having both clay and sandy soils that were burned repeatedly every 1, 3, or 5 years (Bird and others 2000). While sites with clay soils had greater total C than did the sandy soils, they responded similarly to burning. All unburned sites had 40 to 50 percent greater C than burned sites. Low frequency burning (every 5 years) resulted in an increase in soil C of about 10 percent compared to the mean of all burned areas. High frequency burning (every year) decreased C about 10 percent. In another study, Wells and others (1979) reported the results of a 20-year burning study in a pine plantation in South Carolina. They found that periodic burning over 20-year period removed 27 percent of the forest floor. Annual burning conducted in the summer removed 29 percent of the forest floor as compared to a 54 percent loss resulting from winter burning. The total organic matter content of the surface soil (0 to 2 inches or 0 to 5 cm) increased in all cases, but there was no effect on the 2 to 3.9 inches (5-10 cm) soil layer. Interestingly, when they summed the organic matter in the forest floor and in the surface 0 to 3.9 inches (0-10 cm) of soil they found that these low-severity periodic burns sites had not reduced but only redistributed the organic matter.

The incidence of fire has been found to also affect the organic matter composition of savannahlike vegetation (referred to as "cerrado") in central Brazil (Roscoe and others 2000). On plots exposed to more frequent fires (burned 10 times in 21 years) C and N were decreased in the litter by 1.652 and 0.046 tons/acre (3.703 and 0.104 Mg/ha), respectively, although no significant differences were noted in the upper 3 feet (1 meter) of the underlying soil. Interestingly, the increase in fire incidence replaced the C_3-C with C_4-C by about 35 percent throughout the soil profile. This suggested that a more rapid rate of soil organic matter turnover occurred in areas burned by frequent fires, and as a result the soil would not be able to replace sufficient C to maintain long-term productivity of the site.

Prescribed fire was returned into overstocked ponderosa pine stands on the Mogollon Rim of Arizona for the purpose of restoring fire into the ecosystem and removing fuel buildups (Neary and others 2003). Prescribed fires were ignited at intervals of 1, 2, 4, 6, 8, and 10 years to determine the best fire return interval for Southwest ponderosa pine ecosystems (Sackett 1980, Sackett and others 1996). Two sites were treated—one on volcanic-derived soils and the other on sedimentary-derived soils near Flagstaff, AZ. Soil total C and total N levels were highly variable and exhibited an increasing, but inconsistent, concentration trend related to burn interval. They ranged from 2.9 to more than 6.0 percent total C and 0.19 to 0.40 percent total N (fig. 3.6). High spatial variability was measured within treatments, probably due to microsite differences (location of samples in the open, under large old-growth trees, in small-diameter thickets, in pole-sized stands, next to downed logs, and so forth). Stratification of samples by microsite differences could possibly reduce the within-plot variability but add complexity to any sampling design. Although there were statistically significant differences between the total C levels in soils of the unburned plots and the 8-year burning interval, there were no differences between burning intervals. There also was a statistically significant difference between unburned and 2-year burning interval and the 8-year burning interval in total soil N. This study determined that burning increased mineral soil C and N, which conflicted with Wright and Hart's (1997) contention that the 2-year burning interval could deplete soil N and C pools. This study did not examine the mineral fractions of the soil N pool, NH_4-N, and NO_3-N. Although the mineral forms of N are small (less than 2 percent of the total soil N pool), they are important for plant nutrition and microorganism population functions

Soil organic matter: Summary reports have described the effect of different management activities (including the effect of fire) on the organic matter and N found in the mineral portion of forest soils (D.W. Johnson 1992, Johnson and Curtis 2001). A metaanalysis on the results of 13 studies completed between 1975 and 1997 (table 3.3) show that the C and N contents of both the A-horizon and the underlying mineral soil layers change only a small amount (less than 10 percent) in the long term (fig. 3.7 and 3.8). These results agreed with the conclusions of another review (E. A. Johnson 1992) that indicated the overall effect of fire was not significant, although there was a significant effect of time since fire. It must be remembered, however, that although small changes in soil organic matter and C occurred in the soils during these studies, that substantial amounts of both organic litter and duff were most likely consumed during these fires. Organic matter and N losses from the forest floor could have a lasting effect on the long-term productivity and sustainability of forest sites, particularly when they occur on nutrient-deficient sites (see the later discussions in this chapter on N loss and ecosystem productivity).

Cation Exchange Capacity

Cation exchange is the interchange between cations in solution and different cations adsorbed on the surface of any negatively charged materials such as a clay or organic colloids (humus). *Cation exchange capacity* is the sum of the exchangeable cations found on organic and inorganic soil colloids (fig. 3.9). It arises from the negatively charged particles found on clay particles and colloidal organic matter in the soil. Cation exchange capacity sites are important storage places for soluble cations found in the soil. The adsorption of cations prevents the loss of these cations from the soils by leaching following fire. Although most of the exchange sites in soils are negative and attract cations, there are some positively charged sites that can attract anions (anion exchange has been reported to occur on clay particles).

The relative contribution of clay particles and organic matter to the cation exchange capacity of the soil depends largely upon the proportion of the two components and the total quantities of each present (Tate 1987). Cation exchange capacity also depends upon the type of clay and organic matter present. Clay materials such as montmorillonites have large exchange capacities, and other clays such as kaolinite are much lower. Other mineral particles such as silt and sand contain few adsorption sites for cations. In organic matter, the degree of humification affects the cation exchange capacity, and the more extensive the decomposition of organic material, the greater the exchange capacity.

Soil heating during a fire can affect cation exchange capacity in at least two ways. The most common change is the destruction of humus compounds. The location of the humus layer at, or near, the soil surface

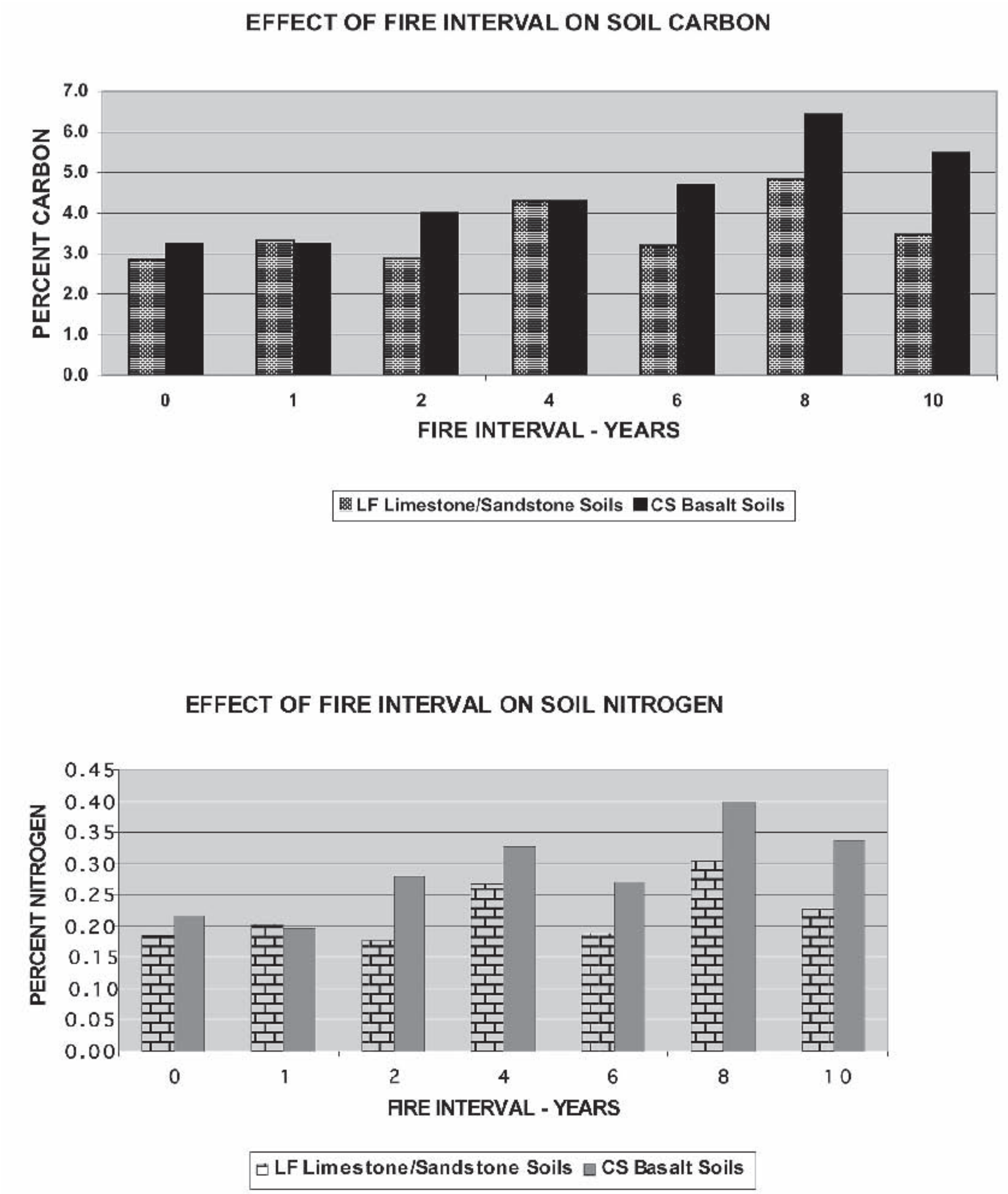

Figure 3.6—Effect of fire interval on (A) 0 to 2 inch (0-5 cm) soil total carbon, and (B) 0 to 2 inch (0-5 cm) soil nitrogen, Limestone Flats and Chimney Springs burning interval study, Arizona. (From Neary and others 2003).

makes it especially vulnerable to partial or total destruction during a fire because organic and humic materials start decomposing at about 212 °F (100 °C) and are almost completely destroyed at 932 °F (500 °C). These temperatures are easily reached during brushland and forest fires (see chapter 2 discussion on soil temperatures). In contrast, the cation exchange capacity of the clay materials is more resistant to change because heating and temperatures of 752 °F (400 °C) must be reached before dehydration occurs. The complete destruction of clay materials does not occur until temperatures of 1,292 to 1,472 °F (700 to 800 °C) are reached. In addition, clay material is seldom located on the soil surface but instead is located at least several centimeters below the soil surface in the B-horizon where it is well protected from

Table 3.3—References for the Johnson and Curtis (2001) meta-analysis of the effects of forest fires on soil C and N contents.

Location	Species	Fire type	Reference
Southern U.S.A.			
SC	Longleaf pine	PF[1]	Binkley and others 1992
SC, AL, FL, LA	Loblolly and other pines	PF	McKee 1982
Southwest U.S.A.			
AZ	Ponderosa pine	PF	Covington and Sackett 1986
AZ	Pinyon-juniper	PF	Klopatek and others 1991
Northwest U.S.A.			
WA	Mixed conifer	WF	Grier 1975
WA, OR	Douglas-fir, conifer mix	BB	Kraemer and Hermann 1979
OR	Ponderosa pine	PF	Monleon and others 1997
MT	Mixed conifer	PF	Jurgenson and others 1981
Alaska and Canada			
AL	Mixed spruce and birch	WF	Dyrness and others 1989
BC	Sub-boreal spruce	BB	Macadam 1987
World			
Australia	Eucalyptus	BB	Rab 1996
Algeria	Oak	WF	Rashid 1987
Sardinia	Chaparral	PF	Giovannini and others 1987

[1]PF: prescribed fire; WF: wildfire; BB: cut and broadcast burn.

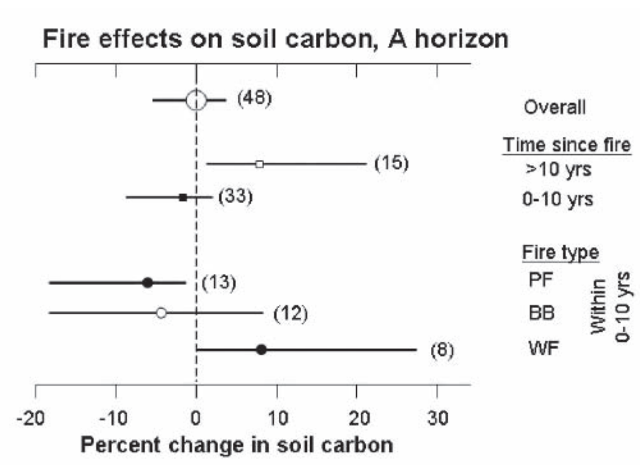

Figure 3.7—Fire effects on soil C, A-horizon nonparametric metaanalysis results; 99 percent confidence intervals (bars) and number of studies (in parentheses); PF = prescribed fire, WF = wildfire, and BB = broadcast burning of slash after harvest) (After Johnson and Curtis 2001, Forest Ecology and Management, Copyright © 2001, Elsevier B.V. All rights reserved).

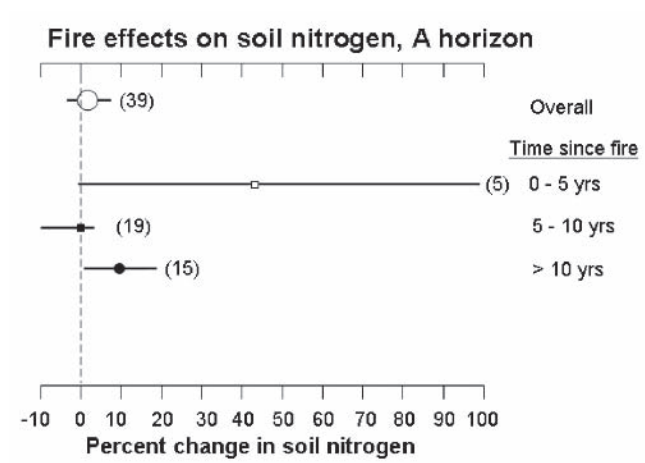

Figure 3.8—Fire effects on soil N, A-horizon nonparametric metaanalysis results; 99 percent confidence intervals (bars) and number of studies (in parentheses). (After Johnson and Curtis 2001).

Figure 3.9—Cation exchange capacity in soil is provided by both organic matter (humus) found in the forest floor and upper soil horizons as well as inorganic mineral particles such as silts and clays found lower in the profile. (Soil profile from Appalachian Mountains, Nantahala National Forest; photo by Daniel Neary).

surface heating. In general, the reduction in exchange capacity as the result of a fire is proportional to the amount of the total cation capacity that is provided by the organic component (DeBano and others 1998). The amount of cation exchange capacity remaining after a fire affects the leaching losses of soluble nutrients released during the fire. For example, the prefire cation exchange capacity of sandy soils may consist mainly of exchange sites found on the humus portion of the soil. If large amounts of humus are destroyed in these sandy soils during burning, then no mechanisms are available to prevent large losses of soluble nutrients by leaching.

The loss of cation exchange capacity, as the result of organic matter destroyed by fire, has been reported on by several authors. Soto and Diaz-Fierros (1993) intensively monitored changes in soil cation exchange capacity for one of the six soils they exposed to increasing temperatures. They found that cation exchange capacity decreased from 28.4 meq/100 g (28.4 cmol/kg) at 77 °F (25 °C) down to 1 meq/100 g (1 cmol/kg) when exposed to 1,292 °F (700 °C). The largest decrease occurred between 338 and 716 °F (170 and 380 °C), dropping from 28.1 to 6.9 meq/100 g (28.1 to 6.9 cmol/kg). Sands (1983) examined two adjacent radiata pine sites on sandy soils in Southeastern Australia 24 years after cutting. He found that the sites that received an intense site preparation burning before planting had decreased cation exchange capacity downward in soils to 20 cm compared to no changes on unburned naturally regenerated sites.

A stand replacement fire in the Southern Appalachians that resulted in a mosaic burn pattern similar to a wildfire produced a slight but significant decrease in cation exchange capacity 3 months after burning (Vose and others 1999). Also associated with the change in cation exchange capacity on midslope areas (medium-severity burn) was a decrease in exchangeable K and Mg, along with an increase in soil pH.

Cations

Cations found in the soil that are affected by fire include Ca, Mg, Na, K, and ammonia (NH_4), although these cations are not usually deficient in most wildland soils (DeBano 1991). In many studies, a significant increase in soil cation concentration following either prescribed burning or a wildfire has been reported (Grove and others 1986, Raison and others 1990, Soto and Diaz-Fierros 1993). The NH_4 cation, which is an important component of N cycling and soil productivity, responds differently from the other cations. With the exception of NH_4, cations have high temperature thresholds and, as a result, are not easily volatilized and lost from burned areas. The ash deposited on the soil surface during a fire contains high concentrations of cations, and their availability is increased, including NH_4 (Marion and others 1991, DeBano and others 1998; fig. 1.4). The amount of NH_4 released by burning depends upon fuel loading and the quantity of fuel combusted (Tomkins and others 1991). Some of the cations can be lost through particulate transfer in the smoke (Clayton 1976).

Monovalent cations, such as Na and K, are present largely as chlorides and carbonates that are readily mobilized (Soto and Diaz-Fierros 1993). Divalent ions, such as Ca and Mg, are less mobile and are commonly present as oxides and carbonates. The formation of insoluble calcium carbonate can occur, which limits the availability of P following fire. Although these readily available monovalent and divalent cations probably do not materially affect plant growth directly, their amount and composition determines base saturation, which plays an important role in controlling the pH regimes in soils (DeBano and others 1998).

Soil pH and Buffer Capacity

Soil *pH* is a measure of the hydrogen ion activity in the soil and is determined at specified moisture contents. Neutral soils have a pH of 7, acidic soils have a pH less than 7, and basic soils are those with a pH greater than 7. *Buffer capacity* is the ability of ions associated with the solid phase to buffer changes in ion concentration of the soil solution.

The combustion of organic matter during a fire and the subsequent release of soluble cations tend to increase pH slightly because basic cations are released during combustion and deposited on the soil surface. The increase in soil pH, however, is usually temporary depending upon the original soil pH, amount of ash released, chemical composition of the ash, and wetness of the climate (Wells and others 1979). The ash-bed effect discussed later in this chapter is an example of these factors in which large amounts of nutrients are deposited, with pH values being measurably changed by fire.

The pH of the soil is an important factor affecting the availability of plant nutrients (fig. 3.10). The nutrients released during a fire that are most likely to be affected are P, iron, and copper. P is particularly important because it is a macronutrient that is frequently limiting in wildland ecosystems, and it can also become insoluble at both high or low pHs (see part A). At low pH, P forms insoluble compounds with iron and at high pH, Ca compounds tend to immobilize it.

Nitrogen

Nitrogen is considered the most limiting nutrient in wildland ecosystems and as such it requires special consideration when managing fire, particularly in N-deficient ecosystems (Maars and others 1983). Nitrogen is unique because it is the only soil nutrient that is not supplied to the soil by chemical weathering of parent rock material. Almost all N found in the vegetation, water, and soil of wildland systems has to be added to the system from the atmosphere. A rare exception is the addition of some synthetic N-fertilizers that have been produced industrially and used for fertilizing forested areas. The cycling of N involves a series of interrelated complex chemical and biological processes (also see chapter 4). Only those cycling processes affecting chemical changes in N are discussed in this chapter (that is, N volatilization). Biologically mediated processes affecting N are discussed in more detail as part of chapter 4. The changes in N availability produced during fire are discussed later in this chapter in a section describing the effect of fire on nutrient availability

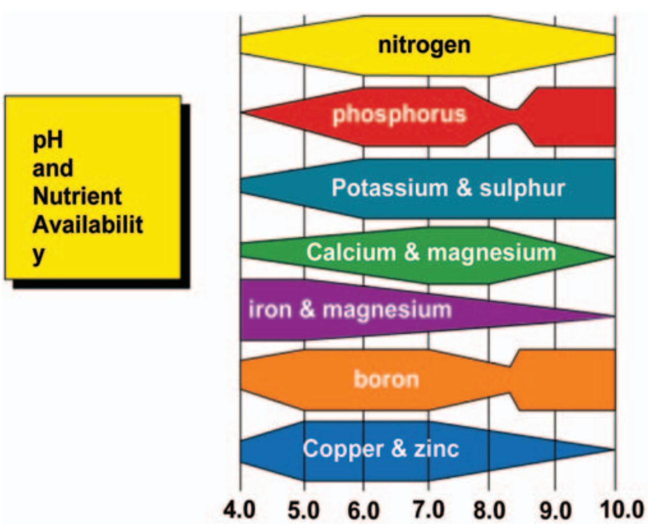

Figure 3.10—The availability of some common soil nutrients at different soil pH. (Figure courtesy of the USDA Forest Service, National Advanced Fire and Resource Institute, Tucson, AZ).

Responses to Soil Heating—Volatilization is the chemically driven process most responsible for N losses during fire. There is a gradual increase in N loss by volatilization as temperature increases (Knight 1966, White and others 1973). The amount of loss at different temperatures has established the following sequence of N losses upon heating:

- Complete loss (100 percent) of N occurs at temperatures above 932 °F (500 °C).
- Between 75 and 100 percent of the N is lost at temperatures of 752 to 932 °F (400 to 500 °C).
- Between 50 and 75 percent of the N is lost at temperatures of 572 to 752 °F (300 to 400 °C).
- Between 25 and 50 percent of the N is lost at temperatures of 392 to 572 °F (200 to 300 °C).
- No N losses occur at temperatures below 392 °F (200 °C).

As a general rule the amount of total N that is volatilized during combustion is directly proportional to the amount of organic matter destroyed (Raison and others 1985a). It has been estimated that almost 99 percent of the volatilized N is converted to N_2 gas (DeBell and Ralston 1970). At lower temperatures, N_2 can be produced during organic matter decomposition without the volatilization of N compounds (Grier 1975). The N that is not completely volatilized either remains as part of the unburned fuels or it is converted to highly available NH_4-N that remains in the soil (DeBano and others 1979, Covington and Sackett 1986, Kutiel and Naveh 1987, DeBano 1991).

Estimates of the total N losses during prescribed fire must be based on both fire behavior and total fuel consumption because irregular burning patterns are common. As a result, combustion is not complete at all locations on the landscape (DeBano and others 1998). For example, during a prescribed burn in southern California, total N loss only amounted to 10 percent of the total N contained in the plant, litter, and upper soil layers before burning (DeBano and Conrad 1978). The greatest loss of N occurred in aboveground fuels and litter on the soil surface. In another study of N loss during a prescribed fire over dry and moist soils, about two-thirds of the total N was lost during burns over dry soils compared to only 25 percent when the litter and soil were moist (DeBano and others 1979). Although these losses were relatively small, it should be remembered that even small losses can adversely affect the long-term productivity of N- deficient ecosystems. The importance of N losses from ecosystems having different pools of N is considered in more detail below.

Monleon and others (1997) conducted understory burns on ponderosa pine sites burned 4 months, 5 years, and 12 years previously. The surface soils, 0 to 2 inches (0 to 5 cm), showed the only significant response. The 4-month sites had increased total C and inorganic N following burning and an increased C/N ratio. Burning the 5-year-old sites resulted in a decrease in total soil C and N and a decrease in the C/N ratio. Total soil C and N in the surface soils did not respond to burning on the 12-year-old site.

Nitrogen Losses—An Enigma— It has been conclusively established by numerous studies that total N is decreased as a result of combustion (DeBano and others 1998). The amount of N lost is generally proportional to the amount of organic matter combusted during the fire. The temperatures at which N is lost are discussed above. In contrast, available N is usually increased as a result of fire, particularly NH_4-N (Christensen 1973, DeBano and others 1979, Carballas and others 1993). This increased N availability enhances postfire plant growth, and gives the impression that more total N is present after fire. This increase in fertility, however, is misleading and can be short-lived. Any temporary increase in available N following fire is usually quickly utilized by plants within the first few years after burning.

Nitrogen Losses and Ecosystem Productivity— The consequences of N losses during fire on ecosystem productivity depend on the proportion of total N lost for a given ecosystem (Barnett 1989, DeBano and others 1998). In N-limited ecosystems even small losses of N by volatilization can impact long-term productivity (fig. 3.11).

The changes in site productivity are related to the proportion of total N in the system that is lost. For example, the left portion of figure 3.11 represents a situation where large quantities of N are presented on a site having high productivity. Moving to the right side of the graph, both total N capital and productivity decrease. This decrease is not linear because there are likely to be greater losses in productivity per unit loss of N capital on sites having lower productivity (right side of fig. 3.11) than on sites having higher site productivity (left side of fig. 3.11). As a result, the losses in site productivity per unit N loss (a_h to b_h) from sites of high productivity (l_h) are less than losses in site productivity per unit N loss (a_l to b_l) from sites having low productivity (l_l). This relationship points to the importance of somehow replenishing N lost during a fire on low productivity sites or when using prescribed fire in these situations, taking special care not to consume large amounts of the organic matter present.

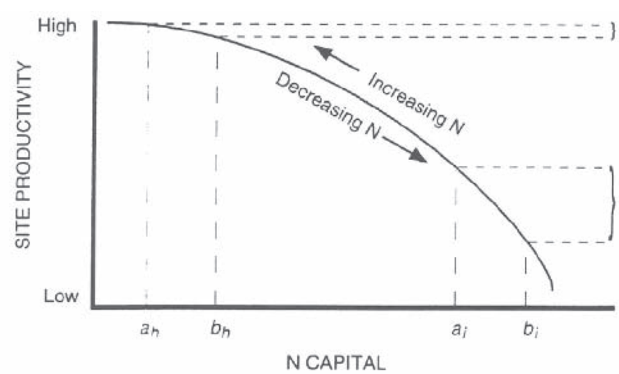

Figure 3.11—Relative importance of nitrogen low at different levels of site productivity. (After Barnett 1989).

Phosphorus

Phosphorus is probably the second most limited nutrient found in natural ecosystems. Deficiencies of P have been reported in P-fixing soils (Vlamis and others 1955) and as a result from N fertilization applications (Heilman and Gessel 1963). Phosphorus uptake and availability to plants is complicated by the relationship between mycorrhizae and organic matter and in most cases does not involve a simple absorption from the soil solution (Trappe and Bollen 1979). Phosphorus is lost at a higher temperature during soil heating than N, and only about 60 percent of the total P is lost by nonparticulate transfer when organic matter is totally combusted (Raison and others 1985a). The combustion of organic matter leaves a relatively large amount of highly available P in the surface ash found on the soil surface immediately following fire (see discussion on nutrient availability later in this chapter; also fig. 1.4). This highly available P, however, can be quickly immobilized if calcareous substances are present in the ash and thus can become unavailable for plant growth.

Sulfur

The role of S in ecosystem productivity is not well understood although its fluctuation in the soil is generally parallel to that of inorganic N (DeBano and others 1998). Sulfur has been reported as limiting in some coastal forest soils of the Pacific Northwest, particularly after forest stands have been fertilized with N (Barnett 1989). The loss of S by volatilization occurs at temperatures intermediate to that of N and P (Tiedemann 1987), and losses of 20 to 40 percent of the S in aboveground biomass have been reported during fires (Barnett 1989). Sulfur is similar to P (and unlike N) in that it cannot be fixed by biological processes, but instead is added primarily by burning fossil fuels (a source of acid rain), as fallout from volcanic eruptions, or by the weathering of rocks during soil development (DeBano and others 1998).

Soil Chemical Processes

Nutrient Cycling

Nutrients undergo a series of changes and transformations as they are cycled through wildland ecosystems. The sustained productivity of natural ecosystems depends on a regular and consistent cycling of nutrients that are essential for plant growth (DeBano and others 1998). Nutrient cycling in nonfire environments involves a number of complex pathways and includes both chemical and biological processes (fig. 3.3). Nutrients are added to the soil by precipitation, dry fall, N- fixation, and the geochemical weathering of rocks. Nutrients found in the soil organic matter are transformed by decomposition and mineralization into forms that are available to plants (see chapter 4). In nonfire environments, nutrient availability is regulated biologically by decomposition processes. As a result, the rate of decomposition varies widely depending on moisture, temperature, and type of organic matter. The decomposition process is sustained by litter fall (that is, leaf, wood, and other debris that falls to the forest floor). Through the process of decomposition, this material breaks down, releases nutrients, and moves into the soil as soil organic matter. Forest and other wildland soils, unlike agricultural soils where nutrients from external sources are applied as needed, rely on this internal cycling of nutrients to maintain plant growth (Perala and Alban 1982). As a result, nutrient losses from unburned ecosystems are usually low, although some losses can occur by volatilization, erosion, leaching, and denitrification. This pattern of tightly controlled nutrient cycling minimizes the loss of nutrients from these wildland systems in the absence of any major disturbance such as fire.

Fire, however, alters the nutrient cycling processes in wildland systems and dramatically replaces long-term biological decomposition rates with that of instantaneous thermal decomposition that occurs during the combustion of organic fuels (St John and Rundel 1976). The magnitude of these fire-related changes depends largely on fire severity (DeBano and others 1998). For example, high severity fires occurring during slash burning not only volatilize nutrients both in vegetation and from surface organic soil horizons, but heat is transferred into the soil, which further affects natural biological processes such as decomposition and mineralization (fig. 3.12; see also chapter 4). The effects of fire on soil have both short- and long-term consequences (that is, direct and indirect effects) on soil and site productivity because of the changes that occur in both the quantity and quality of organic matter.

In summary, many nutrients essential for plant growth including N, P, S, and some cations described earlier are all affected to some extent by fire. Nitrogen is likely the most limiting nutrient in natural systems (Maars and others 1983), followed by P and S. Cations released by burning may affect soil pH and result in the immobilization of P. The role of micronutrients in ecosystem productivity and their relationship to soil heating during fire is for the most part unclear. One study, however, did show that over half of the selenium in burned laboratory samples was recovered in the ash residue (King and others 1977).

Nutrient Loss Mechanisms

Nutrient losses during and following fire mainly involve chemical processes. The disposition of nutrients contained in plant biomass and soil organic matter during and following a fire generally occurs in one of the following ways:

- *Direct gaseous volatilization into the atmosphere takes place during fire*. Nitrogen can be transformed into N_2 along with other nitrogenous gases (DeBell and Ralston 1970).
- *Particulates are lost in smoke*. Phosphorus and cations are frequently lost into the atmosphere as particulate matter during combustion (Clayton 1976, Raison and others 1985a,b).
- *Nutrients remain in the ash deposited on the soil surface*. These highly available nutrients are vulnerable to postfire leaching into and through the soil, or they can also be lost during wind erosion (Christensen 1973, Grier 1975, Kauffman and others 1993).

Figure 3.12—Pinyon-juniper slash fire, Apache-Sitgreaves National Forest, Arizona. (Photo by Malchus Baker).

- *Substantial losses of nutrients deposited in the surface ash layer can occur during surface runoff and erosion*. These losses are amplified by the creation of a water-repellent layer during the fire (see chapter 2; DeBano and Conrad 1976, and Raison and others 1993).
- *Some of the nutrients remain in a stable condition*. Nutrients can remain onsite as part of the incompletely combusted postfire vegetation and detritus (Boerner 1982).

Although the direct soil heating effect is probably limited to the surface (1 inch or 2.5 cm), the burning effect can be measured to a greater depth due to the leaching or movement of the highly mobile nutrients out of the surface layers. For example, leaching losses from the forest floor of a Southern pine forest understory burn increased from 2.3 times that of unburned litter for monovalent cations Na and K to 10 to 20 times for divalent cations Mg and Ca (Lewis 1974). Raison and others (1990) noted that while K, Na, and Mg are relatively soluble and can leach into and possibly through the soil, Ca is most likely retained on the cation exchange sites. Soil Ca levels may show a response in the surface soils for many years following burning. However, some cations more readily leached and as a result are easily lost from the site. For example, Prevost (1994) found that burning *Kalmia* spp. litter in the greenhouse increased the leaching of Mg but none of the other cations. Although ash and forest floor cations were released due to burning, there was no change in surface soil cation concentrations (0-2 inches or 0-5 cm). Soto and Diaz-Fierros (1993) measured changes in the pattern of cation leaching at differing temperatures for the six soils that represented six different parent materials. Leaching patterns were similar for all soil types. Leaching of divalent cations, Ca, and Mg, increased as the temperatures reached during heating increased, with a peak at 860 °F (460 °C). Monovalent cations, K, and Na, differed in that initially leaching decreased as temperature increased, reaching a minimum at 716 °F (380 °C). Then, leaching increased up to 1,292 °F (700 °C). The nutrients leached from the forest floor and the ash were adsorbed in the mineral soil. Surface soils were found to retain 89 to 98 percent of the nutrients leached from the plant ash (Soto and Diaz-Fierros 1993). As the leachates moved through the mineral soil, the pH of the solution decreased.

Nutrient Availability

The increased nutrient availability following fire results from the addition of ash, forest floor leachates, and soil organic matter oxidation products as the result of fire. The instantaneous combustion of organic matter described earlier directly changes the availability of all nutrients from that of being stored and slowly becoming available during the decomposition of the forest floor organic matter to that of being highly available as an inorganic form present in the ash layer after fire. Both short- and long-term availability of nutrients are affected by fire.

Extractable Ions—Chemical ions generally become more available in the surface soil as a result of fire. Grove and others (1986) found that immediately after fire, extractable nutrients increased in the 0 to 1.2 inch (0-3 cm) depth. Concentrations of S, NH_4, P, K, Na, zinc (Zn), Ca, and Mg increased. Everything except Zn and organic C increased in terms of total nutrients. At the lower depths sampled, 1.2 to 3.9 inches and 3.9 to 7.9 inches (3-10 and 10-20 cm), only extractable P and K were increased by burning. One year later nutrient levels were still greater than preburn concentrations, but had decreased. A study on an area of pine forest burned by a wildfire reported that in the soil, concentrations of P, Ca, and Mg, aluminum (Al), iron (Fe) had increased in response to different levels of fire severity (Groeschl and others 1993). In the areas exposed to a high-severity fire, C and N were significantly lower and soil pH was greater. In another study the soil and plant composition changes were studied in a jack pine (*P. banksiana*) stand whose understory had been burned 10 years earlier (Lynham and others 1998). In this study the soil pH increased in all soil layers following burn—O horizon, 0-2 inches (0-5 cm), and 2-3.9 inches (5-10 cm)—and remained 0.5 units greater than preburn 10 years later in the O horizon. Phosphorus, K, Ca, and Mg all increased in the mineral soil; P and K were still greater than preburn levels 10 years later. A stand replacement fire in the Southern Appalachians that resulted in a mosaic burn pattern affected exchangeable ions (Vose and others 1999). The midslope areas of this fire burned at a moderate severity, and a decrease in exchangeable K and Mg along with an increase in soil pH was measured. Soil Ca, total C and N did not respond in any of the burned areas. There are other studies that have also shown no effect or decreases of soil nutrients following burning (Sands 1983, Carreira and Niell 1992, Vose and others 1999).

Nitrogen—The two most abundant forms of available N in the soil are available NH_4- and $N0_3$-N. Both forms are affected by fire. Burning rapidly oxidizes the soil organic matter and volatilizes the organic N contained in the forest floor and soil organic matter, thereby releasing NH_4-N (Christensen 1973, Jurgensen and others 1981, Kovacic and others 1986, Kutiel and Naveh 1987, Marion and others 1991, Knoepp and Swank 1993a,b). The release of NH_4-N has been found by more detailed chemical analysis to involve the thermal decomposition of proteins and other nitrogen-rich organic matter. Specifically, the production of

NH_4-N is related to the decomposition of secondary amide groups and amino acids. These secondary amide groups are particularly sensitive to decomposition during heating and decompose when heated above 212 °F (100 °C) to yield NH_4-N (Russell and others 1974). The volatilization of more heat-resistant N compounds can occur up to 752 °F (400 °C). The temperatures required to volatilize nitrogen compounds are increased by the presence of clay particles in the soil (Juste and Dureau 1967).

Most NH_4-N that is volatilized is lost into the atmosphere, but significant amounts can move downward and condense in the mineral soil as exchange N. The ash produced by the fire can also contain substantial amounts of NH_4-N. As a result of these two processes, the inorganic N in the soil increases during fire (Kovacic and others 1986, Raison and others 1990, Knoepp and Swank 1993a,b). In contrast, $N0_3$-N is usually low immediately following fire and increases rapidly during the nitrification of NH_4 (see chapter 4). These $N0_3$-N concentrations may remain elevated for several years following fire (fig. 3.13).

The production of $N0_3$-and NH_4-N by fire depends on several factors. These include fire severity, forest type, and the use of fire in combination with other postharvesting activities.

Effect of fire severity: The amounts of NH_4-N that are produced as a result of fire generally increase with the severity and duration of the fire and the associated soil heating (fig. 3.13, 3.14, 3.15). Although large amounts of the total N in the aboveground fuels, litter, duff, and upper soil layers are lost into the atmosphere by volatilization, highly available NH_4-N still remains in the ash or in the upper mineral soil layers following fire (fig. 3.13). During both high and low severity fires, increase occurs in the amounts of NH_4-N that can be found both in the ash remaining on the soil surface following fire and in the upper mineral soil layers (Groeschl and others 1990, Covington and Sackett 1992). The

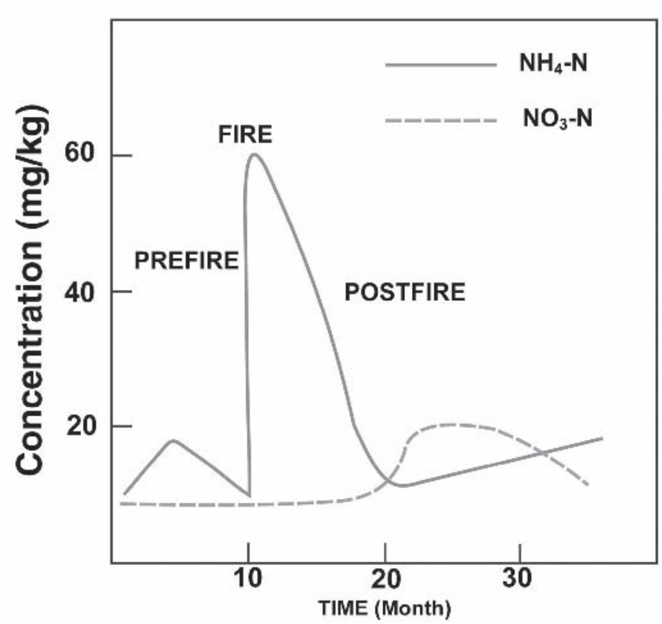

Figure 3.13—Soil NO_3-N and NH_4-N concentrations before, immediately following, and for several months following fire in Arizona chaparral. (Adapted from DeBano and others 1998. Fire's Effects on Ecosystems. Copyright © 1998 John Wiley & Sons, Inc. Reprinted with permission of John Wiley & Sons, Inc.).

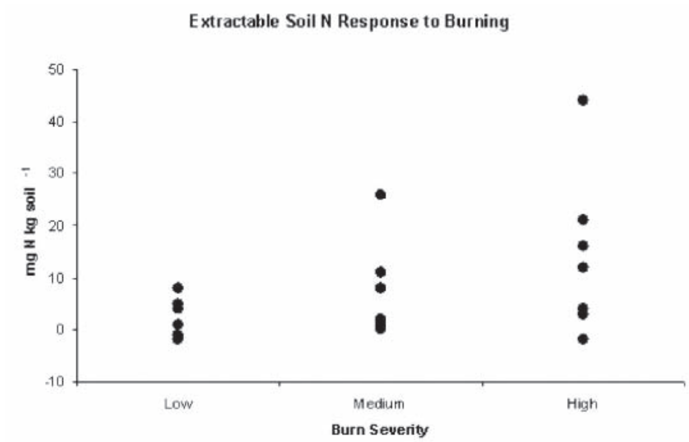

Figure 3.14—Extractable soil N in response to burn severity measured on eight studies.

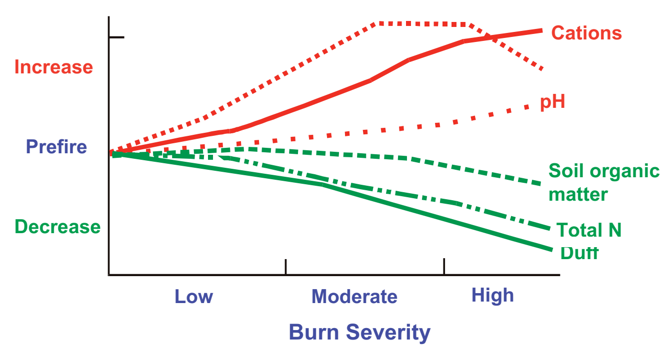

Figure 3.15—Generalized patterns of decreases in the forest floor (duff), total N, and organic matter, and increases in soil pH, cations, and NH_4 associated with increasing levels of fire severity. (Figure courtesy of the USDA Forest Service, National Advanced Fire and Resource Institute, Tucson, AZ).

concentrations of NH_4-N found in the ash depend on the severity of the fire and amount of forest floor consumed, which in turn reflects stand age. During low severity fires, less of the forest floor is consumed and correspondingly less amounts of NH_4-N are produced (fig. 3.15). The amounts of NH_4-N produced during a fire may vary from increases of less than 10 ppm (10mg/kg) during low severity fires to 16 and 43 ppm (16 mg/kg and 43 mg/kg) during medium and high severity fires, respectively. The variability in the amounts of NH_4-N produced also increases with fire severity. The NH_4-N concentrations following burning are short lived (fig. 3.13), although Covington and Sackett (1992) reported that NH_4-N levels remained elevated for at least 1 year following a prescribed burn in ponderosa pine stands in northern Arizona.

The direct effects of fire on $N0_3$-N concentrations are less predictable. For example, the results of a study in the Shenandoah National Park showed that although total extractable inorganic N was elevated for 1 year following the fire, the $N0_3$-N on areas burned by either high or low severity fires increased only slightly (Groeschl and others 1990). A more common scenario for $N0_3$-N changes following fire results from the increased nitrification of the highly available N produced directly during the fire (for example, NH_4-N). The end result being that NH_4-N produced directly during fire is rapidly nitrified, and $N0_3$-N begins to increase following fire, depending on temperature and moisture conditions (fig. 3.13)

The effects of these increased levels of highly available N during and following fire are often beneficial to the recovering plants by providing a temporary increase in site fertility. However, these short-term benefits must be carefully weighed against the overall and long-term effect that the loss of total N during a fire has on the sustained productivity of the site (see the previous discussion on Nitrogen Losses –An Enigma).

Effect of forest type: Forest type can also affect the amount of NH_4-N produced by fire. This occurs mainly as a result of nature of the fire behavior and the amounts of litter that accumulate under different forest types. Such was the case in a study conducted in Idaho on pine/hemlock sites compared to sites occupied by a Douglas-fir/western larch forest (Mroz and others 1980). In this study the NH_4-N did not increase following burning of the pine/hemlock forest in contrast to the Douglas-fir/western larch forest site where it did increase. Within 3 to 7 days, however, mineralization and/or nitrification had begun on most sites.

Frequency of burning: Repeated burning and its frequency are often-asked questions by fire managers when conducting prescribed burns. In terms of N availability, the effects of burning frequency depends largely upon the ecosystem, its inherent fertility (in terms of total and available N), total amounts of organic matter destroyed, and its ability to replenish the N lost by volatilization (DeBano and others 1998).

Several studies have focused on the effect of different frequencies of prescribed burning on changes in total and concentrations of total and available nitrogen. For example, a study on the effect of repeated burn low-intensity burning in Australian eucalypt forests (*Eucalyptus* spp.)showed that fire-free periods of about 10 or more years were required to allow natural processes time to replace the amount of N lost during the burning, assuming that about 50 percent of the total N in the fuel was volatilized (Raison and others 1993). In contrast, a study of repeated burning at 1-, 2-, and 5-year intervals in ponderosa pine forests in northern Arizona showed no significant differences in total N among the different burning frequencies, but available N (NH_4- and $N0_3$-N) was higher on the sites that repeatedly burned in comparison to the unburned controls (Covington and Sackett 1986). Researchers concluded from this Arizona study that frequent periodic burning can be used to enhance N availability in Southwestern ponderosa pine forests.

Studies on the effects of burning frequency on grasslands and shrubs have been reported to have less desirable outcomes. For example, the annual burning of tall grass prairies in the Great Plains of the Central United States resulted in greater inputs of lower quality plant residues, causing a significant reduction in soil organic N, lower microbial biomass, lower N availability, and higher C:N rations in soil organic matter (Ojima and others 1994). Likewise, increases in available N may have adverse effects on some nutrient-deficient shrub ecosystems as has been reported by a study in a shrubland (fynbos) in South Africa. In a study of lowland fynbos, a twofold increase in soil nutrient concentrations produced by fire were detrimental to the survival of indigenous species that had evolved on these nutrient-impoverished landscapes (Musil and Midgley 1990).

Phosphorus—Responses of available soil P to burning are variable and more difficult to predict than those of other nutrients (Raison and others 1990). Phosphorus volatilizes at temperatures of about 1,418 °F (770 °C). The fate of this volatilized P is not well understood. One study indicated that the only response was on the surface soil, and P did not appear to move downward in the soil via volatilization and condensation, as N does (DeBano 1991). Grove and others (1986) found the opposite. They measured responses in all major cations, S, NH_4, and Zn following burning in the surface 0 to 1.2 inches (0-3 cm) of soil. In their study, only P and K concentrations also responded in the lower soil depths (1.2-3.9 inches and 3.9-7.9 inches; 3-10 cm and 10-20 cm).

As in the case of N, fire severity affects changes in extractable P. During high-severity fires, 50 to 60 percent of the total fuel P might be lost to volatilization (Raison and others 1990, DeBano 1991). Part of this volatilized P ends up as increased available P in both the soil and ash following burning. An extensive study of P responses to different burning severities was reported for eucalypt forests (Romanya and others 1994). The study sites included unburned, burned, and in an ash bed found under a burned slash pile. The greatest effects occurred in the surface soil (0-1 inch; 0-2.5 cm), and the response was dependent on fire severity. Extractable P concentrations increased with increasing fire severity, but the response decreased with depth. Organic P on the other hand reacted oppositely; concentrations were lower in the intensively burned areas and greater in the unburned and low-severity burned sites.

Fire affects the enzymatic activity and mineralization of P. One study compared these P responses in a controlled burn versus a wildfire (Saa and others 1993). When temperatures reached in the forest floor of the controlled burn were less than 329 °F (50 °C), extractable P concentrations (ortho-phosphate) showed no significant response. In contrast, a wildfire that produced higher soil temperatures reduced phosphatase activity and increased the mineralization of organic P, which increased ortho-phosphate P and decreased organic P. Laboratory experiments showed that phosphatase activity can be significantly reduced when heating dry soils but was absent in wet soils (DeBano and Klopatek 1988). In the pinyon-juniper soils being studied, bicarbonate extractable P was increased although the increases were short lived.

Ash-Bed Effect

Following fire, variable amounts of ash are left remaining on the soil surface until the ash is either blown away or is leached into the soil by precipitation (fig. 1.4). On severely burned sites, large layers of ash can be present (up to several centimeters thick). These thick accumulations of ash are conspicuously present after piling and burning (for example, burning slash piles). Ash deposits are usually greatest after the burning of concentrated fuels (piled slash and windrows) and least following low-severity fires.

The accumulation of thick layers of ashy residue remaining on the soil surface after a fire is referred to as the "ash bed effect" (Hatch 1960, Pryor 1963, Humphreys and Lambert 1965, Renbuss and others 1972). The severe burning conditions necessary to create these thick beds of ash affect most of the physical, chemical, and biological soil properties. Soil changes associated with ash beds can occur as a result of a fire itself (soil heating), the residual effect of the ash deposited on the soil surface (that is, the ash bed), or a combination of both (Raison 1979).

The amount and type of ash remaining after fire depend upon the characteristics of the fuels that are combusted, such as fuel densities (packing ratios), fuel moisture content, total amount of the fuel load consumed, and severity of the fire (Gillon and others 1995). As a result of the fire, the ash remaining after a fire can range from small amounts of charred dark-colored fuel residues to thick layers of white ash that are several centimeters thick (DeBano and others 1998). When densely packed fuels are completely combusted, large amounts of residual white ash are usually in one place on the soil surface following burning (such as after piling and burning slash). The severe heating during the fire will change the color of the soil mineral particles to a reddish color, and where extreme soil heating has occurred, the mineral soil particles may be physically fused together. Silicon melts at temperatures of 2,577 °F (1,414 °C; see chapter 2).

Chemically, fire consumption of aboveground material determines the amount of ash produced. Ash consists mostly of carbonates and oxides of metals and silica along with small amounts of P, S, and N (Raison and others 1990). Calcium is usually the dominant cation found in these ash accumulations. Most of the cations are leached into the soil where they are retained on the cation exchange sites located on clay or humus particles and increase the mineral soil cation content (fig. 3.15). The pH may exceed 12. However, the composition of the preburn material and the temperature or severity of the fire determines the chemical properties of ash. Johnston and Elliott (1998) found that ash on uncut forest plots generally had the highest pH and the lowest P concentrations.

Physical changes associated with the ash bed effect mainly include changes in soil structure and permeability to water. The combustion of organic matter in the upper part of the soil profile can totally destroy soil structure, and the ashy material produced often seals the soil to water entry.

The biological impact of the ash bed effect is twofold. During the fire the severe soil heating can directly affect the long-term functioning of microbial populations because the high temperature essentially sterilizes the upper part of the soil. Plant roots and seeds are also destroyed so that the revegetation of these sites depends on long-term ecological succession to return to its former vegetative cover. Indirectly, the large amounts of ash can affect soil microbial populations. A study of the effects of ash, soil heating, and the ash-heat interaction on soil respiration in two Australian soils showed that large amounts of ash slightly decreased respiration, but small amounts had no effect (Raison and McGarity 1980). Additions of ash to sterilized soil produced no effect, indicating that ash acted via its influence on active soil biological populations. The chemical nature of ash was hypothesized to affect soil respiration by its effect on:

- Increasing pH.
- Changing the solubility of organic matter and associated minerals in water.
- Adding available nutrients for microbial populations.

Management Implications

Understanding the effects of fire on soil chemical properties is important when managing fire on all ecosystems, and particularly in fire-dependent systems. Fire and associated soil heating combusts organic matter and releases an abundant supply of highly soluble and available nutrients. The amount of change in the soil chemical properties is proportional to the amount of the organic matter combusted on the soil surface and in the underlying mineral soil. Not only are nutrients released from organic matter during combustion, but there can also be a corresponding loss of the cation exchange capacity of the organic humus materials. The loss of cation exchange capacity of the humus may be an important factor when burning over coarse-textured sandy soils because only a small exchange capacity of the remaining mineral particles is available to capture the highly mobile cations released during the fire. Excessive leaching and loss can thus result, which may be detrimental to maintaining site fertility on nutrient-limiting sandy soil.

An important chemical function of organic matter is its role in the cycling of nutrients, especially N. Nitrogen is most limiting in wildland ecosystems, and its losses by volatilization need to be evaluated before conducting prescribed burning programs. Nitrogen deficiencies often limit growth in some forest ecosystems. Xeric and pine dominated sites, which are typically prone to burning, often exhibit low N availability, with low inorganic N concentrations and low rates of potential mineralization measured on these sites (White 1996, Knoepp and Swank 1998). Forest disturbance, through natural or human-caused means, frequently results in an increase of both soil inorganic N concentrations and rates of potential N mineralization and nitrification. The N increases resulting from a combination of changes in soil moisture and temperature and the decreased plant uptake of N make more N available for sustaining microbial populations in the soil.

Historically, some wildland ecosystems have been exposed to frequent fire intervals. Many of these ecosystems are low in available N and other nutrients such as P and cations. The cycling of nutrients, especially N, may be slow, and the exclusion of fire from these systems often results in low N mineralization and nitrification rates. Frequent fire, however, can accelerate these biological rates of N mineralization because it destroys the inhibiting substances that hinder these processes. For example, in ponderosa pine forests in the Southwest, monoterpenes have been found to inhibit nitrification (White 1991). These monoterpenes are highly flammable and as a result are combusted during a fire. As a result, the removal of this inhibition by fire allows N mineralization and nitrification to proceed. It is hypothesized that these inhibitory compounds build up over time after a fire and decrease N mineralization. Significant differences in monoterpenes concentrations have been established between early and late successional stages, although specific changes over time have not been detectable because of the large variability between sites (White 1996).

Another study has shown that the xeric pine-hardwood sites in the Southern Appalachians are disappearing because of past land use, drought, insects, and the lack of regeneration by the fire-dependent pine species (Vose 2000). This information was used to develop an ecological model that could be used as a forest management tool to rejuvenate these *Pinus rigida* stands (fig. 3.16). This model specifies that a cycle of disturbance due to drought and insect

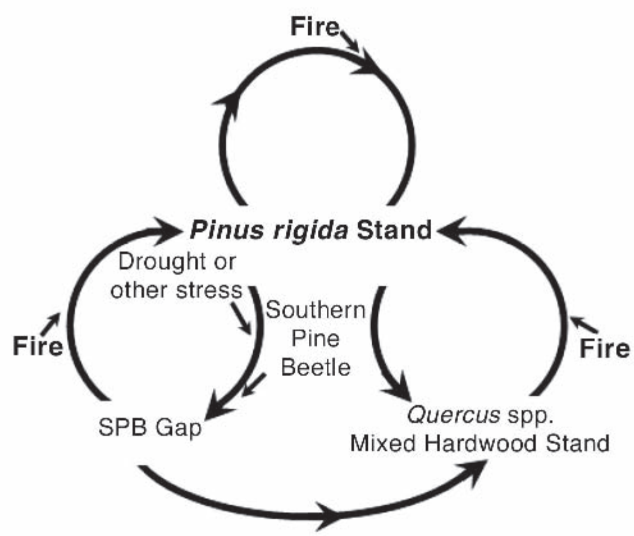

Figure 3.16—Effect of fire on nutrient cycling processes in *Pinus ridgida* stands. (Adapted from Vose 2000).

outbreaks followed by fire is necessary to maintain the pine component of these ecosystems. Without fire, mixed hardwood vegetation dominates the stand. Therefore, prescribed burning is proving to be an effective tool for enhancing ecosystem health and for sustaining, preserving, and restoring these unique habitats.

Fire severity is probably the one most important feature of a fire that affects the chemical soil properties. Generalized relationships for several soil properties at different severities are presented in figure 3.15. Nitrogen, organic matter, and duff decrease as fire severity increases. Available NH_4-N and cations increase. The pH of the soil generally increases because of the loss of organic matter and its associated organic acids, which are replaced with an abundance of basic cations in the ash.

Summary

The most basic soil chemical property affected by soil heating during fires is organic matter. Organic matter not only plays a key role in the chemistry of the soil, but it also affects the physical properties (see chapter 2) and the biological properties (see chapter 4) of soils as well. Soil organic matter plays a key role in nutrient cycling, cation exchange, and water retention in soils. When organic matter is combusted, the stored nutrients are either volatilized or are changed into highly available forms that can be taken up readily by microbial organisms and vegetation. Those available nutrients not immobilized are easily lost by leaching or surface runoff and erosion. Nitrogen is the most important nutrient affected by fire, and it is easily volatilized and lost from the site at relatively low temperatures. The amount of change in organic matter and N is directly related to the magnitude of soil heating and the severity of the fire. High- and moderate-severity fires cause the greatest losses. Nitrogen loss by volatilization during fires is of particular concern on low-fertility sites because N can only be replaced by N-fixing organisms. Cations are not easily volatilized and usually remain on the site in a highly available form. An abundance of cations can be found in the thick ash layers (or ash-bed) remaining on the soil surface following high-severity fires.

Notes

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Chapter 4: Soil Biology

Introduction

Soil biological properties involve a wide range of living organisms that inhabit the soil, along with the biologically mediated processes that they regulate. The welfare of these soil organisms directly affects the short- and long-term productivity and sustainability of wildland ecosystems (Borchers and Perry 1990). Soils are alive with large populations of microorganisms, roots and mycorrhizae, invertebrates, and burrowing animals that inhabit the upper part of the soil profile (Singer and Munns 1996). The biological component of soil also includes plant roots and their associated rhizosphere, vegetative reproductive structures, and seeds. These organisms proliferate in the soil matrix, particularly the upper layers that contain substantial amounts of organic matter. Collectively, these organisms contribute to soil productivity by enhancing decomposition, nitrogen (N) cycling, humus formation, soil physical and chemical properties, plant reproduction, disease incidence, and plant nutrition and stability.

Biological Components of Soils

The biological component of soils is made up of both living and dead biomass. Dead biomass consists of organic matter that is in various stages of decomposition, extending from undecomposed plant parts in the litter layer to highly decomposed humus materials that can be thoroughly mixed with the upper mineral layers of the soil profile. Both living and dead components are affected by fire. The effect of fire on organic matter is discussed in detail in chapter 3. This current chapter is devoted to the description of the important organisms that influence soil-litter systems and the effects that fire has upon them.

Living organisms can be classified several ways. One method of classification is whether they are flora or fauna. Soil flora includes algae, cyanobacteria, mycorrhiza, and plant roots. Soil fauna includes protozoa, earthworms, and insects. The category "soil fauna" has been further divided into micro-, meso-, and macrofauna based on body lengths of less than 0.2 mm, .20 to 10.4 mm, and greater than 10.4 mm, respectively (Wallwork 1970) (fig. 4.1). The term "*microorganisms*" encompasses a diverse group including bacteria, fungi, archaebacteria, protozoa, algae, and viruses. Some common representatives of the intermediate-sized soil organisms (that is, mesofauna) commonly found in soils are round worms, springtails, and mites, whereas the macrofauna are represented by a wide range of larger invertebrates such as many insects, scorpions, and earthworms.

Soil organisms ranging in size from microbes to megafauna are commonly concentrated in the surface horizons, because these soil layers contain large amounts of organic matter (see fig. A.1, part A), and are active sites for microbial processes, including decomposition and mineralization. Plant roots and invertebrates can occupy the forest floor (that is, L-, F-, and H- layer) or can be found in the uppermost layers of organic-rich mineral soil (that is, the upper part of the A-horizon). The H-layer represents the end product of the microbial decomposition activity that occurs in many soils.

Soil Microorganisms

Soil is teeming with life. Hundreds of millions of microorganisms are found in each handful of forest soil. No other living component in a forest (vegetation, wildlife, insects, and so forth) comes close to matching the sheer numbers and diversity of soil microorganisms. More important, microorganisms play a major role in nutrient cycling processes, decomposition of organic material, improvement of soil physical characteristics, and disease. They also play an important role in providing a labile pool of nutrients (Stevenson and Cole 1999). Hence, their influence on life in naturally occurring ecosystems is substantial. Also, some microorganisms form symbiotic relationships with plants, thereby creating a unique biological entity that can be easily affected by fire.

Free-Living Fungi and Bacteria—*Fungi* and *bacteria* are the workhorses of the forest soil organisms (fig. 4.2). Their functions in part include decomposition, nutrient turnover and acquisition (for

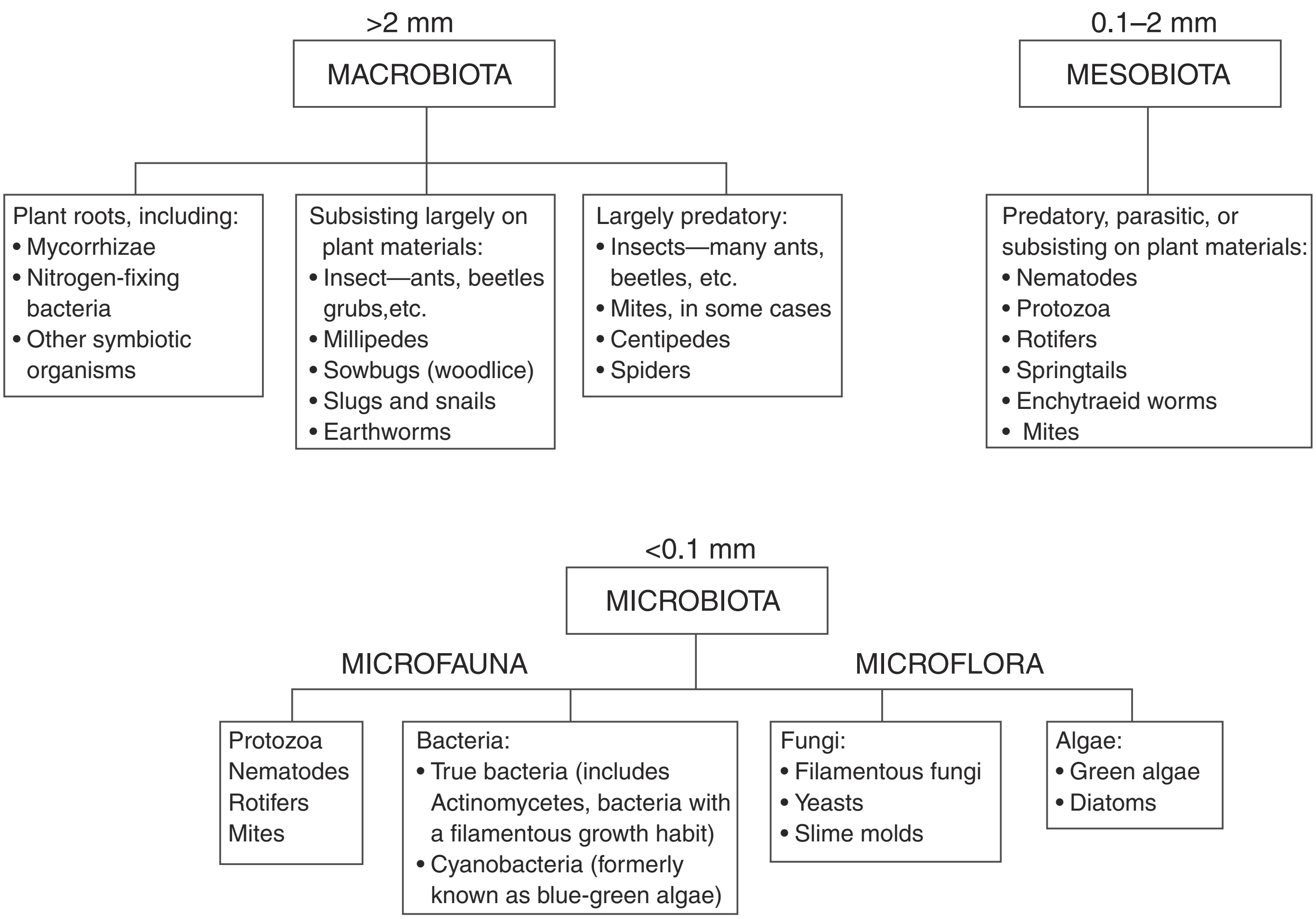

Figure 4.1—Types and sizes of soil organisms that can be affected by fire. (Adapted from Wallwork 1970).

Figure 4.2—Fungi (A) and bacteria (B) are important forest soil organisms due to their roles in decomposition, nutrient turnover and acquisition, disease occurrence, and suppression and degradation of toxic materials. (Photos by Daniel Neary and Shirley Owens, Michigan State University).

example, mycorrhizal fungi and N-fixing bacteria), disease occurrence and suppression, and degradation of toxic materials. Bacteria (including actinomycetes and cyanobacteria) are the most numerous, commonly numbering about 100 million individuals in a gram of fertile agricultural topsoil (Paul and Clark 1989, Singer and Munns 1996). In forest soils, the numbers of bacteria are less, but fungal numbers can range from 5,000 to 900,000 individuals per gram of soil (Stevenson and Cole 1999). Bacteria and fungi have been estimated to contribute 2.2 tons/acre (5.0 Mg/ha) of biomass to some forest soils (Bollen 1974). Fungi, although less numerous than bacteria, often account for greater biomass due to their larger size and dominance in woody, organic material. The diversity of these two groups is almost unimaginable; thousands of species of each can be found in 0.002 pound (1 g) of soil (Torsvik and others 1990, Molina and others 1999). And this diversity creates the collective ability of these microorganisms to adapt to fire or other environmental disturbances (Atlas and others 1991).

Other microbial groups also serve key roles in the biologically induced changes in forest floor litter and soil. Protozoa, by feeding on bacteria, release plant-available nutrients that were previously tied up in bacterial cells. The role of viruses and archaebacteria (formerly classified as bacteria) in forests, however, is not well understood. These viruses are protein-coated, acellular strands of DNA or RNA that can be predatory to many microorganisms, suggesting a contribution to nutrient turnover. Archaebacteria are common in extreme environments where drastic temperature, moisture, acidity, or nutrient conditions preclude other organisms. Their numbers in forest soils are assumed to be low.

A particularly important group of free-living microorganisms includes those concerned with the nutrient cycling processes described later in this chapter. Some of these biologically mediated processes include N fixation, mineralization, ammonification, nitrification, and the overall decomposition process.

Specialized Root-Microbial Associations—Some bacteria and fungi are in close association with plant roots and develop a symbiotic relationship with them. This suite of organisms is different than the free-living bacteria and fungi described above because they depend on a mutual relationship with plant roots. They are also distinct in their relationship to fire because both they and their host plants can be affected. The most common symbiotic relationships involve the root and:

- Mycorrhizal fungi, which enhance the plant's ability to obtain nutrients such as phosphorus (P) and zinc from the soil.
- Nitrogen-fixing bacteria, which convert N gas in the air to a form usable by plants. The effects of fire on N-fixation and other biological processes are discussed in more detail later in this chapter.

A useful concept used for describing the close relationship between roots and microorganisms is the *rhizosphere*. The rhizosphere is a cylindrical volume of the soil space that extends about 0.04 inch (1 mm) from the surface of roots (Singer and Munns 1996). The outer boundary of the rhizophere is diffuse and inexactly defined because the effect of the root can extend variable distances into the soil. Simply, the rhizosphere may be thought of as including the root and its surrounding soil environment (fig. 4.3). Functionally, the rhizosphere is important because it contains a combination of the roots and associated microorganisms. The roots secrete products that stimulate

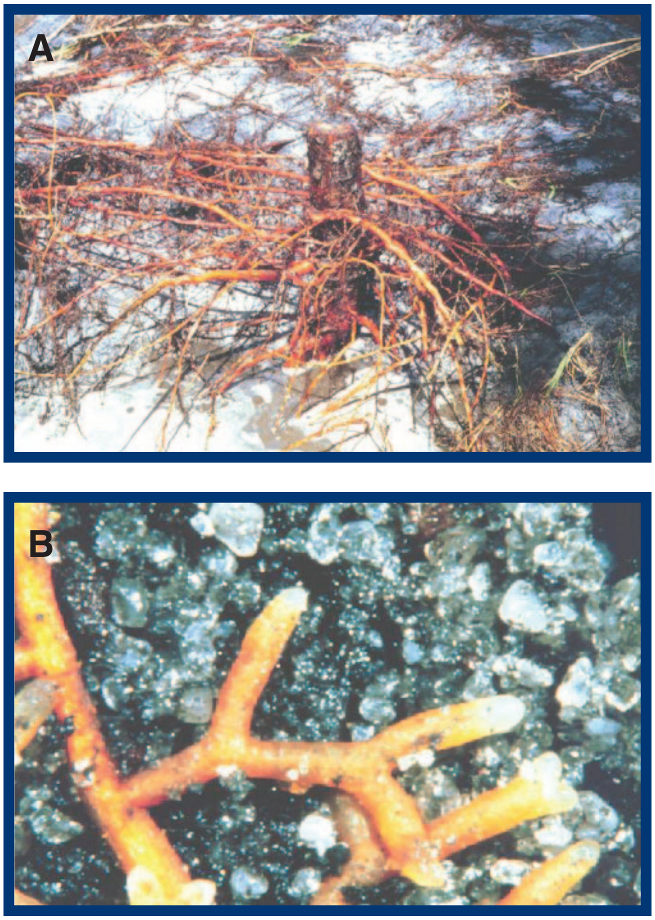

Figure 4.3—The rhizosphere includes the root system (A), associated mycorrhizae, and varying volumes of the soil surrounding roots (B). (Photos courtesy of Nicholas Comerford, University of Florida).

the bacterial and fungal activity so that the fast-growing heterotrophic microbes are at least 10 times denser near the root than in the rest of the soil (Singer and Munns 1996). The heterotrophic microbial population plays an important role in the decomposition of organic matter and contributes to a desirable soil structure (granular) in the root zone.

Mycorrhizal fungi found in the rhizosphere depend on host plants for their well being (Borchers and Perry 1990). The development of mycorrhizae provides a way for the plant roots to extend farther into the soil. The thin fungal hyphae of the mycorrhizal fungi form a mutual relationship with the roots of some plants, thereby allowing the plant roots to proliferate a greater soil mass than by the roots alone. Root-fungal associations use about 5 to 30 percent of the total photosynthate that is translocated below ground by plants. Two types of mycorrhizae are found in soil, endo- (arbuscular) and ectomycorrhizae.

The endomycorrhizae produce structures within the plant roots (in deciduous trees, most annual crops, and other herbaceous species) that are called arbuscules (Coleman and Crossley 1996, Singer and Munz 1996). Arbuscular mycorrhiza fungi also send out hyphae, but only a few centimeters into the surrounding soil (Coleman and Crossley 1996).

In contrast to endomycorrhize fungi, the ectomycorrhizae, which are primarily basidiomycetes, grow between plant root cells (in many evergreen trees and shrubs), but not inside them as do endomycorrhizae (Coleman and Crossley 1996, Munns 1996). Ectomycorrhizae can send out hyphae for several meters into the surrounding soil to forage for nutrients and water that are essential for the host plant. Because of their ability to proliferate the soil, hyphae of ectomycorrhizae constitute a significant proportion of the C allocated to belowground net primary productivity in coniferous forests (Read 1991). The hyphae facilitate nutrient uptake, particularly of P, and are avid colonizers of organic matter where they enhance soil structure. Ectomycorrhizae are located at shallow depths in the soil profile and tend to be concentrated in the woody material during dry seasons and in the H-layer during moist conditions. Ectomycorrhizae are important decomposers and, as a result, obtain reduced C from the decomposing litter layer. Because the ectomycorrhizae are so near the soil surface, both the resting stages and the hyphae are easily damaged by soil heating during a fire.

Several groups of microorganisms form N-fixing symbiotic relationships with plants (Singer and Munns 1996). This type of symbiosis, commonly found in rhizosphere, involves a group of actinomycetes and rhizobia bacteria. The actinomycetes (*Frankia* spp.) infect the roots of many genera of shrubs and trees where they form N-fixing nodules. Rhizobial bacteria (*Rhizobium* and *Bradyrhizobium* spp.) also form nitrogen-fixing nodules with a large variety of plants belonging to the legume family, including alfalfa, clover, bean, pea, soybean, vetch, lupine, and lotus, among many others. Many of these are agricultural plants, although lupines, lotus, and clover are frequently found in abundance on freshly burned wildland areas.

Biological Crusts—*Biological crusts* (fig. 4.4) are found in hot, cool, and cold arid and semiarid regions throughout the world and frequently occupy the bare areas where vegetation cover is spare or totally absent (Belnap 1994, Belnap and others 2001). These surface communities are generically referred to as biological soil crusts, although they may specifically be called cryptogamic, cryptobiotic, microbiotic, or microphytic soil crusts. The biological soil crusts are made up of a complex community of cyanobacteria (blue-green algae), lichens, mosses, microfungi, and other bacteria

Figure 4.4—Biological crusts are found in hot, cool, and cold arid and semiarid regions through the world. (Photo courtesy of the U.S. Geological Survey).

(Isichei 1990, Johansen 1993, Loftin and White 1996, Belnap and others 2001). The algal component of this community has the ability to fix atmospheric C and is best recognized in forests as a component of lichens that colonize rocks, tree trunks and exposed soil surfaces. Lichens represent a symbiosis among fungi and blue-green algae or cyanobacteria. The blue-green algae photosynthesize and fix N, while the fungi provide water and mineral nutrients. They are typically found on erodible soils with some topographic relief, or in shallow soil pockets in slickrock habitats (Johansen 1993). In this environment, the filamentous growth of cyanobacteria and microfungi proliferates in the upper few millimeters of the soil, gluing loose particles together and forming a matrix that stabilizes and protects the soil surface from erosion (Belnap and others 2001). Cryptogamic crusts are also found in semiarid forest and shrublands, such as pinyon-juniper woodlands and sagebrush communities (Johansen 1993). These biological crusts can account for as much as 70 percent of all living ground cover in arid Western Hemisphere ecosystems (Belnap 1994).

The distribution of biological crusts is influenced by several environmental factors (Belnap and others 2001). The total amount of crust that develops is inversely related to vascular plant cover (in other words, the less plant cover, the more surface available for colonization and growth). Elevation also affects crust distribution and cover, and both are greatest at low elevation inland areas (less than 3,300 feet or 1,000 m) compared to mid-elevations 3,300 to 8,200 feet (1,000 to 2,500 m). Stable soils and rocks near, or at, the soil surface enhance crust cover by collecting water and armoring the surface against physical disturbance (Belnap and others 2001). Stable, fine-textured soils support a greater percentage cover and a more diverse population of organisms making up these surface crusts. The season of precipitation also has a major influence on the dominance of biological crusts, and as a result, the regions that receive monsoons have the greatest diversity of cyanobacteria and the lowest abundance of lichens. Areas that are frequented by fog (such as portions of the California chaparral) frequently support lichens that intercept moisture from the air. Finally, but not least, surface disturbance has a strong influence on the welfare of biological crusts. The two most important historical impacts on crusts have been grazing and fire (Belnap and others 2001).

Cryptogamic crusts benefit soils in a number of ways. The ability of these crusts to fix N, accumulate C, and capture P enhances nutrient cycling in soils, especially during the early successional stages (DeBano and others 1998, Evans and Johansen 1999). Some specific benefits of cryptogamic crusts in soils arise from their ability to:

- Retard erosion on desert and steppe rangelands by binding individual soil particles (particularly in sandy soils). Soil particles are believed to be bound as a result of the production of extracellular polysaccharides (Lynch and Bragg 1985).The crusts can increase or decrease infiltration rates, although their effect on conserving soil moisture is not well-defined (Johansen 1993).
- Fix gaseous N. The cyanobacteria component of this symbiosis has been estimated to fix as much as 22 pounds/acre (25 kg/ha) of N annually (West and Skujins 1977). The amount of N fixed, however, is largely dependent upon the abundance and activity of the crusts and favorable climatic conditions (Loftin and White 1996).
- Enhance organic matter buildup. Biological crusts in semiarid areas of the Southwestern United States have been estimated to accumulate 5.3 to 20.5 pounds/acre (6 to 23 kg/ha) of C annually (Jeffries and others 1993).
- Increase P levels by retaining fine soil particles from loss by erosion (Kleiner and Harper 1977).
- Facilitate seedling establishment. Although germination can be inhibited by allelopathic substances produced by the crust, seedling establishment generally seems to benefit by their presence (St. Clair and others 1984).

Soil Meso- and Macrofauna

The most common members of the mesofaunal group of soil organisms are the mites (members of the order Acari). The mites can be abundant in soils, particularly forest soils, where a 0.2 pound (100 g) sample of soil may contain as many as 500 mite species representing almost 100 genera (Coleman and Crossley 1996). Other mesofauna found in soil include *Rotifera*, *Nematoda* (round worms), and *Collembola* (springtails).

Macrofauna occupying the soil and litter in forest, shrubland, and grassland can be placed in two broad classes—those that spend all or most of their time in the litter and uppermost mineral soil layers, and those that inhabit these habitats only temporarily or not at all. Several of the more permanent faunal groups include those that reside in the soil and litter but also a group that dwells beneath stones, logs, under bark, or in similar protected habitats. These organisms are collectively called the cryptozoa. Three orders of higher insects—Isoptera (termites), Hymenoptera (ants, bees, wasps, and sawflies), and Coleoptera (beetles)—all play a major role in improving soil structure and enriching soil chemistry and associated food webs (Coleman and Crossley1996). The cryptozoa group includes millipedes, centipedes, and scorpions. These macrobiota enhance decomposition of organic matter, nutrient cycling, soil structure, and the long-term primary productivity of these ecosystems.

Earthworms (Oligochaeta) are a special class of the macrofauna that has long been recognized as an important component of healthy soil systems (Coleman and Crossley 1996, Lavelle 1988).Their large abundance and biomass in some soils make them a major factor in soil biology (fig. 4.5). Earthworms fall into three general groups: those that dwell in the surface litter, those that are active in the mineral soil layers, and those that move vertically between deeper soil layers and the soil surface. These organisms act as biological agents that decompose litter and mineralize C in both the litter and underlying soil (Zhang and Hendrix 1995). Earthworm digestion increases both the mineralization and humification of organic matter (Lavelle 1988). Earthworms are best known for their beneficial effects of building and conserving soil structure, which results from burrowing and soil ingestion activities (Lavelle 1988). Earthworm activities enhance soil structure by increasing the number of water-stable aggregates, and the creation of burrows and casts increases soil porosity, which improves both aeration and water movement through the soil.

Roots and Reproductive Structures

Many of the plant roots, vegetative reproductive structures, and seeds are found immediately on the surface of the soil or are distributed downward throughout the soil profile. These plant parts include tap roots, fibrous surface roots, rhizomes, stolons, root crowns, and bulbs. The welfare of the plant roots and reproductive structures is important to the sustainability of plant biomass and productivity of all terrestrial ecosystems found throughout the world, particularly in those ecosystems that experience repeated and regular fires. Closely associated with the roots are a suite of symbiotic microorganisms found in the soil (see the previous section Specialized Root-Microbial Associations).

Amphibians, Reptiles, and Small Mammals

Amphibians, reptiles, and small mammals inhabit holes and cavities in the upper part of the soil where they feed on invertebrates, plant parts (seeds), and other organic debris found on or near the soil surface. Burrowing activities and deposition of fecal material by these larger animals contribute to the aeration and fertility of wildland soils.

Biologically Mediated Processes in Soils

The living organisms described above are involved in numerous biological processes that regulate nutrient cycling and contribute to soil productivity and ecosystem health. The nutrient cycling processes for nonfire environments were presented in figure 3.2.

Figure 4.5—Earthworms have long been recognized as an important component of healthy soil systems that function as key biological agents in decomposing litter and mineralizing carbon in both the litter and underlying soil. (Photo courtesy of Earl Stone, University of Florida).

Nutrients enter the soil by precipitation, dry fall, N-fixation, and geochemical weathering of rocks. Nutrients in soil organic matter that accumulates on the soil surface are slowly changed to forms that are available to plants by the biological processes of decomposition and mineralization. The leakage of nutrients from these nonfire ecosystems is usually low, and only small losses occur via volatilization, erosion, leaching, and denitrification. Nutrient cycling in nonfire ecosystems represents a dynamic balance among the processes regulating decomposition, mineralization (ammonification and nitrification), N-fixation and denitrification, and nutrient immobilization and uptake by plants. The long-term sustainability of wildland ecosystems depends on this regular and consistent cycling of nutrients in order to sustain plant growth (DeBano and others 1998).

Decomposition and Mineralization

Organic matter that is deposited on a soil surface is decomposed by soil organisms and incorporated into the underlying soil (mainly as humus). During *decomposition,* C stored in the organic matter is recycled into the atmosphere as CO_2, while N is stored in microbial biomass until it is released during mineralization. Some of the C and N immobilized as microbial tissue can be microbially converted into resistant humic substances (via a process called humification). The humus fraction of soils in natural ecosystems undergoes both continuous decomposition and mineralization, so that the total soil organic matter and N content may remain in a steady state condition until disturbance, such as fire, occurs. Several factors that affect the decomposition of organic matter are the composition of the decomposing litter (substrate quality), environmental factors (moisture and temperature), types of microorganisms involved, and other soil factors including soil pH and deficiencies of inorganic nutrients (P, K, and micronutrients).

An important end product of decomposition is the release of nutrients that can be used by plants. These highly available nutrients are formed during mineralization, which is defined as the conversion of an element from an organic form to an inorganic state as the result of microbial activity (Soil Science Society of America 2001). The mineralization of organic matter involves the transformation of organic N compounds (such as proteins and amino acids) into NH_4 (ammonification) and, subsequently, into nitrite (NO_2) and NO_3 (nitrification); and the conversion of organic C into CO_2 (DeBano and others 1998). Both ammonification and nitrification are affected by fire. Nitrification is carried out mainly by autotrophic microorganisms, which derive their energy solely from the oxidation of NH_4 and NO_3 (Haynes 1986). Several genera of autotrophic bacteria that can oxidize NH_4-N to NO_2 are *Nitrosomonas*, *Nitrosolobus*, and *Nitrospira*. The oxidation of NO_2 to NO_3, however, is almost exclusively done by *Nitrobacter* in natural systems (Haynes 1986). Approximately 30 percent of all the N that is nitrified in natural ecosystems is done by faunal populations (Verhoef and Brussaard 1990).

Nitrogen Cycling Processes

Nitrogen is unique among the soil nutrients because it is present in the soil almost entirely as part of organic compounds. No inorganic N reserve is normally available to replace the soil N lost by volatilization during a fire (Harvey and others 1976). Therefore, N is recovered from the atmosphere by N-fixation and atmospheric deposition (DeBano and others 1998).

Nitrogen-Fixation—*Nitrogen-fixation* is the conversion of molecular N (N_2) to ammonia and, subsequently, to organic combinations or forms utilizable in biological processes (Soil Science Society of America 2001). The atmosphere supplies N to soil in natural ecosystems mainly through organisms that fix inert N_2 into forms that can be used by plants. Nitrogen additions to the soil by N-fixing organisms (free-living and symbiotic) counterbalance the volatilized N that is lost during combustion and subsequent leaching of soluble N compounds into and through the soil following fire (DeBano and others 1998). Nitrogen-fixation is primarily by two groups of microorganisms found in the soil, namely those that fix N symbiotically and those that are free-living.

Symbiotic N-fixation: This form of N-fixation is carried out by symbiotic microorganisms that are associated with the roots of higher plants (symbiotic) and obtain the energy required for fixing N being from the host plant. The most common symbiotic relationships found in wildland ecosystems are those formed by rhizobia or actinomycetes associated with plant roots. *Rhizobium* bacteria are found associated with the roots of leguminous plants that make up about 700 genera in the *Leguminosae* family (Haynes 1986). Locust trees are an example of legumes that enhance the N-status of forest soils (Klemmedson 1994). Nitrogen-fixation by actinomycetes is also widespread in wildland ecosystems, and an example of forest trees having this type of symbiosis is the genus *Alnus*, which has been reported to have enriched the soil with 18 to 129 pounds/acre (20 to156 kg/ha) of N per year depending on the local site conditions and stand density (Van Cleve and others 1971, Jurgensen and others 1979). Brush species having actinomycete-driven N-fixing capabilities are bitterbrush (*Purshia tridentata*) and mountain mahogany (*Cercocarpus* spp.).

Nonsymbiotic N-fixation: Over the past 3 to 4 decades, studies on N-fixation in soils have determined that free-living microbes are also able to fix N in substantial amounts, particularly in forest soils (Jurgensen and others 1997). These free-living microbes obtain the energy necessary for N-fixation during the decomposition of large woody debris (Harvey and others 1989). Coarse woody debris consists of tree limbs, boles, and roots that are greater than 3.0 inches (7.5 cm) in diameter and are in various stages of decay (Graham and others 1994). It is produced when trees die and their boles fall to the soil surface. The death of trees occurs because of old age, insect and disease attacks, devastating natural events, and human activities (timber harvesting debris, as one example). Much of the coarse woody debris found in forests is on the soil surface and is partially or totally covered by soil and humus layers. This coarse woody debris, and associated smaller organic matter, enhances the physical, chemical, and biological properties of the soil and thereby contributes directly to site productivity. In addition, it provides a favorable microenvironment for seedling establishment and growth.

A particularly important role of coarse woody debris is that it serves as a potentially valuable source of nonsymbiotic N-fixation. Extensive studies in the Inland Pacific Northwest show that up to 50 percent of the total N fixed on a site is contributed from the large woody debris component (Jurgensen and others 1997). Soil wood found in these forests is a product of brown-rot decay and is made up of heartwood from pine and Douglas-fir trees. It is relatively resistant to decay and, as a result, can remain in the soil for hundreds of years. The accumulation of coarse woody debris and the fixation of N on warm, dry sites are lower than on the wetter, more productive sites.

Denitrification—*Denitrification* is the reverse process whereby NO_3 is reduced to N_2 and N_2O biologically. It is the major mechanism that returns N, which was originally fixed from the atmosphere, back into the atmosphere (Richardson and Vepraskas 2001). Denitrification typically occurs in saturated and water-logged soils found in wetlands (see chapter 8). Nitrogen losses are generally related to its availability, and significant amounts of N up to 54 pounds/acre/year (60 kg/ha/year) have been reported where NO_3 loadings occur as a result of nonpoint runoff or from other sources.

Fire Effects on Organisms and Biological Processes

Fire affects biological organisms either directly or indirectly. Direct effects are those short-term changes that result when any particular organism is exposed directly to the flames, glowing combustion, hot gases, or is trapped in the soil and other environments where enough heat is transferred into the organism's immediate surroundings to raise the temperature sufficiently to either kill or severely injure the organism. Indirect effects usually involve longer duration changes in the environment that impact the welfare of the biological organisms after the fire has occurred. These indirect effects can involve habitat, food supply, competition, and other more subtle changes that affect the reestablishment and succession of plants and animals following fire.

Soil Microorganisms

Environmental Constraints and Microbial Growth—In this section we provide a brief introduction on the effects of different environmental factors—temperature, moisture, and substrate availability—on microbial life so the reader will understand microbial reactions to fire. A subsequent section then describes the response of different microorganisms to fire.

Temperature: Microbial communities are well adapted to a wide range of prevailing temperature regimes. Drastic changes in temperature during soil heating can result in mortality and shifts in species composition of survivors (Baath and others 1995, Pietikainen and others 2000). Lethal temperatures are as low as 122 °F (50 °C) for some bacteria, particularly for important gram-negative organisms, such as nitrifiers, that have thin cell walls (Wells and others 1979). Above 392 °F (200 °C), virtually all bacteria are killed. Fungi are generally killed at lower temperatures than bacteria, and lethal temperatures range between 122 and 311 °F (50–155 °C) (Wells and others 1979). From a practical standpoint, the threshold temperatures for bacteria and fungi are usually reached to a depth of 2 inches (5 cm) or more in the mineral soil during medium- or high-severity fires (Theodorou and Bowen 1982, Shea 1993, Giardina and others 2000). As a result, decreases in microbial populations due to the direct effect of soil heating are common immediately following either wildfire (Acea and Carballas 1996, Hernandez and others 1997, Prietro-Fernandez and others 1998) or some higher severity prescribed burns (Ahlgren and Ahlgren 1965, Pietikainen and Fritze 1993). Indirect long-term change in soil temperature can also result from modifications to vegetation cover and microclimate following fire. However, compared to the immediate and detrimental effects of soil heating, these longer term temperature changes are subtle and can even produce a slight stimulation of microbial population size and activity (Bissett and Parkinson 1980).

Moisture and oxygen: Microorganisms thrive in moist soil. In fact, most can be considered aquatic organisms by nature of their requirement of aqueous solution for cell movement. Filamentous fungi and actinomycetes are an exception. Optimum moisture content for microbial activity is typically near 55 percent of soil water-holding capacity (Horwath and Paul 1994). A combination of water and oxygen (O_2)content in the soil affects the level of microbial activity. As soils dry, increased cellular energy is required to maintain turgor, resulting in reduced growth and activity. Microbial processes essentially cease when soil becomes air-dry, at which time cells become dormant. Conversely, excessive moisture displaces O_2 and inhibits the metabolic activity of the dominant aerobic microbial population.

Soil moisture is a crucial factor in determining microbial survival during fire. Water is capable of absorbing large amounts of heat energy, thereby resulting in less temperature rise and reduced fire severity for a given heat input. Frandsen and Ryan (1986) found a temperature reduction of more than 932 °F (500 °C) when soil was wetted prior to burning, thus providing a presumed advantage to microorganisms. Conversely, more biological damage can result in moist soil compared to dry soil at a given temperature maximum because water is a better conductor of heat than air, and microorganisms are more metabolically active in moist soil. For example, Dunn and others (1985) estimated that 95 percent of bacteria are killed in moist soil and only 25 percent are killed in dry soil at equivalent soil temperature (158 °F or 70 °C). Burning when soils are dry is recommended if severe soil heating is anticipated. An additional concern is the potential decrease in soil water availability following fire. Less soil water is available after forest fires as a result of decreased water infiltration and storage due to water repellency, and increased water loss by soil surface evaporation. Only one report was found where moisture changes following fire were suspected of altering microbial properties (Raison and others 1986). The reserchers found up to a 34 percent reduction in eucalyptus litter decay after moderate-severity prescribed fire and hypothesized that the change in decomposition resulted from moisture limitations created by increased surface evaporation. This response was suspected to be short-lived, however, because of the rapid reaccumulation of forest floor material in these systems and the anticipated reduction of surface evaporation.

Substrate availability: Most soil microorganisms are heterotrophic, meaning they require preformed, organic material in the forest floor or mineral soil for their source of energy. Therefore, any fire-induced changes in the quality or quantity of organic matter may have long-lasting implications for the biological activity of soil (Lucarotti and others 1978, Palmborg and Nordgren 1993, Pietikainen and others 2000). Surprisingly, most forests have insufficient reserves of degradable organic material to provide for optimal microbial growth. Forest soil microorganisms, in effect, are more likely to be C-limited than by water or other essential nutrients (that is, N, P). As a corollary, the removal of surface organic matter by high-severity fires can reduce microbial population size and activity (Fernandez and others 1997, Prietro-Fernandez and others 1998). Even with fires of low to medium severity, the consumption of downed woody material can influence an important reservoir of mycorrhizal fungi (Harvey and others 1976, 1980a).

In addition to heterotrophs, the microbial community includes a small yet biologically important group of bacteria that obtain their energy from the oxidation of inorganic compounds (autotrophs). Nitrifiers have received the most attention among autotrophic organisms because of their role in the N cycle and their sensitivity to soil heating (Dunn and others 1985). Nitrifiers obtain energy from the oxidation of NO_4 and NO_2, and release NO_3 as an end product.

Response of Soil Microorganisms to Fire—Fire affects most organisms that inhabit the belowground environment in both direct and indirect ways (Ahlgren and Ahlgren 1965, Borchers and Perry 1990). Fire impacts soil organisms directly by killing or injuring the organisms, and indirectly by its effect on plant succession, soil organic matter transformations, and microclimate. Heat penetration into the soil during a fire affects biological organisms located below the soil surface, depending on the heat transfer mechanism, soil moisture content, and duration of combustion.

Because many living organisms and the organic matter in soils are located on, or near, the soil surface, they are exposed to heat radiated by flaming surface fuels and smoldering forest floor fuels (fig. 4.6).

Figure 4.6—High-severity wildfires can remove nearly all the litter and duff and associated microbial populations present on the forest floor, leaving only gray ash. (Photo by Kevin Ryan).

Depending on fire severity, organisms in the organic forest floor can be killed outright, although those in the deeper soil horizons, or in isolated unburned locations, survive. Even low-severity fires can damage organisms that are on or near the soil surface because most biota are damaged at lower temperatures than those that cause changes in physical and chemical soil properties during fire (table 4.1).

How do microorganisms respond to fire? Without question, fire is lethal. It also modifies the habitat of microorganisms by destroying organic matter, altering soil temperature and moisture regimes, and changing the postfire vegetation community and rates of organic matter accumulation. Consequently, changes in microbial population size and activity are common following wildfire and prescribed fire (see Ahlgren 1974, Raison 1979, Borchers and Perry 1990, Neary and others 1999 for reviews). Most reviews are quick to point out, however, that microbial responses are variable, or even unpredictable, depending on site conditions, fire intensity and severity, and sampling protocol. But if microbial responses are unpredictable, then no assistance can be offered to managers in developing fire prescriptions that meet operational objectives while minimizing risks to soil biota. The following discussion will challenge this concept in attempting to offer practical guidelines for fire managers.

If microbial life was only a function of temperature, moisture, and substrate availability, then the refinement of fire prescription guidelines based on soil biotic responses would be straightforward. Unfortunately, it is not that simple because the ability of most microorganisms to recover from disturbance is complex. Microbial communities are unmatched in physiological diversity and genetic malleability—properties that permit growth in any environment—and they have the ability to degrade nearly all known compounds. Resilience, therefore, is a trademark of the microbial community. As an example, rapid declines in soil microbial populations due to fire are usually transitory. Population sizes often match or surpass preburn levels within a growing season (Ahlgren and Ahlgren 1965, Renbuss and others 1973). Successful recolonization following fire is a function of several factors, including incomplete mortality of native populations, spore germination, influx of wind-blown organisms, and microbial growth stimulations from available nutrients. An important caveat is that not all microorganisms respond alike; differential responses by community members have been observed following fire. For example, Harvey and others (1980a) found poor recolonization of ectomycorrhizae, and Widden and Parkinson (1975) found a similar response for genera *Trichoderma*, an important antagonist to plant pathogens, following slash burning.

Some obvious observations can be made. Microbial responses to fire are easiest to predict at the opposing ends of the fire-severity continuum: (1) intense wildfire can have severe and sometimes long-lasting effects on microbial population size, diversity, and function; (2) low-severity underburning generally has an inconsequential effect on microorganisms. From there it becomes more difficult to predict microbial adaptation because results from medium- to high-severity slash fire and underburn studies vary widely among habitat types.

Wildfire: High-severity wildfires can remove nearly all the litter and duff and associated microbial populations present on the forest floor, leaving only gray or orange ash (fig. 4.6). Even mineral soil C is consumed during wildfires. Recent studies show a range of 5 to 60 percent loss of organic material in the surface mineral soil (Fernandez and others 1997, Hernandez and others 1997, Prietro-Fernandez and others 1998). Related declines in microbial biomass immediately following high-severity fires have been as high as

Table 4.1—Threshold temperatures for key biological organisms (Adapted from DeBano 1991, Neary and others 1999).

Biological component	Temperature		Reference
	°F	*°C*	
Plant roots	118	48	Hare 1961
Small mammals	120	49	Lyon and others 1978
Protein coagulation	140	60	Precht and others 1973
Fungi—wet soil	140	60	Dunn and others 1975
Seeds—wet soil	158	70	Martin and others 1975
Fungi—dry soil	176	80	Dunn and others 1975
Nitrosomonas spp. bacteria—wet soil	176	80	Dunn and DeBano 1977
Nitrosomonas spp. bacteria—dry soil	194	90	Dunn and DeBano 1977
Seeds—dry soil	194	90	Martin and others 1975
VA mycorrhizae	201	94	Klopatek and others 1988

96 percent (Hernandez and others 1997). Microbial respiration and extracellular enzyme production (Saa and others 1998)—strong indicators of microbial activity and viability—also decline dramatically following wildfire. Fungi appear more sensitive to wildfire than bacteria. For example, fungal propagules were undetectable 1 week after a stand-replacing fire in a central Oregon ponderosa pine (*Pinus ponderosa*) forest, while the viable bacteria population was only slightly reduced (D. Shields, personal communication). Similar responses were reported following wildfire in a pine forest of Spain (Vazquez and others 1993). Differences in response between bacteria and fungi may be attributable to greater heat sensitivity of fungi, soil pH increases after burning that favor bacteria, or the excessive loss of organic material.

No ecosystem remains sterile, even after severe disturbance. Most studies show stable recovery of microbial populations in the mineral soil to prefire levels within 1 to 4 years after wildfire (Vazquez and others 1993, Acea and Carballas 1996, Prietro-Fernandez and others 1998). However, reduced microbial biomass has been reported for as many as 11 years after fire (Dumontet and others 1996). Whether the microbial community will fully recolonize depends on the time required for recovering the forest floor layer. Not only are microbial populations and processes suppressed during this recovery period (Lucarotti and others 1978), but the potential erosive loss of soil is high. Or, as suggested by Giovannini and others (1987), burnt soils will slowly regenerate as long as erosive processes can be avoided.

Low-severity prescribed fire: Almost by definition, low-severity prescribed fire has a minimal effect on soil biota (fig. 4.7). The maximum temperatures are generally nonlethal, except for the upper litter layer (Shea 1993), and therefore the consumption of forest floor habitat is limited. Changes in microbial activity, in fact, often show a positive response to this type of fire, particularly with respect to N-fixation (Jorgensen and Wells 1971) and N availability (Schoch and Binkley 1986, White 1986, Knoepp and Swank 1993a,b). Rates of litter decay (White 1986, Monleon and Cromack 1996) and enzyme activity (Boerner and others 2000) are generally unaffected by low-severity underburning. Such results are not universal, however. Monleon and others (1997) found that N mineralization was reduced at sites burned either 5 or 12 years earlier by low- to medium-severity prescribed fire. They suggested that fire-induced changes in N mineralization possibly contributed to a decline in the long-term site productivity of ponderosa pine stands in central Oregon.

Figure 4.7—Low-severity prescribed fire has a minimal effect on soil biota because maximum temperatures are generally nonlethal, except for the upper litter layer, and consumption of forest floor habitat is limited. (Photo by Daniel Neary).

While single-entry underburning is generally considered harmless, repeated burning has been shown to substantially reduce microbial population size and activity (Jorgensen and Hodges 1970, Bell and Binkley 1989, Tongway and Hodgkinson 1992, Eivazi and Bayan 1996). This observation reflects a cumulative reduction in forest floor and total nutrients with frequent burning. Most studies have compared either annual burning or short-term repeated fires (2 to 4 years). The long-term impact of repeated burning every 7 to 20 or more years on soil organic matter, nutrient content, and microbial processes is not understood. As a consequence, Tiedemann and others (2000) urge caution in the use of frequent fire and suggest including partial harvesting as a complementary practice to reduce wildfire risk and extend the period between prescribed burning.

Slash burning: The effect of slash burning will depend on both the pattern and amounts of fuels burned. When slash is piled and burned (for example, pushing and burning operations used in some pinyon-juniper eradication programs) the burned areas are highly visible after the fire (fig. 4.8). On these burned areas, deep layers of white ash may accumulate, and the underlying soil is usually exposed to extended soil heating, which can sterilize to a depth of several centimeters. Although the severe heating under the piles of fuel are damaging to the soil, only a small percentage of the total area may be affected. To avoid this damage to the soil, the slash can be scattered and burned, thereby minimizing the severe soil heating that occurs under piled fuels.

The literature contains numerous references to site-specific responses of microorganisms to medium and high-severity slash burning. A common theme is that the response of microorganisms is dictated by their habitat: organisms in the forest floor struggle to survive and recolonize, while those in the mineral soil do

Figure 4.8—Piling and burning slash after a fuel harvesting operation in pinyon-juniper woodlands can create thick layers of white ash and extensive soil heating. (Photo by Malchus Baker).

not. This comes as little surprise because the temperature in the forest floor can easily reach 1,110 °F (600 °C) or higher during burning (Renbuss and others 1973, Shea 1993), consuming both the microorganisms and their habitat. In fact, substantial changes in forest floor microbial biomass and community structure have been reported to occur during a soil heating experiment at a temperature of 445 °F (230 °C) (Pietikainen and others 2000). Soil heating was less severe in the mineral soil, resulting in a much shorter fluctuation period before the microbial community stabilizes. The following examples illustrate this theme.

Recent findings from slash burns in Finland show detrimental effects of slash burning on microbial biomass (Pietikainen and Fritze 1993), activity (Fritze and others 1994, 1998), and community structure (Baath and others 1995, Pietikainen and others 2000) in the forest floor. Failure of the microbial community to respond rapidly has been attributed to a decline in organic matter quantity (Baath and others 1995) and quality (Fritze and others 1993), and the pyrolytic production of toxic compounds (Fritze and others 1998). Also, microbial populations were unable to respond to the input of nutrient-rich ash. These studies were relatively short term, ranging from 1 month to 3 years postfire. As a result, the length of time required before the forest floor microbial community reaches preburn levels is unclear, yet it might take up to 12 years (Fritze and others 1993). Related declines in microbial function have been reported in other forest types. Staddon and others (1998) found microbial-mediated enzyme activity was suppressed 4 years after slash burning in jack pine (*P. banksiana*). Meanwhile, Jurgensen and others (1992) found a 26 percent decline in N-fixation by free-living bacteria during the first 2 years following a relatively high-severity slash fire that consumed 61 percent of the forest floor in a cedar-hemlock forest.

In contrast with the forest floor, microbial recovery in mineral soil following intense slash burning is impressive. For example, Renbuss and others (1973) examined viable bacterial and fungal populations after a high-severity log pile burn that produced temperatures of 735 to 1,110 °F (400–600 °C) in the upper 2 inches (5 cm) of soil. Although the soil was initially sterilized by fire, bacteria had recolonized to preburn levels within 1 week after ignition. Their population size remained at or above the level of the control soil for the length of the study (1 year). Fungi and actinomycetes were slower to recolonize, yet their populations returned to prefire level by the end of the study. Chambers and Attiwill (1994) confirmed the "ash-bed" effect in a controlled soil heating experiment. No differences in microbial population sizes were found 133 days after heating soil to 1,110 °F (600 °C) when compared to unheated soil. Similar responses are common in field studies (Ahlgren and Ahlgren 1965, Theodorou and Bowen 1982, Deka and Mishra 1983, Van Reenen and others 1992, Staddon and others 1998), whereas some controlled soil heating experiments have found longer delays in recolonization (Dunn and others 1979, Diaz-Ravina and others 1996, Acea and Carballas 1999). Differences between field and controlled-environment studies suggest the importance of wind- or animal-transported inoculum for recolonization.

Specialized Root-Microbial Associations—Mycorrhizal fungi are easily affected by fire, and the extent of damage depends upon fire severity, the reproductive structures exposed to soil heating (such as spores or hyphae), and the type of fungi (such as endo- or ectomycorrhiaze). Mycorrhizae and roots frequently occupy the uppermost duff layers of soil and as a result are subjected to lethal soil temperatures during a fire because these layers are frequently combusted, particularly during medium- and high-severity fires. In general, vesicular arbuscular mycorrhizae are less affected by disturbances that destroy aerial biomass (including fire) than are ectomycorrhizal fungi because they form symbiotic relationships with a wider range of plant species (Puppi and Tartaglini 1991). Also, ectomycorrhizae are more abundant in the litter layer compared to vesicular arbuscular mycorrhizae, which tend to concentrate in the lower mineral soil horizons (Reddell and Malajczuk 1984). As a result, fire that destroys only the litter layer would favor vesicular arbuscular mycorrhizae.

The general relationships discussed above have been documented by several studies that show a decline in the formation of both ectomycorrhizae (Harvey and others 1980b, Schoenberger and Perry 1982, Parke and others 1984) and vesicular arbuscular mycorrhizae (Klopatek and others 1988, Vilariño and Arines 1991) within the first growing season following fire. Mycorrhizae were not per se eliminated by fire in these studies. Instead, the percentage of roots infected by mycorrhizal fungi was reduced. As an example, Klopatek and others (1988) found vesicular arbuscular mycorrhizae infection declined from 41 percent at preburn to 22 percent within 24 hours of burning the organic layer of a pinyon pine (*P. edulis* Engelm.) soil. Work by Harvey and others (1976, 1980a,b, 1981) has also clearly established a relationship between reduced ectomycorrhizal root tip formation and slash burning on difficult-to-regenerate sites in western Montana. They emphasize the importance of maintaining adequate soil humus and wood (up to 45 percent by volume) as refugia for mycorrhizae on these sites.

Not all studies, however, have reported a detrimental relationship between mycorrhizal formation and fire. Several studies have shown no effect of burning on mycorrhizae by the end of the first growing season (Pilz and Perry 1984, Deka and others 1990, Bellgard and others 1994, Miller and others 1998). In fact, Herr and others (1994) found a slight increase in ectomycorrhizal infection on eastern white pine (*P. strobus*) with increasing fire severity. Visser (1995) examined ectomycorrhizal development in an age sequence of jack pine stands regenerating following fire. More than 90 percent of the root tips were mycorrhizal regardless of time since burning (from 6 to 122 years). She suggested that successful recolonization of mycorrhizae on jack pine seedlings was a function of (1) avoidance of lethal temperatures by location in soil profile, (2) resistance of spores and resting structures to lethal temperatures, (3) wind or animal dispersal of spores, and (4) survival on alternative plant hosts such as manzanita (*Arctostaphylos* spp.).

Finally, there is no clear evidence that fire impairs the function of mycorrhizae in plant nutrition and growth. Studies showing a decline in mycorrhizal infection after fire have seen minor or no decline in seedling survival or short-term growth (Schoenberger and Perry 1982, Parke and others 1984). Two simplified explanations are plausible: (1) either the flush of nutrients after fire makes mycorrhizae temporarily superfluous, or (2) the decline in root infection (such as a decline from 40 to 20 percent root tip colonization following burning) has no relationship to function. The argument in the second explanation is that mycorrhizal function is still effective whether root systems are completely or partially infected. Intermediate- or long-term studies are needed to resolve this issue.

Biological Crusts—The patchy nature of native plant communities in arid and semiarid lands produces a discontinuous source of fuels and results in a mosaic of fire intensities (Whisenant 1990). The biological crusts themselves provide little fuel to carry fire and thereby provide a "refugia" that slows down the spread of fire and minimizes its severity (Rosentreter 1986).

However, once fire destroys cryptogamic crusts, it can take several years for their populations to redevelop to prefire levels (DeBano and others 1998). High-severity fires during the dry summer months cause the greatest damage to biological crusts. The recovery following fire can be fast or slow, depending upon the type of crust (algal or lichenous), soil conditions, and climate. For example, crusts can take much longer to develop in hotter and more arid shrublands such as occurred on a site in a blackbrush (*Coleogyne ramosissima*) community in southern Utah (Callison and others 1985). This site did not show any cryptogamic development following a severe range fire after 37 years. Likewise, annual fires for 7 years have been reported to destroy cryptogamic crusts on degraded semiarid woodlands in Australia (Greene and others 1990).

Low-severity fires, in contrast, may leave the structural matrix of the crust intact (Johansen and others 1993). For example, after a single fire in a semiarid shrub-steppe it only took 4 years for the crusts to reach prefire levels. This study showed that the components making up the cryptogamic crust (such as algae, cyanobacteria, or lichens) affect the rate of recovery following disturbance. Algae recovered from disturbance most rapidly and returned to prefire densities within 1 to 5 years. Historically, fires in the Southwestern deserts probably were exposed to small, low-intensity, and patchy fires because of the sparse and discontinuous vegetation (Allen 1998, Belnap and others 2001). This type of fire behavior most likely had a minimum effect on the biological crusts common to this area.

In general, algal cells of many species are usually able to survive even the most severe disturbance, so the dispersal into and recolonization of burned areas is faster (Johansen 1993). For example, the first organisms to recolonize the soil under burned English heaths were algae (Warcup 1981). Algae are also favored by the higher pH after fire, and species with windblown propagules are able to recolonize disturbed areas rapidly (DeBano and others 1998). In contrast, filamentous cyanophytes (blue-green bacteria) are less likely to recolonize by wind dispersal because of their size. Compared to algae, mosses and lichens are slower to reestablish themselves. Acrocarpic mosses were found to reoccupy burned sites in the Mediterranean area within 9 to 15 months after fire (De Las Heras and others 1993). Lichens that produce vegetative diaspores can move into disturbed areas more quickly than lichens that do not produce these propagules (Johansen and others 1984). Some lichens can take 10 to 20 years to develop diverse and abundant communities (Anderson and others 1982).

Soil Meso- and Macrofauna

Most research results on the effect of fire on meso- and macrofauna are reported in terms of general groups of soil invertebrates, including insects and other arthropod assemblages, and earthworms. Insects, for example, may have representatives in both the meso- and macrofauna groups (fig. 4.1; also see earlier discussion in this chapter on soil meso- and macrofauna). Another important ramification of fire and these organisms is the postfire infestation of forests and other ecosystems by insects as a result of the effect of fire on the health of the postfire vegetation. The response of all the above organisms depends to a large degree on the frequency and severity of fire.

The magnitude of short-term changes undergone by invertebrate populations in response to fire depends on both fire severity and frequency, the location of these organisms at the time of the fire, and the species subjected to fire. In the case of either an uncontrolled wildfire (high fire severity), and prescribed fire (low fire severity), the effect on invertebrates can be transitory or longer lasting (Lyon and others 1978). Both types of fires contain zones of high and low fire severity, but wildfires are more likely to burn larger, contiguous areas. The long-term abundance of arthropod populations, however, can remain high because of their resiliency to both intensity and frequency of burning (Andersen and Müller 2000).

Some studies show the effect of different severities and frequencies of fire on invertebrates. One study in *P. sylvestris* forests in Sweden showed that the overall mortality of invertebrates depended on the proportion of organic soil consumed by the fire and that the mortality ranged from 59 to 100 percent. Invertebrates that lived deeper in the soil had less mortality than those that colonized the vegetation and litter layers (Wikars and Schimmel 2001). Other characteristics that favored survival included greater mobility in the soil and thick protective cuticles (as is found in the taxa Oribatediae and Elateridae).

A study of the response of insects and other arthropods to prescribed burn frequency in prairie ecosystems showed that the changes in the physical environment and plant communities following prescribed fires can result in the development of distinctly different arthropod communities on the frequently burned sites compared to sites that were protected from burning (Reed 1997). Distinctive arthropod species and groups were supported by the changing succession stages following fire. In general, landscapes that have a range of sites representing different successional stages and sites that have different burn frequencies support the most species. However, on individual sites that are burned at intervals, a cycle of arthropod species richness, species composition, and numbers of individuals occur. The combined effect of fire frequency and time of burn on arthropod taxa were reported for tropical savannas found in Australia (Andersen and Muller 2000). A substantial resilience to fire of the arthropod assemblages was found. Only four of the 11 arthropod taxa were significantly affected by fire. Ants, crickets, and beetles declined in the absence of fire. Late season fires decreased spiders, homopterans, silverfish, and caterpillars.

Some invertebrates have traits that allow them to survive fire. These traits may arise in a variety of ways. Some may not have evolved specifically as an adaptation to fire, but rather more generally to hot and arid conditions. For example, some invertebrates have adaptations that enable them to conserve water and resist high ambient temperatures in seasonally dry habitats. Other groups of invertebrates possess traits, such as high mobility, that appear to be characteristic to particular taxonomic groups and not related to specific ecosystems or fire regimes. Still other adaptations appear to have evolved primarily in response to fire and can involve the complex long-term evolution of some rather esoteric anatomical features. For example, a recent study on detailed morphological and anatomical characteristics of a subfamily of beetles (Coleoptera: Clerinae) suggests that these invertebrates evolved thermoreceptor antenna, which enable the beetles to avoid death by fire in xeric environments (Opitz 2003). The evolution of this feature occurred over a span of tens of thousands of years.

The effects of fire on soil invertebrates have been reviewed by several authors (DeBano and others 1998, Lyon and others 2000a 2000b, Andersen and Müller 2000). Many of the reports cited describe invertebrates in savannas and other grasslands. These reports indicate that the effects of fire on soil invertebrates can occur via several mechanisms (Andersen and Müller 2000) that include direct mortality, through forced emigration, or through the immigration of pyrophilous species (such as wood-boring beetles that are attracted by heat and smoke to a burned area where they infest injured or dead trees). Short-term indirect effects include modification of the habitat and foraging sites, food supplies, microclimate, and rates of predation. Long-term indirect effects are manifest mainly in nutrient cycling and primary productivity.

Invertebrates residing more permanently in the upper soil layers are most likely affected when these soil layers are heated to lethal temperatures. Macroinvertebrates dwelling exclusively in litter were found to be particularly vulnerable to wildfire that destroys surface fuels and litter (Sgardelis and others 1995). However, invertebrates that permanently occupy deeper soil horizons are usually protected from even high-severity fires. Some macroinvetebrates have been found to move deeper into the soil during the summer and, as a result, they are insulated from lethal soil temperatures during fire. Most of the invertebrates in the top 1.0 inch (2.5 cm) of soil survive relatively cool burning wildfires or prescribed fires (Coults 1945). A reduction in litter quantity after fire can indirectly decrease both the number of invertebrate species and the species density of soil and litter invertebrates as unprotected mineral soil warms (Springett 1976).

The effect of soil heating on earthworms is not well understood. One study in tallgrass prairie, however, showed that the indirect effects of a fire are probably more important than direct heating of earthworm populations (James 1982). This study showed that fire increased earthworm activity because of differences in plant productivity following the fire. In general, the subsurface soil horizons are usually proliferated with roots and rhizomes, which in combination with more favorable soil moisture conditions create an ideal environment for earthworms at about 4 to 8 inches (10 to 20 cm) below the soil surface. A location this deep in the soil most likely protects earthworms from the direct effects of soil heating during fuel combustion, except in the case of severe long-duration fires that might occur under piles of slash and logs or in smoldering duff and roots. Other studies have shown that fire (in prairie grasslands and mixed forest types) frequently leads to an increase in exotic earthworm species at the expense of endemic species (Bhadauria and others 2000, Callaham and others 2003).

Roots and Reproductive Structures

Many of the plant roots, vegetative reproductive structures, and seeds are found immediately on the surface of the soil or distributed downward throughout the soil profile. These plant parts include tap roots, fibrous surface roots, rhizomes, stolons, root crowns, and bulbs. Many plant roots are found in the surface organic layers (L-, F-, and H-layers) and can be directly affected whenever these layers are heated or destroyed during a fire. Plant roots are sensitive to both duration of heating and the magnitude of the temperature reached. Temperatures of 140 °F (60 °C) for 1 minute are sufficient to coagulate protein (Precht and others 1973). Plant roots are sensitive to soil heating, and lethal temperatures can occur before proteins began to coagulate. The lethal temperature of plant tissue is highly dependent on the moisture content of the tissue. Those tissues containing higher moisture contents tend to be killed at lower temperatures and during a shorter interval of heating (Zwolinski 1990). Miller (2000) gives additional information on the effects of fire and soil heating on root mortality and the welfare of vegetative reproduction.

Plant roots that are insulated by the soil have a lower risk of being subjected to lethal temperatures during a fire (DeBano and others 1998). The two most important factors that insulate roots against soil heating are their depth in the soil and the soil water content. Generally, the deeper the plant roots are located in the soil, the greater will be the survival rate (Flinn and Wein 1977). Low-severity fire that destroys only the plant litter may kill only aboveground plant parts. In contrast, high-severity fires can consume all the surface organic matter and easily heat the mineral soil above the lethal temperature for roots.

Seed banks that are stored in the soil can be affected by fire. A majority of the seeds are stored in the litter and upper part of the soil beneath the vegetative canopy. Medium to high-severity fires heat the surface layers sufficiently to destroy any seeds that have been deposited. The lethal temperature for seeds is about 160 °F (70 °C) in wet soils and 190 °F (90 °C) in dry soils (Martin and others 1975). Although fire can destroy seeds, it also can enhance reproduction by seeds (Miller 2000). For example, fire can destroy allelopathic substances that inhibit seed production. Or, in the case of ponderosa pine regeneration, fire can provide a mineral seedbed required for germination and growth. The heating associated with fires may also stimulate the germination of seeds that lie dormant in the soil for years because of impermeable seed coats (such as seeds of chamise, hoaryleaf ceanothus).

Soil heating, heat transfer, and the effect of the lethal temperatures on the welfare of seeds and roots are more complicated in moist soils than when the soil is dry. Dry soil is a poor conductor of heat and, as a result, heat does not penetrate deeply in the soil, particularly if the residence time of the flaming front is short. The surface of dry soil can easily exceed the lethal temperature of living tissue of roots, while ambient daily soil temperatures can prevail 0.8 inch (2 cm) downward in the soil, with little damage occurring to the roots. Therefore, when the roots are in dry soil below 0.8 inch (2 cm), they are not likely to be damaged by soil heating unless the residence time of the flaming front is long. Conversely, those plant structures on or near the soil surface can easily be damaged. Also, the presence of moisture in the soil affects plant root and seed mortality (in other words, living biomass is killed at lower temperatures when the soil is wet compared to a dry soil).

Amphibians, Reptiles, and Small Mammals

The ability of amphibians, reptiles, and small mammals to survive wildland fires depends on their mobility and the uniformity, severity, size, and duration of any fire (Wright and Bailey 1982). Fire can cause direct injury and kill the animals themselves depending on how capable they are of avoiding and escaping the fire itself (Lyon and others 2000a). The effects of wildland fires on these animal populations can be found in a separate volume of this series (Smith 2000). Fire also affects the long-term welfare of these larger animals by changing their habitat (Lyon and others 2000b).

Biologically Mediated Processes

A wide range of microorganisms participate in cycling carbon and plant nutrients in a systematic and sustainable rate that is necessary for maintaining healthy ecosystems. Important biologically regulated processes carried out by these microbes that can be affected by fire include decomposition, mineralization, ammonification, nitrification, nitrogen-fixation, and denitrification.

Decomposition—Fire affects decomposition in two general ways (DeBano and others 1998). First, moderate- and high-severity fires kill the biological organisms that decompose organic matter (see the earlier discussions on microorganisms). The microorganisms most affected by fire include bacteria and fungi, which are numerically the most abundant organisms in terrestrial ecosystems and are the primary decomposers of organic matter in soil (Van Veen and Kuikman 1990). Second, a rapid, strictly chemical combustion process replaces the slower, biologically mediated decomposition processes that occur under nonfire conditions.

Nitrogen Mineralization—Two important mineralization process affected by fire are ammonification and nitrification. The sensitivity of both ammonifying and nitrifying bacteria to soil heating most likely plays an important role in the nutrition of plants because N is frequently limiting in wildland soils (DeBano and others 1998). This relationship has been demonstrated in unburned chaparral stands where high levels of total N can occur as organic N, but only relatively low levels of inorganic mineral N (NH_4- and NO_3-N) have been measured. It has been hypothesized that the low rate of mineralization in chaparral soils occurred because heterotrophic microorganisms responsible for mineralization were inhibited by allelopathic substances present, or because high lignin contents of chaparral plant leaves resist decomposition and subsequent mineralization of N (Christensen 1973). The hypothesis that higher concentrations of NH_4- and NO_3-N are generally present after a fire is based on the idea that NH_4- and NO_3-N are formed by different processes as a result of burning (Christensen and Mueller 1975, DeBano and others 1979). According to this hypothesis, relatively large amounts of NH_4-N are produced chemically by soil heating during a fire (see chapter 3) as well as being microbially produced following a fire. Nitrogen in the form of NO_3, however, is not produced directly by heating during a fire, but instead is formed during subsequent nitrification of excess NH_4-N produced as a result of burning. This process is further complicated by the observation that postfire nitrification does not appear to be carried out by the classical nitrifying bacteria (for example, *Nitrosomonas* and *Nitrobacter*) because these bacteria are particularly sensitive to soil heating and other disturbances and, as a result, are absent (or at extremely low levels) for several months following burning (Dunn and others 1979). The absence of nitrifying bacteria after fire suggests that nitrification may be carried out by heterotrophic fungi. Dormant forms of heterotrophic fungi have been reported to be stimulated by mild heat treatments (Dunn and others 1985) and are thought to have contributed to the fungal growth that paralleled NO_3 production (Dunn and others 1979). Suppression of mineralization rates by allelopathic substances has been further substantiated by other studies in ponderosa pine forests in New Mexico (White 1991, 1986) and in pinyon-juniper woodlands (Everett and others. 1995).

Studies have shown that both *Nitrosomonas* and *Nitrobacter* are sensitive to soil heating during fire such as occurs with microorganisms described earlier (table 4.1). Studies in chaparral soils have shown that *Nitrosomonas* bacteria are killed in dry soil at temperatures of 250 to 280 °F (120 to 140 °C) as contrasted to a moist soil where the lethal temperature is between 165 and 175 °F (75 and 80 °C) (Dunn and DeBano 1977, Dunn and others 1985). *Nitrobacter* bacteria are even more sensitive and are killed at 212 °F (100 °C) in dry soil and at 120 °F (50 °C) in a moist soil. Unlike heterotrophs, which must adapt to decreased organic matter availability, nitrifiers are provided with a sharp increase in available substrate ($NH_{4)}$ after fire (Raison 1979). Consequently, the initial nitrifying bacterial population decline due to soil heating typically is reversed within the first year in response to the "flush" of available substrate (Jurgensen and others 1981, Acea and Carballas 1996).

Unfortunately, the increases in NH_4- and NO_3-N following a fire are relatively short-lived and can return to prefire levels within 2 years after burning (DeBano and others 1998). Studies conducted after burning tropical forests in Costa Rica showed that the increase in available N returned to background levels in 6 months (Matson and others 1987). Likewise, the increased NH_4-N produced during a fire in an Arizona chaparral ecosystem remained at an elevated level for about 6 months and then began decreasing, at which time NO_3-N began increasing (see fig. 3.6). Within a year, both NH_4- and NO_3-N levels returned to prefire levels. In another study, N mineralization increased and remained elevated for 1 year in *Agropyron spicatum* and *Stipa comata* grasslands for 2 years following prescribed burning in mountain shrublands (Hobbs and Schimel 1984).

Nitrogen-Fixation—The effect of fire on N-fixation involves the effect of heating on the living protoplasm present in symbiotic and nonsymbiotic microorganisms that fix N. Symbiotic N-fixing microorganisms can be affected in at least two ways. First, the destruction of the host plant during combustion affects the symbiotic relationship by removing the source of energy. Second, the symbiotic microorganisms present in the roots may be killed if the upper organic layers are consumed during a high-severity fire. Conversely, little or no direct damage would be expected in the case of deeper roots, which are far removed from the soil surface, or during a cooler burning prescribed fire (DeBano and others 1998). Nonsymbiotic bacteria respond similarly to other microbes discussed earlier.

On the other hand, nonsymbiotic processes, which receive energy from the biological oxidation of organic matter, also have been reported to fix substantial amounts of N. For example, more than one-third of the N-fixing capacity of forest soils has been reported to be provided by microorganisms responsible for decaying wood on the surface and in the soil profile (Harvey and others 1989). The management of woody residues (coarse woody debris) within a fire prescription thus becomes an important consideration in forest management. Therefore, it is important to retain a substantial amount of large woody debris on forest sites after timber harvesting or when using prescribed fire. For example, the amounts of residual woody debris recommended in figure 4.9 are considered necessary for maintaining the productivity of forests in Arizona, Idaho, and Montana.

The indirect effect of fire on N-fixation focuses on the role that both symbiotic and nonsymbiotic organisms play in N-fixation and replenishment in wildland ecoystems (DeBano and others 1998). Currently this role is still under debate. For example, the results reported from a study in forests of the Northwestern United States showed that soil N-additions by symbiotic N-fixation were not as large as previously assumed for these ecosystems (Harvey and others 1989). However, some vegetation types, such as alder trees in riparian ecosystems and dense stands of snowbrush (*Ceanothus velutinus*), have been reported to fix substantial amounts of N (Jurgensen and others 1979). The fixation of N in these forest and brushland areas is by symbiosis.

Burning may create a favorable environment for the establishment of N-fixing plants in some plant communities that are subjected to frequent fire. For example, fire exclusion in ponderosa pine-Douglas fir forests in the Northwest has been reported to lead to such widespread changes in forest structure, composition, and functioning that N-fixing plants species have been reduced (Newland and DeLuca 2000).

Denitrification—Little research has been done on the biological losses of N in relation to fire, partly because of the overwhelming losses that occur chemically by volatilization (DeBano and others 1998). Denitrification, however, can be an important factor when using fire in wetlands (see chapter 8).

Management Implications

Rarely are microorganisms and their processes more than a passing thought in forest management plans. For example, fuel-reduction programs rarely consider the potential effects of surface fire on soil organisms such as mycorrhizal fungi or autotrophic bacteria (W. Johnson, personal communication). A recent survey and management policy developed for protecting a small percentage of the fungi and lichen within the critical range of the northern spotted owl is the exception to the rule (Molina and others 1999). This situation is not surprising for at least two reasons. First, most microorganisms are invisible to the naked eye and are thus "out of sight" and, as a result, "out of mind." Second, no simple, inexpensive field test is available to measure microbial populations or their processes. Thus, managers have no practical means of determining microbial responses to operational prescribed burns. In addition, the results published from fire effects studies have not always presented a clear picture of how microorganisms respond to fire, thus leaving managers guessing at how responsive microorganisms are (or wondering whether they should care) for given forest types and anticipated fire severities.

However, the following general concepts of microbial responsiveness to fire can be gleaned from past studies:

- Microorganisms are skilled at recolonizing disturbed forests. Their resiliency is a function of unsurpassed physiological and genetic diversity.
- Fire effects are greatest in the forest floor and decline rapidly with mineral soil depth. Recovery of microbial populations in the forest floor is not guaranteed, particularly in dry systems with slow reaccumulation of organic material.
- Severe wildfire and prescribed fire reduce organic matter content and increase the potential for loss of soil by erosion.
- Prescriptions that avoid drastic changes in the environmental factors controlling microbial life—soil temperature, moisture, and substrate availability—will be the most successful at meeting operational objectives while ensuring a functioning soil biotic community.

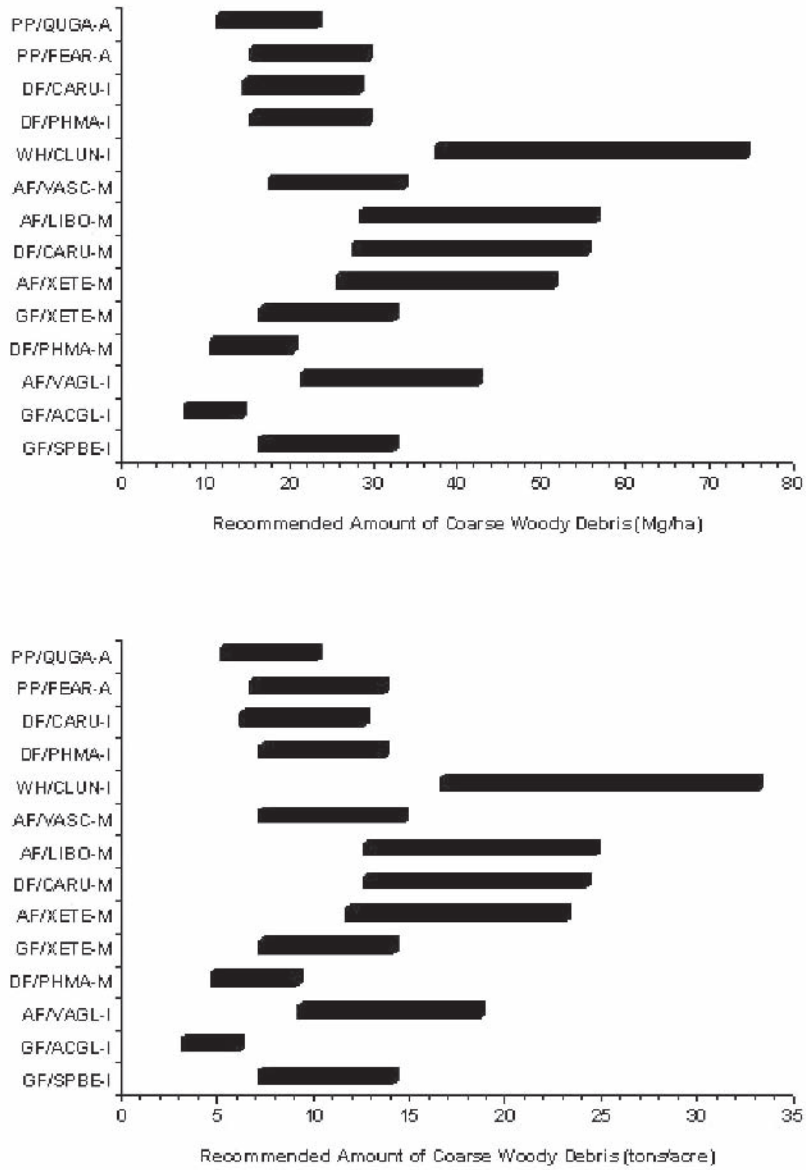

Figure 4.9—Amounts of residual coarse woody debris to leave after timber harvesting necessary to maintain site productivity in Arizona, Idaho, and Montana forests. Accompanying chart lists habitat type acronyms. (Adapted from Graham and others 1994).

State	Acronym	Habitat type
Idaho	GF/SPBE-I	Grand fir / snowberry
	GF/ACGL-I	Grand fir / mountain maple
	AF/VAGL-I	Subalpine fir / huckleberry
	WH/CLUN-I	Western hemlock / queencup beadlily
	DF/PHMA-I	Douglas-fir / ninebark
	DF/CARU-I	Douglas-fir / pinegrass
Montana	DF/PHMA-M	Douglas-fir / ninebark
	GF/XETE-M	Grand fir / bear grass
	AF/XETE-M	Subalpine fir / beargrass
	DF/CARU-M	Douglas-fir / pinegrass
	AF/LIBO-M	Subalpine fir / twintower
	AF/VASC-M	Subalpine fir / whortleberry
Arizona	PP/FEAR-A	Ponderosa pine / Arizona fescue
	PP/QUGA-A	Ponderosa pine / Gambel oak

- Knowledge gaps persist, particularly regarding repeated fire and its effect on microorganisms.

Based on the above concepts, the following recommendations are offered:

- Minimize loss of forest floor (litter and duff). Microorganisms are most vulnerable to heat damage and habitat changes in this layer. This presents a quandary for prescribed fire practioners: How much organic material (fuel) should be removed to reduce wildfire danger while still maintaining an adequate supply for forest function? Tiedemann and others (2000) recommend burning when the upper layer of the forest floor is dry enough to carry fire and the lower layers are wet enough to avoid consumption. Further, they recommend extending the recovery time between repeated fires if these conditions are not achieved or if exposure of mineral soil is desired for tree regeneration.
- Avoid burning when soil is moist *if* the anticipated fire severity is high. Mortality of microorganisms is greater in moist soil than in dry soil at high temperatures.
- Provide adequate inoculum for microbial recolonization by burning with mosaic patterns (there is no assurance that indigenous populations will survive soil heating).
- Supplement burning with other silvicultural practices (partial harvest, crushing, mulching) to reduce fuel buildup. Repeated burning of the forest floor can result in detrimental effects to microbial biomass and activity.

Summary

Soil microorganisms are complex. Community members range in activity from those merely trying to survive to others responsible for biochemical reactions that are among the most elegant and intricate known. How microorganisms respond to fire will depend on numerous factors, including fire intensity and severity, site characteristics, and preburn community composition. Some generalities can be made, however. First, most studies have shown strong resilience by microbial communities to fire. Recolonization to preburn levels is common, with the amount of time required for recovery generally varying in proportion to fire severity. Second, the effect of fire is greatest in the forest floor (litter and duff). We recommend prescriptions that consume major fuels but protect forest floor, humus layers, and soil humus.

Notes

Part B

Effects of Fire on Water

Daniel G. Neary
Peter F. Ffolliott

Part B—The Water Resource: Its Importance, Characteristics, and General Responses to Fire

Introduction

Effects of fire on the hydrologic cycle are determined largely by the severity of the fire, decisions made relative to any suppression activities, and the immediate postfire precipitation regime. Because information is typically scarce for portions of the spectrum of conditions in which a fire might occur, it is not possible to adequately describe the possible impacts of fire in all conceivable situations. But by understanding the nature of the hydrologic processes impacted, we can interpret the impact of fire on these processes at least to the degree needed to make adequate management decisions.

This chapter covers the hydrologic processes represented by the components of the hydrologic cycle (Brooks and others 2003). We review how changes in hydrologic processes that are brought about by, or attributed to, the occurrence of fire, can translate into changes in streamflow regimes.

Hydrologic Cycle

The *hydrologic cycle* represents the processes and pathways by which water is circulated from land and water bodies to the atmosphere and back again. While the hydrologic cycle is complex in nature and dynamic in its functioning, it can be simplified as a system of water-storage components and the solid, liquid, or gaseous flows of water within and between storage points (fig. B.1). Precipitation inputs (rain, snow, sleet, and so forth) to a watershed are affected little by burning. However, interception, infiltration, evapotranspiration, soil moisture storage, and the overland flow of water can be significantly affected by fire. It must be kept in mind that these components of the hydrologic cycle are closely interrelated, and therefore, it is difficult in practice to isolate the impacts of fire on one component alone.

A generalized percentage breakdown of water inputs, fluxes, and outputs in undisturbed forested watersheds is shown in figure B.2 (Hewlett 1982). These

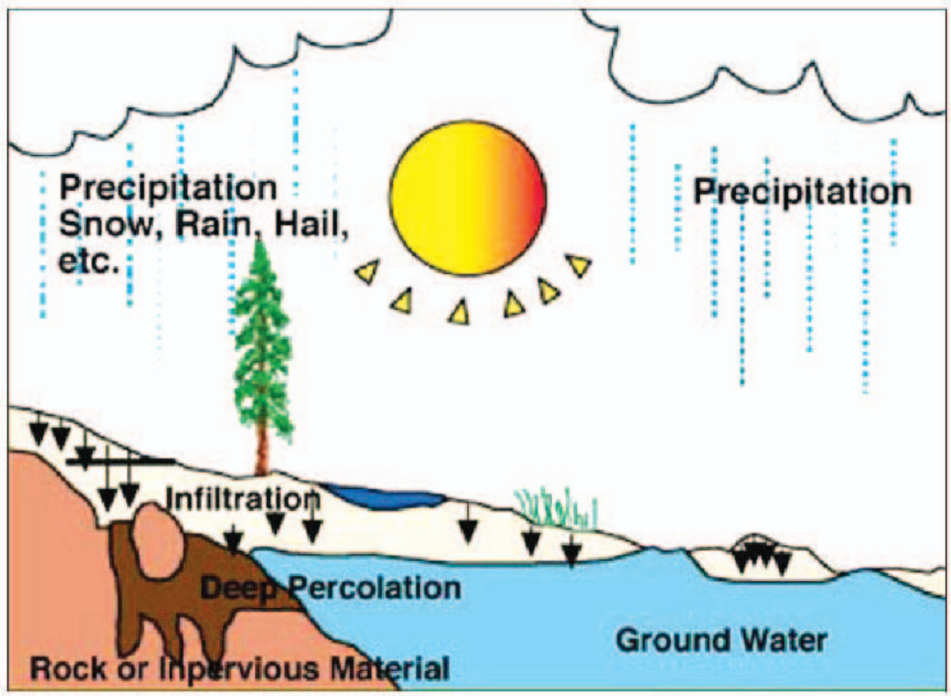

Figure B.1—Generalized diagram of the hydrologic cycle. (Figure courtesy of the USDA Forest Service, National Advanced Fire and Resource Institute, Tucson, AZ, Tucson, AZ).

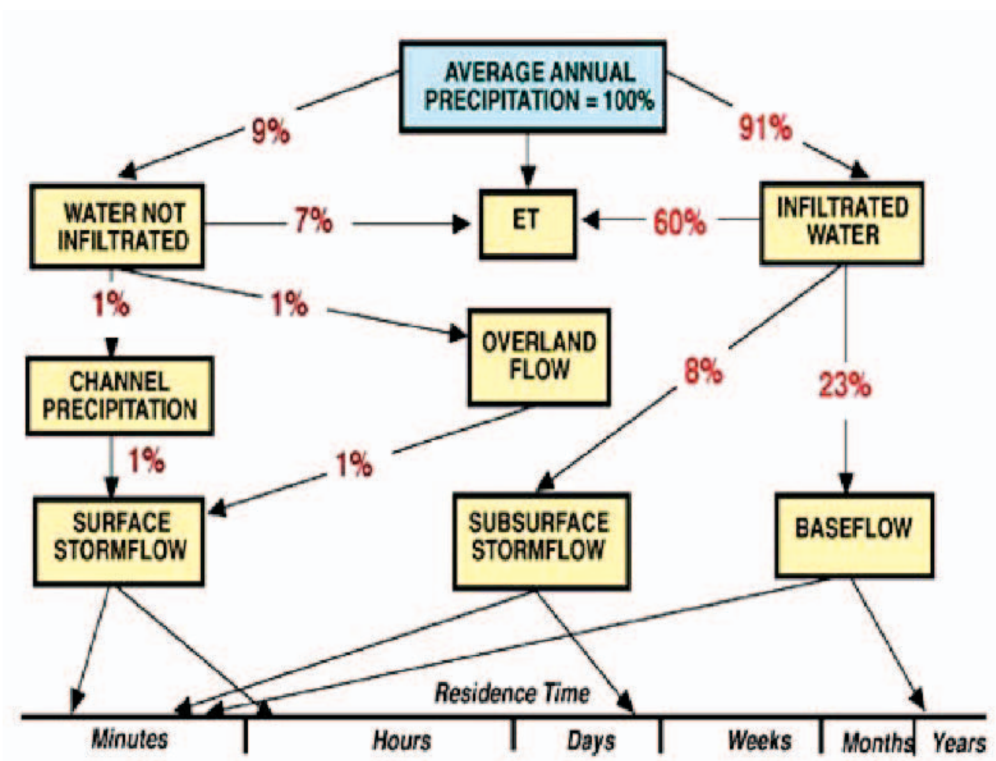

Figure B.2—Inputs, outputs, and fluxes of water in forested watersheds. (Adapted from Hewlett 1982, Principles of Hydrology, Copyright © University of Georgia Press, Athens, GA; Figure courtesy of the USDA Forest Service, National Advanced Fire and Resource Institute, Tucson, AZ).

movements of water can change somewhat in shrub and grassland ecosystems and are altered in watersheds disturbed by harvesting (fig. B.3), burning, insect defoliation, windthrow, land-use conversions, mining, agriculture, and so forth. Precipitation inputs consist of rain, snow, sleet, and so forth. Fluxes, or movement pathways within a watershed, consist of interception, stemflow, throughfall, infiltration, surface runoff, interflow, baseflow, and stormflow.

Interception

Interception is the hydrologic process by which vegetative canopies and accumulations of litter and other decomposed organic matter on the soil surface interrupt the fall of precipitation from the atmosphere to the soil surface. Interception plays a hydrologic role of protecting the soil surface from the energy of falling raindrops. Without this dissipation of energy, the mineral soil surface can become compacted or dislodged by raindrop splash, which then impacts the infiltration characteristics of the soil surface and the pathways of water to stream systems within a watershed.

Much of the precipitation that is intercepted returns to the atmosphere by evaporation and, therefore, becomes a loss of water from the soil surface. As a consequence, interception is a storage term that is subtracted from the gross precipitation input to a watershed in water budget studies. However, not all of the precipitation intercepted by a vegetative canopy or litter layer is returned to the atmosphere. Some of the water intercepted by a vegetative canopy drips off the foliage (*throughfall*) or flows down the stems of trees to the soil surface (*stemflow*). This is especially the case with the occurrence of large storms of long

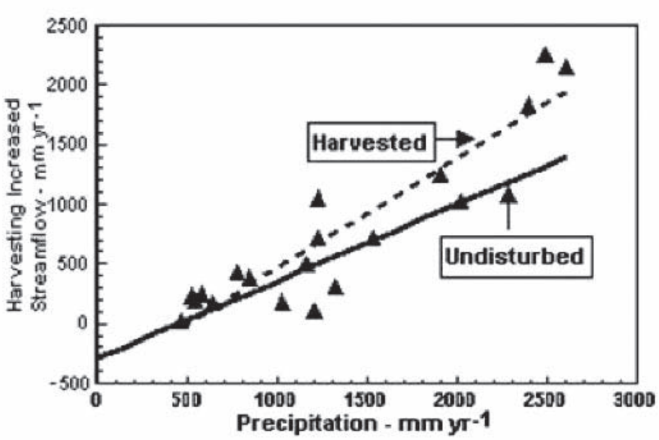

Figure B.3—Annual streamflow response to timber harvesting and precipitation. (From Neary, D.G. 2002. Chapter 5.3: Hydrologic values Page 200, Figure 5.3-6. In: Bioenergy from Sustainable Forestry, All Rights Reserved Copyright © Kluwer Academic Publishers).

duration (Brooks and others 2003). A portion of the water that is intercepted by litter layers also drains to the soil surface.

There is considerable variability in the magnitude of rainfall interception by vegetative canopies. Interception losses in temperate forests of North America range from 0.05 to 0.26 inch (13 to 66 mm) in individual storm events (Helvey 1971, Luce 1995). This amounts to less than 5 to more than 35 percent of the annual rainfall input to a watershed (Aldon 1960, Rothacher 1963, Helvey and Patric 1965, Fuhrer 1981, Roth and Chang 1981, Plamondon and others 1984). Interception losses in the sparsely stocked woodlands, shrublands, and grasslands of arid and semiarid regions are typically less than 10 percent of the annual rainfall (Skau 1964a, Tromble 1983, Haworth and McPherson 1991). Brooks and others (2003) discuss formulas used to calculate rainfall interception. Regardless of the region, however, interception of rainfall represents a transient form of water storage in the vegetative cover of a watershed.

Interception of snowfall is more difficult to quantify than the interception of rainfall, largely because neither the initial amount of snowfall nor the amount of water in the snow that accumulates on foliage of vegetative canopies can be measured adequately. In many situations, much of the intercepted snow is ultimately deposited in the *snowpack* accumulating on the ground through wind erosion and snowmelt, with subsequent dripping and freezing in the snowpack on the ground (Miller 1966, Hoover and Leaf 1967, Satterlund and Haupt 1970, Tennyson and others 1974).

Interception of either rain or snow by vegetative canopies is largely a function of:

- The form (rain or snow), intensity, and duration of the precipitation event.
- The wind velocity, water vapor gradient away from intercepting surfaces, and other storm characteristics.
- The type of vegetation (broad- or needle-leaved), number of vegetative layers in the canopy, and amount of surface leaf area.

Throughfall is precipitation that falls through a plant canopy and lands on bare soil or litter. Interception of throughfall precipitation by litter and other decomposed organic matter on the soil surface ranges from 5 to 35 percent of the gross precipitation input to a watershed (Brooks and others 2003). Storage of intercepted water in litter layers can represent a relatively large proportion of small 1 inch (25 mm) and less rainfall events. It can amount to 0.08 inch (2 mm) on trees, 0.04 inch (1 mm) on shrubs, and 0.12 inch (3 mm) on litter. Interception and the storage of water in litter on the soil surface are related to the depth, density, and relative stage of the development of the layers.

One obvious hydrologic consequence of fire destroying vegetative canopies, reducing litter accumulations, or both is its consequent effects on interception losses. It is one of the largest changes in hydrologic response to short-duration, high-intensity summer rain storms brought about by fire. Most of the vegetative canopy and litter is completely lost in severe wildfires, and as a result, comparatively little postfire interception of precipitation occurs (Bond and van Wilgen 1996, Pyne and others 1996, DeBano and others 1998). The effect of fire on interception in this case is a likely increase in the amount of net precipitation reaching the soil surface—that is, the amount of throughfall. When only small quantities of a vegetative cover or litter are consumed in a fire of low severity, the effect of fire on the interception process is less pronounced. Persistence of prefire levels of litter and other decomposed organic matter is important in protecting the soil surface in those situations where vegetation is destroyed by fire. Increased soil loss through erosive processes is often a consequence when large quantities of both protective layers (vegetation and litter) are lost to fire.

Infiltration

Precipitation that reaches the soil surface moves into the soil mantle, forms puddles of water on the soil surface, or flows over the soil surface. The process of water entering the soil is *infiltration*. The maximum rate at which water can enter the soil is the infiltration capacity. Water that infiltrates into the soil either moves slowly downward and laterally to a stream channel by *interflow* or downward still farther to a groundwater aquifer. When more water is supplied to a site than can infiltrate, the excess waterflows off the surface, by the process of *overland flow* or *surface runoff*, to a stream channel.

The relative proportion of the net precipitation that infiltrates into the soil and moves to a stream channel or percolates through the soil to the groundwater aquifer largely determines the amount and timing of streamflow that ultimately occurs on a watershed. Infiltrometer measurements indicate that undisturbed forest soils have high infiltration capacities compared to other types of soil (Meeuwig 1971, Johnson 1978, Johnson and Beschta 1980, Sidle and Drlica 1981). This high rate of infiltration is a major factor contributing to the popularly held idea that forests have a moderating effect on streamflow regimes.

Forest soils are generally porous and open on the surface because of the accumulations of organic matter on the soil surface, the relatively high organic content of forest soils, and the large number of macropores that typically occur as a result of earthworm, insect, and other burrowing animal activities. Infiltration into forest soils in more humid temperate regions is generally higher than that observed in the soils of arid and semiarid regions because of their more permeable structure and the greater stability of aggregates (Hewlett and Troendle 1975). Infiltration capacities of the soils in arid and semiarid regions are often higher on sites dominated by tree species than on shrub or grass-dominated sites (table B.1). Regardless of region, however, variables affecting the infiltration component include:

- The soil texture, structure, porosity, and so forth.
- The accumulations of litter and other decomposed organic matter on the soil surface.
- The composition and structure of the vegetative cover.
- The land use and resultant vegetative changes, which influence infiltration capacities primarily by altering soil water storage.
- Precipitation rate versus infiltration rate.

The latter factor is an important one to consider in the interior Western United States and parts of the South. Typical summer severe thunderstorms often have high, short-duration rainfall bursts (for example, 10 to 15 minute downpours at the rate of 2 inches/hour

Table B.1—Infiltration rates under various surface conditions. (Adapted from Hewlett 1982, copyright University of Georgia Press, Athens, GA).

Surface conditions	Infiltration Rate		Infiltration Description
	in/hr	*mm/hr*	
1. Intact forest floor	>6.3	>160	Very rapid
2. Vegetation	0.2 – 2.0	5 – 50	Slow to moderate
3. Bare soil	0.0 – 1.0	0 – 25	Very to moderately slow
4. Water repellent soil	0.0 – 0.04	0 – 10	Very slow to none

(50 mm/hr). These rainfalls are often confined to 250 to 500 acres (1 to 2 km^2) (Neary 2002).

Infiltration properties of soils are often altered when fire destroys vegetation and litter covers on a watershed (Pyne and others 1996, DeBano and others 1998, Brooks and others 2003). When the burning has been severe enough to exposes bare soil, infiltration can be reduced due to:

- A collapse of the soil structure and a subsequent increase in bulk density of the soil because of the removal of organic matter, which serves as a binding material.
- The consequent reduction is soil porosity.
- Impacts of raindrops on the soil surface causing compaction and a further loss of soil porosity.
- The kinetic forces of raindrop impact displacing surface soil particles and causing a sealing of surface pores.
- Ash and charcoal residues clogging soil pores.

Variables that affect both the infiltration capacity and cumulative infiltration into the soil can be affected by fire to varying degrees, often resulting in decreased infiltration (Zwolinski 1971, Biswell 1973, MaNabb and others 1989), increased overland flow (DeBano and others 1998, Brooks and others 2003), and, ultimately, increased streamflow discharge. Rates of infiltration are a function of a number of factors such as soil texture, vegetation and litter cover, and soil porosity. Infiltration rates with litter present can often exceed rainfall intensities greater than 6.3 inches/hour (greater than 160 mm/hr). Infiltration decreases when soil particle sizes and cover are reduced. Rainfall infiltration rates less than1 inch/hour (less than 25 mm/hr) in bare sands become less than 0.2 inch/hour (less than 5 mm/hr) in clay-textured soils. The variables that influence infiltration include:

- The vegetative cover type.
- The portion of soil surface covered by litter accumulations and other decomposed organic matter.
- The weight (depth) of the litter and other organic material.
- The soil texture, structure, porosity, bulk density, and so forth.

Another soil property that influences the infiltration process is the *wettability* of the soil (see chapter 2). Soils in some vegetative types and regions can develop a characteristic of water repellency following the occurrence of a fire, which (in turn) can reduce infiltration capacities. Although the presence of these *hydrophobic soils* is frequent in these situations, the causes of this condition are not always well known (DeBano 1981 2000a,b, DeBano and others 1998). Most hydrophobic soils repel water as a result of organic, long-chained hydrocarbon substances coating the soil particles. As a consequence, water "beads up" on the soil surface and will not readily penetrate the surface (see fig. 2.8 and 2.9), resulting in a change in infiltration. With this condition, accelerated overland flow and increased surface erosion can occur, especially on steeper slopes.

Hydrophobic soils are typically found in the chaparral shrublands (comprising *Quercus turbinella* and other sclerophyllous species) of southern California. However, hydrophobic soils can be found after fires in other vegetation types. Fires that occur frequently in the chaparral region intensify the hydrophobic condition and, apparently, volatilize organic substances that accumulate in the litter layer in the interval between fires (DeBano 1981, Dunn and others 1988, DeBano and others 1998). The resulting water repellent layer is then driven deeper into the soil profile. This layering arrangement allows rainfall to infiltrate to only a limited depth before the wetting front reaches the water repellent layer, often causing concurrent increases in the amount of overland waterflow. The soil layer above this water repellent layer is also easily eroded and, therefore, affects sedimentation and debris flow production after fire.

A fire can also influence the microclimate of a site by causing greater air and soil temperature extremes (Fowler and Helvey 1978, Pyne and others 1996, DeBano and others 1998, Brooks and others 2003). In cooler temperate regions, these temperature changes can increase potentials for concrete-type soil frost to form, which can then cause a reduction in infiltration capacity that, therefore, is indirectly related to burning (Bullard 1954).

Evapotranspiration

Evaporation from soils, plant surfaces, and water bodies, and water losses from transpiring plants, are collectively the *evapotranspiration* component of the hydrologic cycle. Part of the evapotranspiration component is when vegetation canopies intercept precipitation that is evaporating from plant foliage. Evapotranspiration is often a high percentage of the precipitation in a water budget, approaching 100 percent on some forested watersheds.

The evapotranspiration component of the hydrologic cycle interests hydrologists and watershed managers because its magnitude largely determines the proportion of the total precipitation input to a watershed that is likely to eventually become streamflow or result in groundwater recharge. Evapotranspiration also represents the component of the hydrologic cycle that is influenced the most by vegetative changes on a watershed that are brought about by planned and unplanned land management activities. The evapotranspiration process largely controls the hydrologic response of a watershed to rainfall and snowmelt events; nevertheless, hydrologists and watershed managers still understand little about the process itself or the feedback mechanisms that control the evapotranspiration process in natural environments (Morton 1990, Ffolliott and Brooks 1996, Brooks and others 2003). It is known, however, that the composition, density, and structure of vegetation influence transpiration losses through time. Differences in the transpiration rates among plant communities and individual plant species on a watershed are attributed largely to:

- Differences in rooting characteristics
- Stomatal response
- Albedo of leaf surfaces
- The length of the growing season

Estimated evapotranspiration values in the temperate forests of North America range from 40 to over 85 percent of the annual precipitation. However, these estimates of evapotranspiration vary greatly with different compositions and structures of forest overstories (Croft and Monninger 1953, Brown and Thompson 1965, Johnson 1970). On a watershed-scale, it has been estimated that 80 to 95 percent of the annual precipitation is evaporated from land surfaces or transpired by plants on the forested watersheds in the Southwestern United States, leaving only 5 to 20 percent available for runoff (Ffolliott and Thorud 1977). By contrast, runoff approaches 50 percent of the rainfall, and there are larger snowmelt inputs on the higher mountain watersheds of the Western United States; nevertheless, the evapotranspiration component is still large and potentially subject to modification.

Evapotranspiration represents the largest loss of water in terms of the components of the hydrologic cycle. This is a problem in arid and semiarid regions because of low precipitation (Pillsbury and others 1963, Skau 1964b, Branson and others 1976). In tropical areas, evapotranspiration is high but so is rainfall. In some situations, soil water storage following the end of the growing season in these harsh environments is nil, regardless of the type of vegetative cover, indicating that large quantities of precipitation are lost through the evapotranspiration process.

Watershed management studies throughout the world have demonstrated that streamflow can increase following vegetative changes that reduce evapotranspiration losses (Bosch and Hewlett 1982, Troendle and King 1985, Hornbeck and others 1993, Whitehead and Robinson 1993). That is, following a vegetative change, less precipitation is converted into vapor through the evapotranspiration process, and as a consequence, more water is available for streamflow. Vegetation-modifying or vegetation-replacing fire, therefore, can change evapotranspiration (Bond and van Wilgen 1996, Pyne and others 1996, DeBano and others 1998). Fire that modifies the composition and structure of the vegetation by removing foliar volume will result in less evapotranspiration losses from a watershed. Fire that causes a replacement of deep-rooted, high profile trees or shrubs by shallow-rooted, low profile grasses and forbs is also likely to reduce evapotranspiration losses. In either instance, less evapotranspiration loss following a fire often translates into increased streamflow.

Soil Water Storage

The maximum amount of water that a soil body retains against the force of gravity is the *field capacity* of the soil. When water is added to a soil that is already charged to field capacity, the excess water either flows overland to a stream channel or drains from the soil. Soil is normally charged to, or near to, field capacity in periods of high precipitation events and at the start of the plant growing season. However, much of the water that is stored in the soil is consumed by plants in periods of sparse precipitation, and by the evapotranspiration process as the growing season progresses. The soil water deficit occurring at the end of the growing season is satisfied when high precipitation amounts occur once again.

The amount of stored soil water that is lost to evapotranspiration is largely a function of the vegetative type occupying the watershed site. Trees and shrubs have roots that can penetrate deep into the soil and, as a consequence, are able to extract water throughout much of the soil body. On the other hand, grasses, grasslike plants, and forbs have relatively shallow root systems and are only able to use water in the upper foot or so of the soil mantle. Water that infiltrates into the soil surface is stored in the upper layers of the soil profile, percolates through the soil body, or both. Vegetative change has a lesser effect on subsoil properties that influence soil water storage than on those properties impacting on infiltration. It follows, therefore, that effects of vegetative change on the subsurface soil properties that influence soil water storage are not likely to be controlling factors in the hydrologic cycle of a watershed (Brooks and others 2003). However, a vegetative change that affects both the evapotranspiration and infiltration processes can influence soil water storage.

The effects of fire on soil water storage result mostly from the loss of vegetation by the burn, which lowers the evapotranspiration losses (DeBano and others 1998, Brooks and others 2003). Lower evapotranspiration losses (in turn) leave more water in the soil at the end of the growing season than would be present if the vegetation had not been burned (Tiedemann and others 1979, Wells and others 1979, DeBano and others 1998). Overland flows of water and, ultimately, streamflow regimes become more responsive to subsequent precipitation events as a consequence of this increased soil water storage. It is often likely that soil water deficits on the burned sites at the end of the growing season will return to prefire levels in time if the vegetative cover also recovers to conditions that characterized the watershed before burning.

Effects of fire on the water storage of rangeland soils are more variable than those in forested soils. Some investigators have reported that the soil water storage is higher on burned sites, others have found lower soil water storage on these sites, and still others observed no change (Wells and others 1979). Varying severities of fire are often cited as the reason for these differences. Increases in soil water content and pore pressures can be similar to those observed after forest harvesting (Sidle 1985).

Snow Accumulation and Melt Patterns

Much of this introduction to the water resource section of this publication has focused on rainfall as the form of precipitation. However, snowfall is also an important form of precipitation input to watershed lands in many regions. The snowpack melts that accumulate at higher latitudes and higher elevations are often a primary source of water to downstream users. It is not surprising, therefore, that hydrologists and watershed managers can be interested in snow accumulation and melt patterns and the effects of vegetative change on these patterns.

The total snow on a watershed at any point in time throughout the winter is largely a function of the total snowfall (Baker 1990, Satterlund and Adams 1992, Brooks and others 2003). However, greater snow accumulations tend to be found at higher elevations on a watershed than at lower elevations because of the generally greater snowfall and lower temperatures at the higher elevations (Anderson and others 1976, Harr 1976; Ffolliott and others 1989, Ffolliott and Baker 2000). More snow accumulates and is retained longer into the winter season on "cooler" than on "warmer" sites because of lower solar radiation levels impinging on the former sites. More snow also accumulates in sparsely stocked forests than in more dense forests, and additional snowfall is deposited in small openings in a forest canopy because of increased turbulence (Troendle 1983, Ffolliott and others 1989, Brooks and others 2003) or through the reduction in the amount of snow intercepted by the forest canopy (Troendle and Meiman 1984, Satterlund and Adams 1992, Brooks and others 2003). Once snowmelt is initiated in the spring, the rate of melt becomes more rapid in the forest openings than under dense vegetative canopies because of greater levels of solar radiation impinging on the open site. The main effect is due to the reduction in snow pack interception in crowns and subsequent sublimation rather than and "redistribution" effect (Troendle and King 1985).

Fire affects snow accumulation and melt patterns when the burn creates openings in formerly dense vegetative canopies. Not only is the amount of snowfall interception decreased after a fire has destroyed the canopies, but additional snowfall is frequently deposited into the created openings due to the disruptions in wind turbulence over the canopy surface (Satterlund and Haupt 1970). The characteristics of these openings are dependent on the severity of the fire. A wildfire of high severity can destroy much of the forest cover on a watershed, creating many large openings in the process. However, only a few relatively small openings are likely to be created by a low severity fire that consumes only the surface fuels.

Charred trees and other black bodies protruding from a snowpack after a fire can change the reflectivity of the ground surface, inducing more surface heating and earlier and more rapid rates of snowmelt. Earlier snowmelt in the spring can also be attributed to a reduced soil water deficit on a burned site. In other words, not as much water is needed to satisfy a deficit, and as a consequence, overland flow originating from snowmelt starts earlier in the season. Such changes in the timing of snowmelt can also alter the timing, magnitude, and duration of the streamflow regimes.

Overland Flow

That portion of the net precipitation that flows off the soil surface is *overland flow*, also called *surface runoff*, which is a major contributor to many streamflow systems and the main contributor to most intermittent streams. This hydrologic process occurs when the rainfall intensity or the rate of snowmelt exceeds the infiltration capacity of a site. Overland flow is the pathway that moves net precipitation most directly to a stream channel and, in doing so, quickly produces streamflow (the following section on Streamflow Regimes discusses the pathways and process by which excess precipitation becomes streamflow).

The relative contribution of overland flow to the streamflow from a watershed is variable, depending largely on how impervious the soil surface is. Overland flow generally occurs on sites that are impervious, locally saturated, or where the infiltration capacity has been exceeded by the net precipitation or rate of snowmelt (Satterlund and Adams 1992, Brooks and others 2003). Some overland flow can be detained enroute to a stream channel by the roughness of the soil surface and, therefore, slowed its movement to the channel. Influences that vegetation and the soil exert on interception, evapotranspiration, infiltration rates, and the soil moisture content ultimately affect the magnitude of overland flow.

Overland flow is a comparatively large component to streamflow hydrographs for highly impervious areas such as urban landscapes, is typically insignificant for forested watersheds with well-drained and deep soils, but is a problem where soils are shallow, rocky, or fine textured (such as high clay or silt content).

An increase in overland flow often results when a fire decreases interception and infiltration rates. This is a major factor in the observed increases in streamflow and flood peakflows, particularly after high severity wildfires. A high severity wildfire can consume all or nearly all of the protective vegetative cover and litter layer over extensive watershed areas, producing a significant effect on the magnitude of overland flow and, as discussed below, on streamflow from a watershed (Tiedemann and others 1979, Baker 1990, DeBano and others 1998). Formation of hydrophobic soils following fire also reduces infiltration, increases overland flow, and speeds delivery of the overland flow to stream channels (Hibbert and others 1974, Rice 1974, Scott and Van Wyk 1990). Persistence of the increased overland flow following fire relates to the rate at which burned sites become revegetated. Prescribed burning often has its greatest hydrologic influence on the infiltration processes and, as a consequence, on the potential for increased overland flow.

Baseflows and Springs

Baseflow is the streamflow between storm events and originates from infiltrated rainfall or groundwater flow. It can increase when the watershed condition is maintained and deep-rooted vegetation on the watershed is harvested or otherwise cut, removed in converting from one vegetation type to another, or killed by fire, insects, or disease. However, baseflow is likely to decrease when the watershed condition deteriorates as a consequence of the disturbance, and more excess precipitation leaves a watershed as overland flow. In extreme situations, perennial streams that are sustained by baseflow become ephemeral.

Baseflows are important in maintaining perennial flow through the year. They are critical for aquatic species habitat and survival. Baseflows can increase if watershed condition remains good (infiltration remains adequate) and deep-rooted vegetation is cut (harvesting), removed (species conversion), or killed by fire, insects, disease, herbicides, and so forth. If watershed condition deteriorates and more precipitation leaves as surface runoff, baseflows will decrease. In extreme conditions, perennial streams become ephemeral. The effect on biota in aquatic ecosystems then becomes devastating. Even in subtropical or tropical areas, deterioration of watershed condition can result in the loss of perennial baseflow and ultimately in desertification.

Crouse (1961) reported increased baseflows from burned watersheds on the San Dimas Experimental Forest in southern California. While these watersheds had been cleared of their chaparral shrubs and associated vegetation by burning, seeded to grass, and maintained in a grass cover to induce higher streamflow discharges, the author and others (Dunn and others 1988, DeBano and others 1998) felt that the wildfire had made a significant contribution to the increased baseflow.

Berndt (1971) observed immediate increases in baseflow following a wildfire on a 1,410 acre (564 ha) watershed in eastern Washington. While the causative hydrologic mechanisms involved were unknown, the removal of riparian (streambank) vegetation by the fire also eliminated diurnal fluctuations of flow. The increased baseflow persisted above prefire levels for 3 years after the fire.

Pathways and Processes __________

Before considering how fire affects streamflow regimes, however, it is useful to review the pathways and processes of waterflow from a watershed's hillslope to a stream channel. *Excess water* represents that portion of total precipitation that flows off the land surface plus that which drains from the soil and, therefore, is neither consumed by evapotranspiration nor leaked into deep groundwater aquifers (table B.2). Various pathways by which excess water eventually becomes streamflow (Brooks and others 2003) include:

- Interception of precipitation that falls directly into a stream channel, a streamflow pathway referred to as *channel interception*.
- Overland flow (see previous discussion).
- *Subsurface flow* that represents the part of precipitation that infiltrates into the soil and arrives at a stream channel in a short enough period to be considered part of the stormflow hydrograph. The stormflow components of a hygrograph are the sum of channel interception, overland flow, and subsurface flow.
- A perennial stream that is fed by baseflow that sustains streamflow between precipitation or snowmelt events.

It is almost impossible to separate pathways of waterflow in most investigations of streamflow responses to the effects of fire or other watershed disturbances (Dunne and Leopold 1978, Satterlund and Adams 1992, DeBano and others 1998, Brooks and

Table B.2—Annual precipitation inputs and resultant annual streamflow totals for different vegetative types. (Adapted from Dortignac 1956).

Vegetation zone	Precipitation		Runoff		Streamflow as a percent of precipitation	Altitude	
	in	*mm*	*in*	*mm*	*Percent*	*ft*	*m*
Spruce fir-aspen	30.39	772	9.49	241	29.0	8,990	2,740
Mountain grassland	22.99	584	5.98	152	26.0	8,000	2,440
Ponderosa pine	22.99	584	3.82	97	17.0	7,510	2,290
Sagebrush	15.20	386	0.79	20	5.3	4,990	1,520
Pinyon-juniper	14.29	363	0.39	10	2.8	4,990	1,520
Semiarid grassland	12.20	310	0.12	3	0.8	4,990	1,520
Greasewood/saltbush	10.39	264	0.16	4	1.4	4,495	1,370
Creosote bush	8.39	213	0.04	1	0.6	4,495	1,370

others 2003). Responses of streamflow to fire are more generally evaluated when possible and where appropriate by separating the stormflow component of the hydrograph from the baseflow (when a baseflow is present) and then studying the two flow components separately (fig. B.4).

The Variable Source Area Concept (Hewlett and Hibbert 1967) describes how the perennial channel system expands during precipitation as areas at the head and adjacent to perennial channels become saturated during the event. Figure B.5 depicts the concept of how a stream channel system expands during precipitation. It is important to understand this concept in order to understand the hydrologic response of a watershed and its component areas, particularly after disturbances such as fire. Important hydrologic and geomorphic processes (sediment transport, channel scour and fill, streambank erosion, fish habitat damage, riparian vegetation damage, nutrient transport, woody debris transport and deposition, and so forth.) occur during stormflows when water volumes and velocities are at their highest. Low severity fires usually do not affect the flow pathways shown in figure B.2, but severe fires shift more of the movement of water to the “water not infiltrated” side of flow diagram. The magnitude and duration of stormflow is a function of the intensity and duration of precipitation as well as a factor called watershed condition.

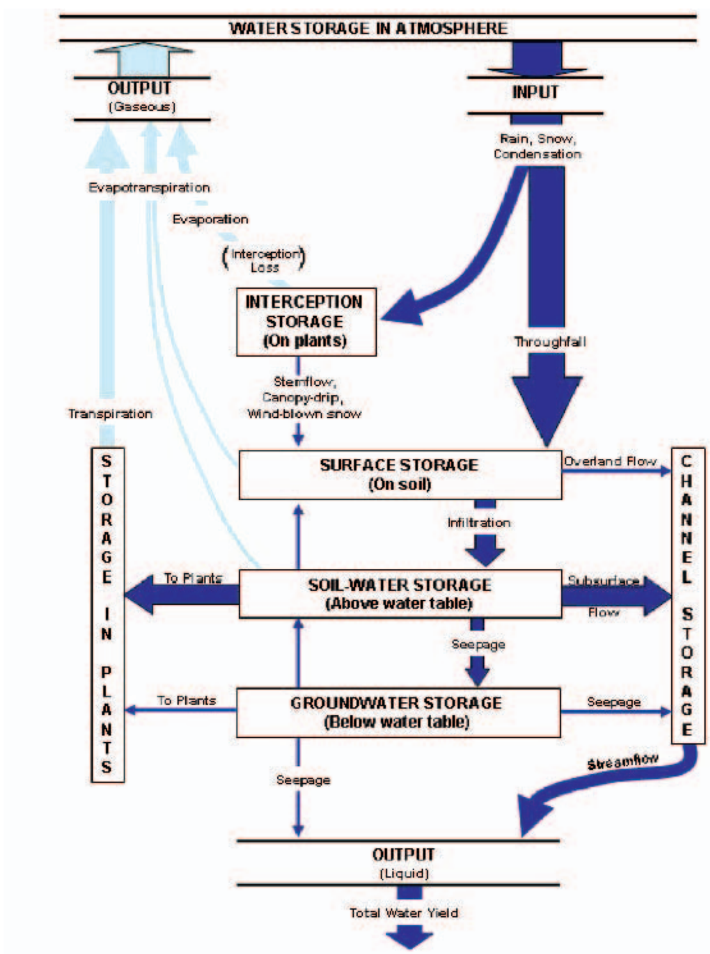

Figure B.4—Relationship between the pathways of flow from a watershed and the resultant streamflow hydrograph. (Adapted from Anderson and others 1976).

Watershed Condition

The timing, magnitude, and duration of a stormflow response to fire are largely a function of the hydrologic condition of the watershed. *Watershed condition* is a term that describes the ability of a watershed system to receive and process precipitation without ecosystem or hydrologic degradation (Brooks and others 2003).

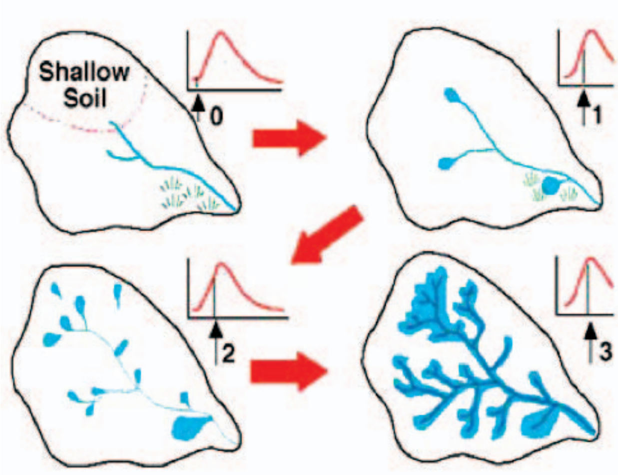

Figure B.5—Variable area source concept of streamflow network expansion, from time = 0 to time = 3, during storm events. (Adapted from Hewlett 1982, Principles of Hydrology, Copyright © University of Georgia Press, Athens, GA; Figure courtesy of the USDA Forest Service, National Advanced Fire and Resource Institute, Tucson, AZ).

Rainfall infiltrates into the soil, and the baseflow of perennial streams is sustained between storms when a watershed is in good condition. In this situation, rainfall does not contribute significantly to increased erosion because most of the excess precipitation does not flow over the soil surface where it can detach and transport sediments.

A severe wildfire can alter the condition of a watershed, however, reducing it to a generally poorer state. With poor watershed condition, the amount of infiltrated rainfall is reduced significantly, more excess precipitation then flows over the surface of the soil, and there is little or no baseflow between storms. Erosion rates are relatively higher because of the excessive overland flow. Watersheds in arid or semiarid regions with rocky and thin soils almost always function hydrologically in this manner because of their inherent climate, soils, and vegetative features. These watersheds are prone to damaging flash floods.

Watershed condition at a point in time is controlled largely by the composition and density of the vegetative cover, the accumulations of litter and other organic material, and the amount of exposed rocks and bare soils that characterizes the watershed. Because a wildfire of high severity can destroy the vegetation and litter layer on a watershed and alter the physical properties of the soil, the infiltration and percolation capacities of the soil are detrimentally impacted (see chapter 2). In turn, these cumulative fire effects can change the watershed condition from good to poor, resulting in ever-increasing overland flow, erosion, and soil loss. An analogy to this situation is a loss of function in the human skin with increasing severity of burning. As watershed condition deteriorates, the inherent hydrologic processes become altered, increasing the likelihood of adverse hydrologic responses to fire. The change in infiltration rates going from "good" (intact forest floor) to "poor" (bare soil plus water repellency) is one example of this response.

Streamflow Discharge

The *streamflow discharge* for a specified time interval (year, month, season, and so forth) is reflected by stormflow, baseflow, or combinations of the two pathways. Stormflow results directly from a precipitation or snowmelt event. When vegetation and organic matter on the soil surface are destroyed by fire, interception and evapotranspiration are reduced, infiltration is decreased, and overland flow and subsurface flow can increase. In turn, increases in overland flow and subsurface flow often translate into:

- Increases in stormflow from the burned watershed.
- Increases in baseflow if the stream is perennial in its flow.
- Increases in streamflow discharge as a consequence of the increases in stormflow, baseflow, or both.

Onsite fire effects must be determined initially, and these effects can then be evaluated within the context of the entire watershed to determine the responses of streamflow discharge and, more generally, the other changes in a streamflow regime to a fire. While determining the onsite effects of a fire can be relatively straightforward through appropriate measurements and evaluations, determining effects of fire on a watershed-scale is more difficult (Pyne and others 1996, DeBano and others 1998). With the latter, combining all processes and pathways of waterflow to the outlet of the watershed, and routing this flow to downstream points of interest or use, are necessary (Brooks and others 2003). Concerning the streamflow discharge at the outlet of a watershed, the effects of fire on the timing can be diminished and the magnitude can be lengthened as the postfire streamflow event moves downstream.

Hydrologists and watershed managers, when studying the effects of fire on the streamflow regime of a watershed, are also concerned with the likelihood of changes in timing of flow. Information on this topic is limited, but some researchers note that streamflow from burned watersheds often responds to rainfall inputs faster than watersheds supporting a protective vegetative cover, producing streamflow events where time-to-peak is earlier (Campbell and others 1977, DeBano and others 1998, Brooks and others 2003). Earlier time-to-peak, coupled with higher peakflow, can increase the frequency of flooding. Timing of snowmelt in the spring can also be advanced by fire in some instances. Early snowmelt can be initiated by lower snow reflectivity (albedo) caused by blackened trees and increased surface exposure where vegetative cover has been eliminated (Helvey 1973).

There is little doubt that wildfire often has an influence on streamflow discharge, especially a wildfire of high severity. The combined effects of a loss of vegetative cover, a decrease in the accumulations of litter and other decomposed organic matter on the soil surface, and the possible formation of water repellent soils are among the causative mechanisms for the increase in streamflow discharge (Tiedemann and others 1979, Baker 1990, Pyne and others 1996, DeBano and others 1998, Brooks and others 2003). While the increases in streamflow discharge are highly variable, they are generally greater in regions with higher precipitation as illustrated by studies in the United States and elsewhere (fig. B.3).

Water Quality

Increases in streamflow following a fire can result in little to substantial impacts on the physical, chemical, and biological quality of water in streams, rivers, and lakes. The magnitude of these effects is largely dependent on the size, intensity, and severity of the fire, the condition of the watershed when rainfall starts, and the intensity, duration, and total amount of rainfall. Postfire streamflow can transport solid and dissolved materials that adversely affect the quality of water for human, agricultural, or industrial purposes. The most obvious effects are produced by sediments. See chapter 2 for more discussion on these components of water quality, which in the following chapters on the water resource are relative to information on municipal water supply quality.

Water quality refers to the physical, chemical, and biological characteristics of water relative to a particular use. Important characteristics of interest to hydrologists and watershed managers include sediment, water temperature, and dissolved chemical constituents such as nitrogen, phosphorus, calcium, magnesium, and potassium. Bacteriological quality is also important if water is used for human consumption or recreation; this is the case with many waters that are both within, and that drain from, forested lands.

A *water quality standard* refers to the physical, chemical, or biological characteristics of water in relation to a specified use. Changes in water quality due to a watershed management practice or natural and human-caused disturbances can make the water flowing from the watershed unsuitable for drinking. However, it might still be acceptable for other uses. In some instances, laws or regulations prevent water quality characteristics from becoming degraded to the point where a water quality standard is jeopardized (DeBano and others 1998, Landsberg and Tiedemann 2000, Brooks and others 2003). The main purpose of these laws and regulations is maintaining the quality of water for a possible and maybe unforeseen future use.

Hydrologists and watershed managers often confront the issue of whether forest or rangeland fires will create conditions in natural or impounded waters that are outside of the established water quality standards. Water quality standards and criteria established by the Environmental Protection Agency (EPA) (1999) are the “benchmarks” for water quality throughout the United States. Water quality criteria are the number or narrative benchmarks used to assess the quality of water. Standards include criteria, beneficial uses, and an antidegradation policy. The most adverse effects from wildfires on water quality standards come from physical effects of the sediment and ash that are deposited into streams.

Several chemical constituents that are regulated by water quality standards are likely to be impacted by burning. *Primary standards* in the EPA regulations cover nitrate-nitrogen, nitrite-nitrogen, and other substances that are not immediately associated with fire. *Secondary standards* apply to pH, sulfate, total dissolved solids, chloride, iron, turbidity, and several other constituents. Secondary standards are also set for color and odor. Chapter 6 examines the water quality standards that can be impacted, and in some cases exceeded, by fire.

Changes in the hydrologic cycle caused by fires can affect the rate of soil erosion, and the subsequent transport and deposition of eroded soil as sediment into streams, lakes, and reservoirs (DeBano and others 1998, Brooks and others 2003). Chapter 5 looks at fire-produced alterations in the hydrologic cycle that in turn affect soil erosion, sedimentation, and water quality.

Maintaining a vegetative cover or a cover of litter and other organic material on the soil surface is the best means of preventing excessive soil erosion rates. However, fire can remove these protective covers and accelerate soil erosion (Dunne and Leopold 1978, Satterlund and Adams 1992, Brooks and others 2003). Increased soil erosion is often the most viable effect of a fire other than the loss of vegetation by burning.

Aquatic Biology

Prior to the 1990s, little information existed on the effects of wildfire on fishes, other aquatic organisms such as macroinvertebrates, and their habitats.

Severson and Rinne (1988) reported that most of the focus of postwildfire effects on riparian-stream ecosystems has traditionally been on hydrological and erosional responses. The Yellowstone Complex Fires in 1988 ushered in an extensive effort to examine both the direct and indirect effects of wildfire on aquatic ecosystems (Minshall and others 1989a, Minshall and Brock 1991). Most of the information available on fire effects on fishes and their habitats was generated in the 1990s and on a regional basis. By the late 1990s, state-of-knowledge papers on the topic of fire, aquatic ecosystem, and fishes were drafted by Rieman and Clayton (1997) and Gresswell (1999). These two papers suggest future research and management direction for both corroborating aquatics-fisheries and fire management and conservation of native, sensitive species. These papers are have become the base for 21st century fisheries and aquatic management relative to both wild and prescription fires. Chapter 7 addresses both the direct and indirect effects of wildland fires and associated suppression activities on aquatic ecosystems.

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Chapter 5: Fire and Streamflow Regimes

Introduction

Forested watersheds are some of the most important sources of water supply in the world. Maintenance of good hydrologic condition is crucial to protecting the quantity and quality of streamflow on these important lands (fig. 5.1). The effects of all types of forest disturbance on storm peak flood flows are highly variable and complex, producing some of the most profound hydrologic impacts that forest managers have to consider (Anderson and others 1976). Wildfire is the forest disturbance that has the greatest potential to change watershed condition (DeBano and others 1998).

Wildfires exert a tremendous influence on the hydrologic conditions of watersheds in many forest ecosystems in the world depending on a fire's severity, duration, and frequency. Fire in these forested areas is an important natural disturbance mechanism that plays a role of variable significance depending on climate, fire frequency, and geomorphic conditions. This is particularly true in regions where frequent fires, steep terrain, vegetation, and postfire seasonal

Figure 5.1—Ponderosa pine watershed, Coconino National Forest, with a good watershed condition. (Photo by Malchus Baker, Jr.).

precipitation interact to produce dramatic impacts (Swanson 1981, DeBano and others 1998, Neary and others 1999).

Watershed condition, or the ability of a catchment system to receive and process precipitation without ecosystem degradation, is a good predictor of the potential impacts of fire on water and other resources (such as roads, recreation facilities, riparian vegetation, and so forth). The surface cover of a watershed consists of the organic forest floor, vegetation, bare soil, and rock. Disruption of the organic surface cover and alteration of the mineral soil by wildfire can produce changes in the hydrology of a watershed well beyond the range of historic variability (DeBano and others 1998). Low severity fires rarely produce adverse effects on watershed condition. High severity fires usually do (fig. 5.2). Most wildfires are a chaotic mix of severities, but in parts of the world, high severity is becoming a dominant feature of fires since about 1990 (Neary and others 1999, Robichaud and others 2000). Successful management of watersheds in a postwildfire environment requires an understanding of the changes in watershed condition and hydrologic responses induced by fire. Flood flows are the largest hydrologic response and most damaging to many resources (fig. 5.3, Neary 1995).

The objective of this chapter is to examine some of the effects of fire on watershed hydrology.

Soil Water Storage

The effects of fire on soil water storage can be illustrated by a wildfire that occurred on a 1,410 acre (564 ha) watershed in eastern Washington that effectively changed the magnitude of the autumnal soil water deficit on the watershed (Klock and Helvey 1976). The mixed conifer forest vegetation on this watershed apparently depleted all of the available soil water in the upper 48 inches (120 cm) of the soil profile immediately before the August 1970 fire (fig. 5.4). The difference between the soil water deficits from 1970 to 1971 was about 4.6 inches (116 mm), which (researchers concluded) contributed a significant part to the increased streamflow discharge reported by Helvey and others (1976). The transpiration draft of large conifer trees had been removed by 1971, and the

Figure 5.2—Flare-up on the Rodeo-Chediski Fire, 2002, Apache-Sitgreaves National Forest. (Photo by USDA Forest Service).

Figure 5.3—Postwildfire flood flows are the largest hydrologic response, and most damaging to many resources after the fire itself. These floods can be 100-fold greater than prefire flood flows. (Photo by John Rinne).

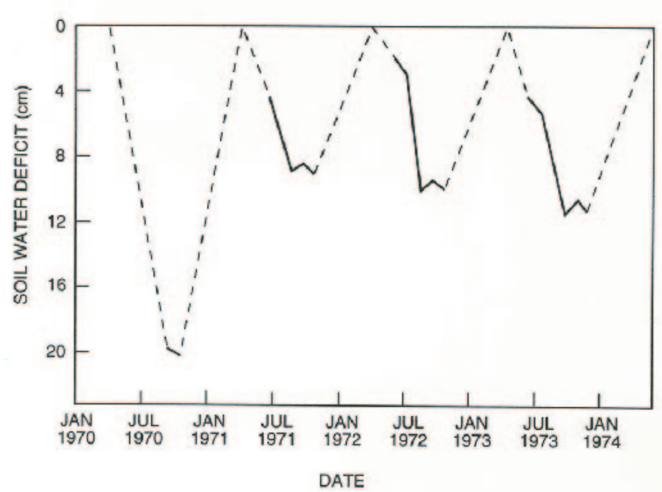

Figure 5.4—Trends in the autumnal soil water deficit in the upper 48 inches (120 cm) of the soil profile for 3 years following the August 1970 wildfire Entiat Experimental Forest, Washington. (Adapted from Klock and Helvey 1976. Copyright © 1976 Tall Timbers Research Station).

observed soil water deficit was an apparent result of surface evaporation and transpiration by the newly established postfire vegetation. The increased autumnal soil water deficit in 1972 and 1974 was caused by the greater evapotranspiration demand by increased vegetative regrowth (Tiedemann and Klock 1976). The trends for postfire years 1971 to 1974 suggest that the minimum soil contents might reach prefire levels in about 5 years after the wildfire.

This trend does not necessarily hold true in all instances, however. For example, a reduction in water storage was observed in the upper 12 inches (30 cm) of the soil on a watershed in northern Arizona where the ponderosa pine (*Pinus ponderosa*) forest had been severely burned, in comparison to the soil storage of an adjacent unburned watershed (Campbell and others 1977). Greater overland flow from the burned watershed was the factor underlying this difference. Water repellency of the soil and the increased drying of the more exposed soil surface contributed to this increased flow.

Effects of fire on the water storage of rangeland soils are more variable than those in forested soils. Some investigators have reported that the soil water storage is higher on burned sites, others have found lower soil water storage on these sites, and still others observed no change (Wells and others 1979). Varying severities of fire are often cited as the reason for these differences.

Baseflows and Springs

Wildfires in 1996 and 2000 resulted in a number of anecdotal reports of springs beginning to flow after years of being dry. This sort of response is common in the Southwest and regions such as central Texas when an area is cleared or burned (Thurow, personal communication). Often trees such as juniper or live oak have increased in density and size along with the onset of effective fire control in the early 1900s. As trees begin to dominate ecosystems such as fire-climax grassland savannas, the trees alter the water balance because they have substantially greater interception loss and transpiration capacity. The local soils and geology determine whether water yield occurs as spring flow or groundwater recharge. The seasonal patterns of the amount and timing precipitation and potential evapotranspiration determine whether there is any excess water to contribute to water yield. For example, on arid sites all precipitation would be lost to evapotranspiration and thus essentially nothing would percolate fast enough beyond the root system, or evaporate from the soil surface, to recharge springs or aquifers. Some shrub sites may yield substantial amounts of water, others may yield nothing (Wu and others 2001).

On the Three Bar chaparral watersheds of the Tonto National Forest in Arizona, watersheds that had no flow or were intermittent responded to a wildfire (Hibbert and others 1974). Watershed B yielded no flow prior to the fire. It flowed continuously for 18 months, then was intermittent until treated with herbicides when it returned to perennial flow for 10 plus years. Watershed D was dry 67 percent of the time prior to the wildfire. Following the fire, it then resumed continuous flow until brush regrew, when it returned to intermittent flow. On the Whitespar Watersheds, Watershed B resumed perennial flow after 38 acres (15 ha) adjacent to the channel were treated with herbicide. Similar responses have been found in California chaparral (Crouse 1961). Treatment of the Natural Drainages, Sierra Ancha, with herbicides did not produce perennial flow but did increase the duration of intermittent flows. The soils at this site are more shallow than those at Three Bar or Whitespar. The key here is control of vegetation in deep-soil, arid systems where transpiration from deep-rooted plants can consume water that would otherwise become perennial streamflow. The response is quickly terminated as deep-rooted, brush-chaparral trees regrow. Perennial flow can only be maintained by converting to (or back to) shallow-rooted herbaceous species. Increases in any single year are affected by rainfall. Often, 80 percent of the increased water yield over a 10-year period occurs in just a few wetter-than-average years, not every year.

Streamflow Regimes

Annual streamflow totals (annual water yields) generally increase as precipitation inputs to a watershed increase. Streamflows originating on forest watersheds, therefore, are generally greater than those originating on grassland watersheds, and streamflows from grasslands are greater than those originating on desert watersheds. Furthermore, annual streamflow totals frequently increase when mature forests are harvested or otherwise cut, attacked by insects, or burned (Bosch and Hewlett 1982, Troendle and King 1985, Hornbeck and others 1993, Whitehead and Robinson 1993). The observed increases in streamflow following these disturbances often diminish with decreasing precipitation inputs to a watershed (see fig. B.3).

Effects of Wildfires

Annual streamflow discharge from a 1,410 acre (564 ha) watershed in the Cascade Range of eastern Washington, on which a wildfire killed nearly 100 percent of the mixed conifer forest vegetation, increased dramatically relative to a prefire streamflow relationship between the watershed that was burned and an unburned control (fig. 5.5). Differences between the measured and predicted streamflow discharge varied from nearly 4.3 inches (107 mm) in a dry year (1977) to about 19.1 inches (477 mm) in a wet year (1972). Soil water storage remained high for the period of record largely because of abnormally high precipitation (rain and snow) inputs (Helvey 1980). As a consequence, the burned and control watersheds became more sensitive to precipitation.

Campbell and others (1977) observed a 3.5 times increase of 0.8 inch (20 mm) in average annual stormflow discharge from a small 20.2 acre (8.1 ha) severely burned watershed following the occurrence of a wildfire in a Southwestern ponderosa pine forest. Average annual stormflow discharge from a smaller 10 acre (4 ha) moderately burned watershed increased 2.3 times to almost 0.6 inch (15 mm) in relation to an unburned (control) watershed. Average runoff efficiency-a term that refers to the percentage of runoff to precipitation-increased from 0.8 percent on the unburned watershed to 3.6 and 2.8 percent on the severely burned and moderately burned watersheds, respectively. In comparison to the moderately burned watershed, the average runoff efficiency on the severely burned watershed was 357 percent greater when the precipitation input was rain and 51 percent less in snowmelt periods. The researchers speculated that the observed differences during rainfall events were largely due to the lower tree density, a greater reduction in litter cover, and a more extensive formations hydrophobic soil, resulting in lower evapotranspiration losses and more stormflow on the severely burned watershed than on the moderately burned watershed. In the spring snowmelt period, the lower tree density of the severely burned watershed allowed more of the snowpack to be lost to evaporation. As a result, less stormflow occurred than on the more shaded, moderately burned watershed.

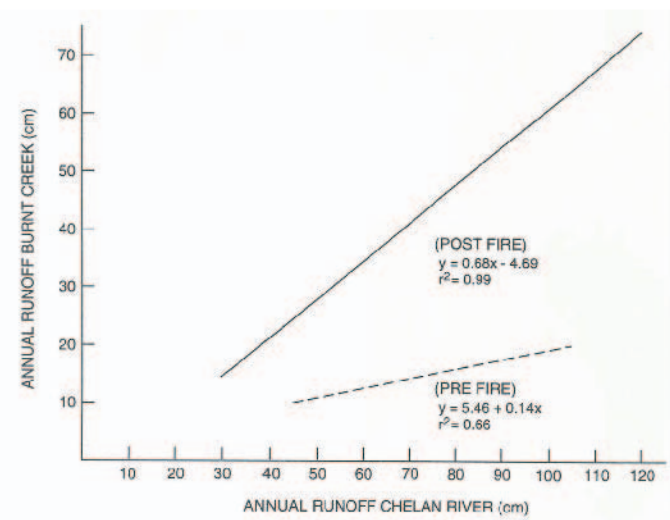

Figure 5.5—Annual streamflow from a burned watershed before and after the fire in relation to annual streamflow from a control. (Adapted from Helvey 1980. Copyright © 1980 American Water Resources Association.)

In the fire-prone interior chaparral shrublands of the Southwestern United States, annual streamflow discharge from their watersheds can increase by varying magnitudes, at least temporarily, as a result of wildfires of high severity (Davis 1984, Hibbert 1984, Baker and others 1998). The combined effects of loss of vegetative cover, decreased litter accumulations, and formation of water repellent soils following the burning are the presumed reasons for these streamflow increases (Rundel 1977, Baker 1999).

In the first year after a 365 acre (146 ha) watershed in southern France, near the Mediterranean Sea, was burned over by a wildfire, streamflow discharge increased 30 percent to nearly 2.4 inches (60 mm) (Lavabre and others 1993). The prefire vegetation on the watershed was primarily a mixture of maquis, cork oak, and chestnut trees. The researchers attributed the increase in annual streamflow discharge to the reduction in evapotranspiration due to the destruction of this vegetation by the fire.

Average annual streamflow discharge increased by about 10 percent to 4.8 inches (120 mm) on a 612 acre (245 ha) watershed in the Cape Region of South Africa following a wildfire that consumed most of the indigenous fynbos (sclerophyllous) shrubs (Scott 1993). This increase was related mostly to the reductions in interception and evapotranspiration losses.

Effects of Prescribed Burning

Streamflow responses to prescribed fire (fig. 5.6) are smaller in magnitude in contrast to the responses to wildfire. It is generally not the purpose of prescribed burning to completely consume extensive areas of litter and other decomposed organic matter on the soil surface (Ffolliott and others 1996, DeBano and others 1998) and, therefore, the drastic alterations in streamflow discharges that are common after severe wildfires do not normally occur. To illustrate this point, the average annual streamflow discharge was not changed relative to prefire levels in 6 years following a prescribed fire on 43 percent of a 1,178 acre (471 ha) watershed in northern Arizona supporting a ponderosa pine forest (Gottfried and DeBano 1990). The

Figure 5.6—Grass prescribed fire in interior Alaska. (Photo by Karen Wattenmaker).

prescribed burning plan specifying a 70 percent reduction in fine fuels and a 40 percent reduction in heavy fuels on the burned area was satisfactorily met, with minimal damage to the residual stand of trees.

A burn that was prescribed to reduce the accumulated fuel loads on a 450 acre (180 ha) watershed in the Cape Region of South Africa resulted in a 15 percent increase to 3.2 inches (80 mm) in average annual streamflow discharge (Scott 1993). Most of the fynbos shrubs that vegetated the watershed were not damaged by the prescribed fire. The effectiveness of the prescribed fire was less than anticipated because of the unseasonably high rainfall amounts at the time of burning.

A prescribed fire in a Texas grassland community (fig. 5.4) resulted in a large increase (1,150 percent) in streamflow discharge in comparison to an unburned watershed in the first year after burning (Wright and others 1982). The increased postfire streamflow discharge was short lived, however, with streamflows returning to prefire levels shortly after the burning.

Burning of logging residues (slash) in timber harvesting operations, burning of competing vegetation to prepare a site for planting, and burning of forests and woodlands in the process of clearing land for agricultural production are common practices in many parts of the United States and the world. Depending on their intensity and extent, the burnings prescribed for these purposes might cause changes in streamflow discharge from the watersheds on which these treatments are conducted. But, in analyzing the responses of streamflow discharge to prescribed fire, it is difficult to isolate effects of these burning treatments from the accompanying hydrological impacts of timber harvesting operations, site preparation, and clearing of forest vegetation.

Results of other studies on the changes in streamflow discharge following either a wildfire or prescribed burning are presented in tables 5.1 through 5.5. These results are summarized on the basis of Bailey's (1995) ecoregion classifications for comparisons purposes.

Peakflows

Peakflows are a special subset of streamflow regimes that deserve considerable attention. The effects of forest disturbance on storm peakflows are highly variable and complex. They can produce some of the most profound impacts that forest managers have to

Table 5.1—Increased water yield from prescribed fire (Rx) burned watersheds, Eastern United States ecoregions.

State	Treatment	Area		Precipitation		1st year runoff		Increase	Recovery
		acre	*ha*	*inch*	*mm*	*inch*	*mm*	*%*	*years*
M221 CENTRAL APPALACHIAN BROADLEAF-CONIFER FOREST PROVINCE [1]									
NC	Hardwoods			67.91	1,725				
	Control	40	16			29.09	739	—	—
	Cut, Rx burn	40	16			35.00	889	20	5
	Swank and Miner 1968								
231 SOUTHEASTERN MIXED FOREST PROVINCE									
SC	Loblolly pine			54.72	1,390				
	Control	5	2			4.88	124	—	—
	Under burn	5	2			7.09	180	45	2
	Rx burn, cut	5	2			8.54	217	75	2
	Van Lear and others 1985								

[1]Bailey's Ecoregions (1995).

Table 5.2—Increased water yield from prescribed fire (Rx) and wildfire burned watersheds, Cascade Mountains ecoregions, U.S.A.

State	Treatment	Area		Precipitation		1st year runoff		Increase	Recovery
		acre	*ha*	*inch*	*mm*	*inch*	*mm*	*%*	*years*
***M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE*[1]**									
WA	Ponderosa pine			22.83	580				
	Prefire basis	1,277	517			8.70	221	—	—
	Wildfire	1,277	517			12.36	314	42	Unknown
	Helvey 1980								
OR	Douglas-fir			97.76	2,483				
	Control	175	71			74.21	1,885	—	—
	Cut, Rx burn	175	71			87.60	2,225	18	>5
	Bosch and Hewlett 1982								
OR	Douglas-fir			94.02	2,388				
	Control	237	96			54.17	1,376	—	
	Cut, Rx fire	237	96			72.36	1,838	34	>5
	Bosch and Hewlett 1982								

[1]Bailey's Ecoregions (1995).

Table 5.3—Increased water yield from prescribed fire (Rx) and wildfire burned watersheds, Colorado Plateau and Arizona-New Mexico mountains semidesert ecoregion, U.S.A.

Ecoregion/State	Treatment	Area		Precipitation		1st year runoff		Increase	Recovery
		acre	*ha*	*inch*	*mm*	*inch*	*mm*	*%*	*years*
***313 COLORADO PLATEAU SEMIDESERT PROVINCE*[1]**									
AZ	Chaparral			29.13	740				
	Control	6	2			2.5		—	—
		9	8			2	64		
	Rx burn	8	3			6.1	15	144	>11
		2	3			4	6		
	Davis 1984								
AZ	Chaparral			23.03	585				
	Control	9	3			3.2		—	—
		6	9			3	82		
	Wildfire	9	3			5.1	30	59	Unknown
		6	9			2			
	Hibbert and others 1982								
AZ	Chaparral			25.79	655				
	Control	4	1			0		—	
		7	9				0		
	Wildfire	4	1			4.8	12	>9,999+	>9
		7	9			8	4		
	Control	4	3			0.7		—	
		7	9			5	19		
	Wildfire	4	3			11.		1,421	>9
		7	9			38	89		
	Hibbert 1971								

[1]Bailey's Ecoregions (1995).

Table 5.4—Increased water yield from prescribed fire (Rx) and wildfire burned watersheds, Arizona-New Mexico mountains semidesert, and Southwest plateau and Plains dry steppe ecoregions, U.S.A.

State	Treatment	Area		Precipitation		1st year runoff		Increase	Recovery
		acre	*ha*	*inch*	*mm*	*inch*	*mm*	*%*	*years*
M313 AZ-NM MOUNTAINS SEMIDESERT-WOODLAND-CONIFER PROVINCE[1]									
AZ	Pinyon-juniper			18.90	480				
	Control	12	5			1.34	34	—	—
	Rx burn	12	5			1.54	39	15	5
	Control	12	5			1.69	43	—	
	Rx burn	12	5			2.20	56	30	>5
	Hibbert and others 1982								
AZ	Pinyon-juniper			18.98	482				
	Control	331	134			0.79	20	—	—
	Slash Rx burn	331	134			0.43	11	–45	4
	Control	363	147			0.71	18	—	
	Herbicide	363	147			1.10	28	56	>4
	Clary and others 1974								
AZ	Ponderosa pine			29.02	737				
	Control	44	18			0.24	6	—	
	Wildfire, low	25	10			0.35	9	50	2
	Wildfire, mod.	10	4			0.79	20	233	7
	Wildfire, high	20	8			1.06	27	350	15
	DeBano and others 1996								
315 SOUTHWEST PLATEAU AND PLAINS DRY STEPPE AND SHRUB PROVINCE[1]									
TX	Juniper/grass	2	<1	25.98	660				
	Control	2	<1			0.08	2	—	
	Rx fire	2	<1			0.98	25	1,150	5
	Rx fire, seeded	2	<1			0.43	10	400	2
	Wright and others 1982								

[1]Bailey's Ecoregions (1995).

Table 5.5—Increased water yield from prescribed burned watersheds, Southern Rocky Mountains ecoregion, U.S.A.

State	Treatment	Area		Precipitation		1st year runoff		Increase	Recovery
		acre	*ha*	*inch*	*mm*	*inch*	*mm*	*%*	*years*
M331 SOUTHERN ROCKY MOUNTAIN STEPPE-WOODLAND-CONIFER PROVINCE[1]									
CO	Aspen, mixed conifer			21.10	536				
	Control	4.88	24			6.18	157	—	—
	Clearcut, Rx burn	4.88	24			7.52	191	22	5
	Bosch and Hewlett 1982								

[1]Bailey's Ecoregions (1995).

consider. The magnitude of increased peakflow following fire (table 5.6) is more variable than streamflow discharges (tables 5.1 to 5.5) and is usually well out of the range of responses produced by forest harvesting. Increases in peakflow as a result of a high severity wildfire are generally related to a variety of processes including the occurrence of intense and short duration rainfall events, slope steepness on burned watersheds, and the formation of soil water repellency after burning (DeBano and others 1998, Brooks and others 2003). Postfire streamflow events with excessively high peakflows are often characteristic of flooding regimes

Peakflows are important events in channel formation, sediment transport, and sediment redistribution in riverine systems (Rosgen 1996, Brooks and others 2003). These extreme events often lead to significant changes in the hydrologic functioning of the stream system and, at times, a devastating loss of cultural resources. Peakflows are important considerations in the design of structures (such as bridges, roads, dams, levees, commercial and residential buildings, and so forth). Fire has the potential to increase peakflows well beyond the normal range of variability observed in watersheds under fully vegetated conditions (table 5.6). For this reason, understanding of peakflow response to fire is one of the most important aspects of understanding the effects of fire on water resources.

Peakflow Mechanisms

A number of mechanisms occur singly or in combination to produce increased postfire peakflows (fig. 5.7). These include obvious mechanisms such as unusual

Table 5.6—Effects of harvesting and fire on peakflows in different habitat types.

Location	Treatment	Peakflow increase factor	Reference
M212 ADIRONDACK-NEW ENGLAND MIXED FOREST PROVINCE			
Hardwoods, NH	Clearcut	+2.0	Hornbeck 1973
M221 CENTRAL APPALACHIAN BROADLEAF-CONIFER FOREST PROVINCE			
Hardwoods, NC	Clearcut	+1.1	Hewlett and Helvey 1970
Hardwoods, WV	Clearcut	+1.2	Reinhart and others 1963
232 COASTAL PLAIN MIXED FOREST PROVINCE			
Loblolly Pine, NC	Rx Fire	0.0	Anderson and others 1976
M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE			
Douglas-fir, OR	Cut 50%, burn	+1.1	Anderson 1974
	Clearcut, burn	+1.3	
	Wildfire	+1.4	
M262 CALIFORNIA COASTAL RANGE WOODLAND-SHRUB-CONIFER PROVINCE			
Chaparral, CA	Wildfire	+20.0	Sinclair and Hamilton 1955
		+870.0	Krammes and Rice 1963
		+6.5	Hoyt and Troxell 1934
313 COLORADO PLATEAU SEMI-DESERT PROVINCE			
Chaparral, AZ	Wildfire	+5.0 (Sum)	Rich 1962
		+150.0 (Sum)	
		+5.8 (Fall)	
		+0.0 (Winter)	
M 313 AZ-NM MOUNTAINS SEMIDESERT-WOODLAND-CONIFER PROVINCE			
Ponderosa pine, AZ	Wildfire	+96.1	Campbell & others 1977
	Wildfire, Mod.	+23.0	
	Wildfire, Severe	+406.6	
	Wildfire, Severe	+2,232.0	Ffolliott and Neary 2003
M331 SOUTHERN ROCKY MOUNTAINS STEPPE-WOODLAND-CONIFER PROVINCE			
Aspen-conifer, CO	Clearcut, Rx burn	–1.50	Bailey 1948
Ponderosa pine, NM	Wildfire	+100.00	Bolin and Ward 1987

Figure 5.7—Flood flow at Heber, AZ, after the Rodeo-Chediski Fire, 2002, Apache-Sitgreaves National Forest. (Photo by Dave Maurer).

rainfall intensities, destruction of vegetation, reductions in litter accumulations and other decomposed organic matter, alteration of soil physical properties, and development of soil hydrophobicity.

A special circumstance sometimes occurs with postwildfire peakflows that can contribute to the large responses (up to three orders of magnitude increase). Cascading debris dam failures have the potential to produce much higher peakflow levels than would be expected from given rainfall events on bare or water repellent soils. This process consists of the establishment of a series of debris dams from large woody debris in and adjacent to stream channels, buildup of water behind the dams, and sequential failure of the first and subsequent downstream debris dams (fig. 5.8).

Figure 5.8—Remnant of a debris dam in the channel of Dude Creek after the Dude Fire, Tonto National Forest, 1991. (Photo by John Rinne).

Concern about this process has led to the use of one type of BAER channel treatment, debris removal. Channels particularly prone to this process would include those with large amounts of woody debris and a high density of riparian trees or boulders, which could act as the dam formation mechanism. After the 1991 Dude Fire in Arizona, Rinne (1994) reported that little of the tagged prefire woody debris moved after a significant postfire flood event. On the other hand, some unusually high flood flows after the 2000 Cerro Grande Fire in New Mexico left distinct evidence of woody debris dam formation and failure (Kuyumjian, Gregpry A., USDA Forest Service, personal communication).

Fire Effects

Anderson and others (1976) provided a good review of peakflow response to disturbance. These responses are influenced by fire severity. Low severity prescribed burning has little or no effect on peakflow because it does not generally alter watershed condition.

Intense short duration storms that are characterized by high rainfall intensity and low volume have been associated with high stream peakflows and significant erosion events after fires (Neary and others 1999). In the Intermountain West, high intensity, short duration rainfall is relatively common (Farmer and Fletcher 1972). Five-minute rainfall rates of 8.38 to 9.25 inches/hour (213 and 235 mm/hour) have been associated with peakflows from recently burned areas that were increased five times that for adjacent, unburned areas (Croft and Marston 1950). A 15-minute rainfall burst at a rate of 2.64 inches/hour (67 mm/hour) after the 2000 Coon Creek Fire in Arizona produced a peakflow that was in excess of sevenfold greater than the previous peakflow during 40 years of streamflow gauging. Moody and Martin (2001) reported on a threshold for rainfall intensity (0.39 inch/hour or 10 mm/hour in 30 minutes) above which flood peakflows increase rapidly in the Rocky Mountains. Robichaud (2002) collected rainfall intensity on 12 areas burned by wildfire in the Bitterroot Valley of Montana. He measured precipitation intensities that ranged from 3 to 15 mm in 10 minutes. The high end of the range was an equivalent to 75 mm/hour (greater than a 100-year return interval). It is these types of extreme rainfall events, in association with altered watershed condition, that produce large increases in stream peakflows and erosion.

Peakflows after forest cutting can increase or decrease depending on location, the percentage of the watershed cut, precipitation regime, and season (table 5.6, fig. B.3). Most studies show increases in peakflows of 9 to 100 percent. The concern with increases in annual flood peakflows is that the increases could lead to channel instability and degradation, and to increased property damage in flood-prone urban areas.

Fire has a range of effects on stream peakflows. Low severity, prescribed fires have little or no effect because they do not substantially alter watershed condition (table 5.6). Severe wildfire has much larger effects on peakflows. The Tillamook Burn in 1933 in Oregon increased the total annual flow of two watersheds by 1.09-fold and increased the annual peakflow by 1.45-fold (Anderson and others 1976). A 127 ha wildfire in Arizona increased summer peakflows by 5- to 150-fold, but had no effect on winter peakflows. Another wildfire in Arizona produced a peakflow 58-fold greater than an unburned watershed during record autumn rainfalls. Campbell and others (1977) documented the effects of fire severity on peakflows. A moderate severity wildfire increased peakflow by 23-fold, but high severity wildfire increased peakflow response three orders of magnitude to 406.6-fold greater than undisturbed conditions. Krammes and Rice (1963) measured an 870-fold increase in peakflow in California chaparral. In New Mexico, Bolin and Ward (1987) reported a 100-fold increase in peakflow after wildfire in a ponderosa pine and pinyon-juniper forest. Watersheds in the Southwest are much more prone to these enormous peakflow responses due to interactions of fire regimes, soils, geology, slope, and climate (Swanson 1981).

Following the Rodeo-Chediski Fire of 2002, peakflows were orders of magnitude larger than earlier recorded. The estimated peakflow on a gauged watershed that experienced high severity stand-replacing fire was almost 8.9 ft^3/sec (0.25 m^3/sec) or nearly 900 times that measured prefire (Ffolliott and Neary 2003). The peakflow on a watershed subjected to low-to-medium severity fire was estimated to be about one-half less, but still far in excess of the previous observations. A subsequent and higher peakflow on the severely burned watershed was estimated to be 232 ft^3/sec (6.57 m^3/sec) or about 2,232 times that measured in snowmelt runoff prior to the wildfire. This latter peakflow increase represents the highest known relative postfire peak flow increase that has been measured in the ponderosa pine forest ecosystems of Arizona or, more generally, the Southwestern United States. However, the specific discharge (94.2 ft^3/sec/mile^2 or 1.02 m^3/sec/km^2) was on the lower end of range of discharges measured by Biggio and Cannon (2001).

Another concern is the timing of stormflows or response time. Burned watersheds respond to rainfall faster, producing more "flash floods." They also may increase the number of runoff events. Campbell and others (1977) measured six events on an unburned watershed after the Rattle Burn and 25 on a high-severity burned watershed. Hydrophobic conditions, bare soils, and litter and plant cover loss will cause flood peaks to arrive faster and at higher levels. Flood warning times are reduced by "flashy" flow, and higher flood levels can be devastating to property and human life. Recovery times after fires can range from years to many decades.

Still another aspect of the postfire peakflow issue is the fact that the largest discharges often occur in small areas. Biggio and Cannon (2001) examined runoff after wildfires in the Western United States. They found that specific discharges were greatest from relatively small areas (less than 0.4 mile^2 or 1 km^2, fig. 5.9). The smaller watersheds in their study had specific discharges averaging 17,664.3 ft^3/sec/mile^2 (193.0 m^3/sec/km^2), while those in the next higher sized watershed category

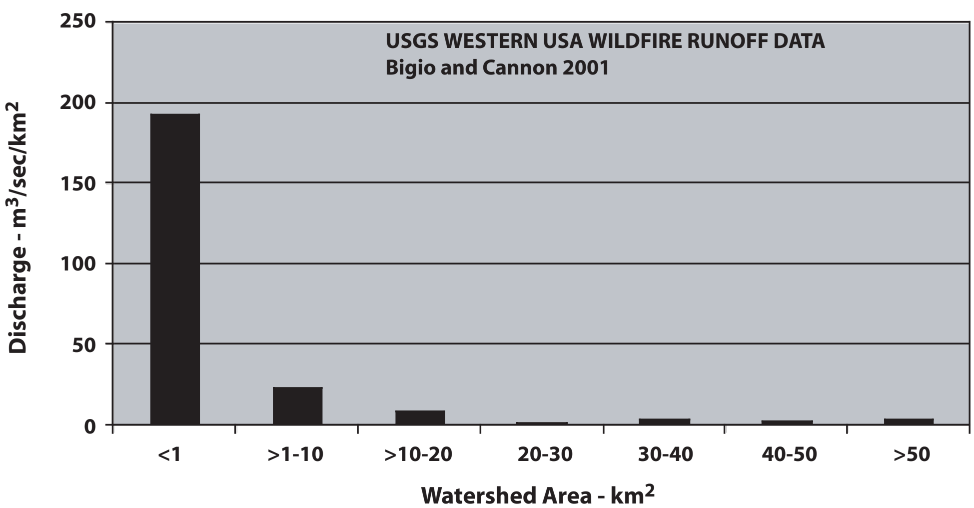

Figure 5.9—Post-wildfire flood specific discharge and watershed area. (Adapted from Biggio and Cannon 2001).

(up to 4 $mile^2$ or 10 km^2) averaged 2,077.6 $ft^3/sec/mile^2$ (22.7 $m^3/sec/km^2$).

So, the net effect on watershed systems and aquatic habitat of increased peakflows is a function of the area burned, watershed characteristics, and the severity of the fire. Small areas in flat terrain subjected to prescribed fires will have little if any effect on water resources, especially if Best Management Practices are used. Peakflows after wildfires that burn large areas in steep terrain can produce significant impacts, but peakflows are probably greatest out of smaller sized watersheds less than 0.4 $mile^2$ (1 km^2). Burned area emergency rehabilitation (BAER) techniques may be able to mitigate some of the impacts of wildfire (see chapter 10). However, the ability of these techniques to moderate the impacts of rainfalls that produce extreme peakflow events is not well documented (Robichaud and others 2000).

Management Implications

Fires affect watersheds, resulting in changes that affect many resource values, including municipal water, visual aspects, recreation, floral and fauna existence and welfare, transportation, and human activity and well being. The vast extent of possible water and watershed changes makes management of these lands crucial.

Prescribed fires with low to moderate burn severity rarely produce adverse hydrologic effects that land managers need to be concerned about. Postwildfire floods are the main concern, particularly the timing of storm flows (response time) and magnitudes of flood peaks. Because burned watersheds respond to rainfall faster, producing more "flash floods," they also may increase the number of runoff events. Flood warning times are reduced by these "flashy" flows, and higher flood levels can be devastating to property and human life. Another aspect of this is the fact that recovery times can range from years to many decades.

So, the net effect on watershed systems and aquatic habitat of increased peak flows is a function of the area burned, watershed characteristics, and the severity of the fire. Small areas in flat terrain subjected to prescribed fires will have little if any effect on water resources, especially if Best Management Practices are utilized. Peak flows after wildfires that burn large areas in steep terrain can produce significant impacts. Burned area emergency rehabilitation watershed techniques may be able to mitigate some of the impacts of wildfire.

However, the ability of these techniques to moderate the impacts of rainfalls that produce extreme peakflows is poorly documented (Robichaud and others 2000). In some circumstances, a lack of willingness has existed to implement effective runoff control measures in a timely and thorough way because of visual and environmental concerns as well as hasty, improper installation techniques. In some instances, BAER techniques such as contour trenching have been able to reduce peakflow events where short duration rainfall intensities are high but total storm volumes are low (Robichaud and others 2000).

Increased flood flow peaks have the potential to damage both natural and cultural resources. Large floods often change stream geomorphology. Aquatic biota and riparian ecosystems can be severely impacted by unusual flood flows. Culverts, bridges, roads, dams, and irrigation structures are at risk by flood flows outside of their design parameters. Recreation facilities, houses, businesses, and community structures can also be affected. Most important, human health and safety can be at serious risk.

Forest managers need to understand these risks in order to design adequate rehabilitation projects, order and enforce recreation area closures, establish flood-warning protocols, and conduct appropriate land management activities.

Summary

Fires affect water cycle processes to a greater or lesser extent depending on severity. Table 5.7 contains a general summary of the effects.

Fires can produce some substantial effects on the streamflow regime of both small streams and rivers. Tables 5.1 to 5.6 shows that fires can affect annual and seasonal water yield, peakflows and floods, baseflows, and timing of flows. Adequate baseflows are necessary to support the continued existence of many wildlife populations. Water yields are important because many forest, scrubland, and grassland watersheds function as municipal water supplies. Peakflows and floods are of great concern because of their potential impacts on human safety and property. Next to the physical destruction of a fire itself, postfire floods are the most damaging aspect of fire in the wildland environment. It is important that resource specialists and managers become aware of the potential of fires to increase peakflows.

Following wildfires, flood peak flows can increase dramatically, severely affecting stream physical conditions, aquatic habitat, aquatic biota, cultural resources, and human health and safety. Often, increased flood peak flows of up to 100 times those previously recorded, well beyond observed ranges of variability in managed watersheds, have been measured after wildfires. Potentials exist for peak flood flows to jump to 2,300 times prewildfire levels. Managers must be aware of these potential watershed responses in order to adequately and safely manage their lands and other resources in the postwildfire environment.

Table 5.7—A summary of the changes in hydrologic processes produced by wildland fires.

Hydrologic process	Type of change	Specific effect
1. Interception	Reduction	Moisture storage smaller Greater runoff in small storms Increased water yield
2. Litter storage of water	Reduced	Less water stored (0.05 in/in or 0.5 mm/cm litter) Overland flow increased
3. Transpiration	Temporary elimination	Streamflow increase Soil moisture increased
4. Infiltration	Reduced	Overland flow increased Stormflow increased
5. Streamflow	Changed	Increased in most ecosystems Decreased in snow systems Decreased in fog-drip systems
6. Baseflow	Changed	Decreased (less infiltration) Increased (less evapotranspiration) Summer low flows (+ and -)
7. Stormflow	Increased	Volume greater Peakflows larger Time to peakflow shorter Flash flood frequency greater Flood levels higher Stream erosive power increased
8. Snow accumulation	Changed	Fires <10 ac (<4 ha), increased snowpack Fires >10 ac (> 4 ha), decreased snowpack Snowmelt rate increased Evaporation and sublimation greater

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Chapter 6: Water Quality

Introduction

Increases in streamflow discharges following a fire can result in little to substantial effects on the physical, chemical, and biological quality of the water in streams, rivers, and lakes. The magnitude of these effects is largely dependent on the size, intensity, and severity of the fire, and on the condition of the watershed at the time of burning (fig. 6.1). Higher postfire streamflow discharges can result in an additional transport to stream channels or other water bodies of solid and dissolved materials that adversely affect the quality of water for human, agricultural, or industrial purposes. The most obvious effects are produced by suspended and bedload sediments. These components of water quality were introduced and discussed in chapter 2, and in this chapter they are referred to in the discussion on water quality relative to municipal water supply quality.

Fire affects water quality characteristics through the changes that the burning causes in the hydrologic cycle and streamflow regimes (see chapter 5). The effect of fire on water quality is the topic of this chapter.

Water Quality Characteristics and Standards

Water quality refers to the physical, chemical, and biological characteristics of water in reference to a particular use. Among the physical characteristics of interest to hydrologists and watershed managers are sediment concentrations, turbidity, and water temperature. Dissolved chemical constituents of importance include nitrogen (N), phosphorus (P), calcium (Ca), magnesium (Mg), and potassium (K). Some of these nutrients are adsorbed on organic and inorganic sediment particles. Bacteriological quality is also important if water is used for human consumption or recreation. The processes in the hydrologic cycle directly or indirectly affect the magnitude of soil erosion and, as a consequence, the transport and deposition of sediment in water and other physical, chemical, and biological quality characteristics that collectively determine the quality of water.

A *water quality standard* refers to the physical, chemical, or biological criteria or characteristics of water in relation to a specified use. It also includes the beneficial uses and antidegradation policy. For example, a water

Figure 6.1—Schoonover Fire just prior to crowning, 2002, Montana. (Photo by USDA Forest Service).

quality standard for irrigation is not necessarily acceptable for drinking water. Changes in water quality due to a watershed management practice could make the water flowing from the area unsuitable for drinking, but at the same time it could be acceptable for irrigation, fisheries, and other uses. In some instances there are laws or regulations to prevent water quality characteristics from becoming degraded to the point where a water quality standard is jeopardized (DeBano and others 1998, Landsberg and Tiedemann 2000, Brooks and others 2003). The main purpose of these laws and regulations is maintaining the quality of water for a possible, unforeseen, future use.

A major issue that hydrologist and watershed managers often confront is whether forest or rangeland fires will create conditions in stream, river, or lake waters that are outside of the established water quality standards and, as a consequence, will require remedial actions to bring the water within the standards. Water quality standards that have been established by the U.S. Environmental Protection Agency (1999) are the "benchmarks" for water quality throughout the United States. The most adverse effects from wildfires on water quality standards come from physical effects of the sediment and ash that are deposited into streams. However, stream chemistry parameters can be exceeded.

Several chemical constituents that are regulated by water quality standards are likely to be impacted by burning. Primary standards in the regulations of the U.S. Environmental Protection Agency cover nitrate-nitrogen (NO_3-N), nitrite-nitrogen (NO_2-N), and other substances that are not immediately associated with fire. Secondary standards apply to pH, sulfate (SO_4-S), total dissolved solids (TDS), chloride (Cl), iron (Fe), turbidity, and several other constituents. Secondary standards are also set for color and odor. Phosphate phosphorus (PO_4-P) can affect water quality because of its ability to affect the color and odor of water by accelerating the eutrophication process. That water quality standards can be impacted and exceeded by fire in some cases is also examined in this chapter.

Soil Erosion and Sedimentation Processes

Changes in the hydrologic cycle caused by fire can also affect the rate of soil erosion, the subsequent transport and deposition of the eroded soil as sediment, and the chemical characteristics that collectively determine the quality of water (DeBano and others 1998, Brooks and others 2003). Alterations that burning can cause in the hydrologic cycle, that in turn affect soil erosion, sedimentation, and water quality, are considered in chapter 5.

Soil erosion is the physical process of the force of raindrops or eddies in overland flow (surface runoff) dislodging soil particles, which are then transported by water or wind or the force of gravity (Dunne and Leopold 1978, Satterlund and Adams 1992, Brooks and others 2003). *Sedimentation* is the process of deposition of sediment in stream channels or downstream reservoirs or other point of use.

Increased soil erosion, or sediment, is often the most viable effect of a fire other than the loss of vegetation by burning. Maintaining a vegetative cover or a cover of litter and other organic material on the soil surface of a watershed is the best means of preventing excessive soil erosion rates. However, fire can cause the loss of these protective covers and in turn cause excessive soil erosion and soil lost from the burned site (Dunne and Leopold 1978, Satterlund and Adams 1992, Brooks and others 2003).

Natural rates of sedimentation are generally lower in high rainfall regions than in arid and semiarid regions (Brooks and others 2003). As a result of the infrequent "big storms" that are characteristic of arid and semiarid environments, sedimentation is often viewed as a discontinuous (unsteady) process where sediment runs from its source through a stream channel system with intermittent periods of storage (Wolman 1977, Baker 1990, Baker and others 1998). The disproportionate amount of sediment transported by these big storms makes it difficult to determine a "normal rate" of sedimentation on either undisturbed or burned watersheds in arid and semiarid regions (DeBano and others 1998).

Only a portion of the sediment is passed through and out of a watershed with a single storm event. Most of the sediment that is generated by a storm is deposited at the base of hillslopes, in floodplains following high overland flows, and within stream or river channels (DeBano and others 1998, Brooks and others 2003). The relationships between hillslope soil erosion and downslope and downstream sedimentation involve a complexity of channel processes and their dynamics, both of which are poorly understood. Nevertheless, the sediment that is eventually deposited into the channels following a fire changes the physical characteristics of the water flowing from the burned watershed.

Physical Characteristics of Water

Among the more important physical characteristics of postfire streamflow regimes of main interest to hydrologists and watershed managers are suspended sediment concentrations and turbidity and elevated streamflow temperatures (thermal pollution). Suspended sediment consisting of silts and colloids of soil materials impacts on water quality in terms of human, agricultural, and industrial uses of the water and aquatic organisms and their environments (see chapter 7). Elevated streamflow temperatures can also impact water quality characteristics and aquatic organisms and environments.

Sediment

Watersheds that have been severely denuded by a wildfire are often vulnerable to accelerated rates of soil erosion and, therefore, can yield large (but often variable) amounts of postfire sediment (fig. 6.2).

Figure 6.2—Stermer Ridge watershed burned at high severity during the Rodeo-Chediski Fire, Arizona, 2002. Note recently deposited sediment in the lower portion of the photo. (Photo by Daniel Neary).

Wildfires generally produce more sediment than prescribed burning. The large inputs of sediment into a stream following a wildfire can tax the transport capacity of the stream and lead to channel deposition (aggradation). However, prescribed burns by their design do not normally consume extensive layers of litter or accumulations of other organic materials. Hence, sedimentation is generally less than that resulting from a wildfire.

Suspended Sediment Concentrations and Turbidity—Suspended sediment concentrations and turbidity are often the most dramatic of water quality responses to fire. *Turbidity* is an expression of the optical property of water that scatters light (Dunne and Leopold 1978, Satterlund and Adams 1992, Brooks and others 2003). Turbidity reduces the depth to which sunlight can penetrate into water and, therefore, influences the rate of photosynthesis. Sediment concentrations are commonly expressed in parts per million (ppm) or milligrams per liter (mg/L) (International System of Units), while turbidity is measured in nephelometeric units. Postfire increases in suspended sediment concentrations and turbidity can result from erosion and overland flow, channel scouring because of the increased streamflow discharge, creep accumulations in stream channels, or combinations of all three actions after a fire.

Less is known about the effect of fire on turbidity than on the sedimentation processes. One problem contributing to this lack of information is that turbidity has been historically difficult to measure because it is highly transient, variable, inconsistent, and varies by instrument used. With the development of continuous turbidimeters or nephelometers, some suspended sediment estimates are now based on continuous turbidity measurements. These turbidity estimates must be translated to suspended sediment using turbidity-to-suspended sediment rating curves that are time consuming, carefully calibrated, site specific, and instrument specific.

Nevertheless, it has been observed that postfire turbidity levels in stream water are affected by the steepness of the burned watershed (table 6.1). The turbidity of overland flow from burned steep slopes in central Texas that had been converted from woodland to an herbaceous cover was higher than that of overland flow from burned slopes of lesser steepness (Wright and others 1976, 1982). Turbidity increases after fires are generally a result of the postfire suspension of ash and silt-to-clay sized soil particles in the water (fig. 6.3).

The primary standards for suspended sediment concentration and turbidity with respect to drinking water are written in terms of turbidity (U.S. Environmental Protection Agency 1999). However, only two of the studies reviewed by Landsberg and Tiedemann

Table 6.1—Water turbidity after fire or in combination with other treatments (Adapted from Landsberg and Tiedemann 2000).

Treatment	Habitat	Location	Treatment Pre	Treatment Post	Reference
			Jackson Turbidity Units		
Rx fire, pile, and burn	Juniper	Central Texas			Wright and others 1976
		3-4% slope	12	12	
		8-20% slope	20	53	
		37-61% slope	12	132	
Pile and burn	Juniper	Central Texas	12	162	Wright and others 1982
Pile, burn, seed			12	72	

Figure 6.3—Ash slurry flow in a ephemeral drainage after the Rodeo-Chediski Fire, Arizona, 2002, Apache-Sitgreaves National Forest. (Photo by Daniel Neary).

(2000) in their synthesis of the scientific literature on the effects of fire on drinking water used turbidity measured in nephelometric units as a measure of the suspended sediment concentrations of water (table 6.1). Other studies reported concentrations of suspended sediment in ppm or mg/L as the measure (table 6.2). While suspended sediment concentrations in ppm (mg/L) have been converted to turbidity in nephelometric units in studies of nutrient losses by soil erosion after wildfire, the relationship is site-specific. Beschta (1980) found that a relationship between suspended sediment concentrations and turbidity can be established for a specified watershed in some instances, but that the relationship differs significantly among watersheds. He suggested, therefore, that this relationship be established on a watershed-by-watershed basis.

The few postfire values for turbidity found in the literature (table 6.1) exceed the allowable water quality standard for turbidty (U.S. Environmental Protection

Table 6.2—Suspended sediment concentrations in streamflow after fire alone or fire in combination with other treatments (Adapted from Landsberg and Tiedemann 2000).

Treatment	Habitat	Location	Treatment Pre	Treatment Post	Reference
			ppm (mg/L)		
Wildfire	Taiga	Interior Alaska	10.6	6.0	Lotspeich and others 1970
Clearcut, slash burn	Douglas-fir	W. Oregon	2.0	150.0	Fredriksen 1971
Wildfire	Ponderosa pine	E. Washington	—	1,200.0	Helvey 1980
Pile and burn	Juniper	Texas	1.1	3.7	Wright and others 1982
Pile, burn, and seed	Juniper	Texas	1.0	3.7	Wright and others 1982
Prescribed fire	Loblolly pine	South Carolina	26.0	33.0	Douglass and Van Lear 1983
Wildfire	Mixed conifer	Montana	<3.0	32.0	Hauser and Spence 1998

Agency 1999). Therefore, Landsberg and Tiedemann (2000) recommended that the effect of fire on turbidity per se needs further investigation to better understand the processes involved. Because of the elevated suspended sediment concentrations after fire and fire-related treatments, it is difficult to imagine that some of these concentrations would not have produced turbidity above the permitted level for water supplies.

Sediment Yields—While the level of suspended sediment concentrations is a primary factor of environmental concern, sediment yield is important in estimating the sediment buildup in reservoirs or other impoundments (Wetzel 1983). The *sediment yield* of a watershed is the total sediment outflow from the watershed for a specified period of time and a defined point in the stream channel. Sediment yields are dependent mostly on the physical characteristics of the sediment, the supply of soil particles to a stream channel, the magnitude and rate of streamflow discharge, and the condition of the watershed. The values presented in tables 2.8 to 2.10 are indicative of the level of variability that can be expected in the changes in sediment yields following a fire. This variability generally reflects the interacting factors of geology, soil, topography, vegetation, fire characteristics, weather patterns, and land use practices on the impacted watershed. The higher values resulted from fires on steep slopes and areas of decomposing granite that readily erode. These higher sediment yields are sufficient in their magnitude to generate concern about soil impoverishment and water turbidity. The lower values were generally associated with flat sites and lower severity fires. Although the magnitudes shown are site specific, the changes in sediment transport presented reflect responses that can be expected from wildfire and prescribed burning.

Postfire sediment yields are generally the highest in the first year or so after burning, especially when the burned watershed has been exposed to large, high-intensity rainfall events immediately after the fire has exposed the soil surface. These sediment yields are indicative of the partial or complete consumption of litter and other decomposed organic matter on the soil surface, a reduction in infiltration, and a consequent increase in overland flow (DeBano and others 1998, Brooks and others 2003). Sediment yields typically decline in subsequent years as the protective vegetation becomes reestablished on the burned watershed.

Water Temperature

Water temperature is a critical water quality characteristic of many streams and aquatic habitats. Temperature controls the survival of certain flora and fauna in the water that are sensitive to water temperature. The removal of streambank vegetation by burning can cause water temperature to rise, causing thermal pollution to occur, which in turn can increase biological activity in a stream (DeBano and others 1998, Brooks and others 2003). Increases in biological activity place a greater demand on the dissolved oxygen content of the water, one of the more important water quality characteristics from a biological perspective.

There are no established national standards for the temperature of drinking water. However, under the Clean Water Act, States are required to develop water quality standards to protect beneficial uses such as fish habitat and water quality restoration. The U.S. Environmental Protection Agency provides oversight and approval of these State standards. Currently, about 86 percent of the national listings of waterbodies with temperature-impaired water quality are in the Pacific Northwest (Ice and others, in press). One of the problems with these standards is identifying natural temperature patterns caused by vegetation, geology, geomorphology, climate, season, and natural disturbance history. Also, increases in stream water temperatures can have important and often detrimental effects on stream eutrophication. Acceleration of stream eutrophication can adversely affect the color, taste, and smell of drinking water.

Severe wildfires can function like streamside timber clearcuts in raising the temperature of streams due to direct heating of the water surface. Increases up to 62 °F (16.7 °C) have been measured in streamflows following fire, and following timber harvesting and fire in combination (table 6.3). When riparian (streamside) vegetation is removed by fire or other means, the stream surface is exposed to direct solar radiation, and stream temperatures increase (Levno and Rothacher 1969, Brown 1970, Swift and Messner 1971, Gibbons and Salo 1973, Brooks and others 2003).

Another important aspect of the temperature issue is the increase in fish mortality posed by stream temperature increases. The main concerns relative to aquatic biota are the reduction in the concentrations of dissolved oxygen (O_2) that occurs with rising temperatures, fish pathogen activity, and elevated metabolic activity. All of these can impair the survivability and sustainability of aquatic populations and communities. Dissolved O_2 contents are affected by temperature, altitude, water turbulence, aquatic organism respiration, aquatic plant photosynthesis, inorganic reactions, and tributary inflow. When O_2 concentrations become less than 10 ppm (less than 10 mg/L), they create problems for salmonid fishes. Increases of 2 to 9 °F (1-5 °C), which are not a problem at sea level, become problematic for salmonids at high altitude. Warm water fishes can tolerate stream temperatures below 10 ppm (10 mg/L) and are not as easily impacted by O_2 concentration declines.

Table 6.3—Temperature increases in streamflow after fire alone and in combination with other treatments.

Location	Treatment	Buffer	Temperature increase		Reference
			°F	*(°C)*	
M221 CENTRAL APPALACHIAN BROADLEAF-CONIFER FOREST PROVINCE					
Pennsylvania	Clearcut	Yes	3.1	(1.7) mn[1]	Lynch and Corbett 1990
North Carolina	Clearcut, farm	No	20.9	(11.6) mx	Swift and Messner 1971
	Clearcut	No	5.9	(3.3) mx	
	Understory cut	No	2.0	(1.1) mx	
M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE					
Oregon	Clearcut	Yes	14.0	(7.8) mn	Brown and Krygier 1970
Oregon	Patch cut	Yes	0.0	(0.0)	Hall and others 1987
	Clearcut	No	30.1	(16.7) d	
Oregon	Clearcut Rx burn	No	13.0	(7.2) sm	Levno and Rothacher 1969
Oregon	Wildfire: 26% slope	No	18.0	(10.0) mx	Amaranthus and others 1989
	Wildfire: 54% slope	No	5.9	(3.3) mx	
Washington	Wildfire	No	18.0	(10.0) sm	Helvey 1980
M331 SOUTHERN ROCKY MOUNTAINS STEPPE-WOODLAND-CONIFER PROVINCE					
Wyoming	Wildfire*	No	18.0	(10.0) mx	Hungerford and others 1991

[1] sm = summer mean temperature, mn = mean temperature, d = daily temperature, mx = maximum.
* = Yellowstone Fires 1988.

pH of Water

The *pH* of water at a point in time is an indication of the balance of chemical equilibria in a water body. Its level affects the presence of some chemicals in the water. The pH of water can be affected by ash depositions immediately after a fire. In the first year after fire, increased pH values of the soil (Wells and others 1979, DeBano and others 1998, Landsberg and Tiedemann 2000) can also contribute to increased values of streamflow pH. A secondary drinking water quality standard for pH is 6.5 to 8.5 (U.S. Environmental Protection Agency 1999). In the investigations reviewed by Landsberg and others (1999), there was only one study that reported pH values outside the U.S. Environmental Protection Agency standards (table 6.4). During the first 8 months after the Entiat Fires in eastern Washington, Tiedemann (1973) detected transient pH values up to 9.5 in streamflow water and, 2 days after fertilizer application, a transient pH value of 9.2. These latter values are generally reflective of toxic limits in the water.

Chemical Characteristics of Water

Watershed-scale studies provide an integrated view of the effects of fire on chemical (ionic) concentrations and losses. Some investigators have reported little effects of burning on the chemical concentrations in water following burning of a watershed and, therefore, have concluded that reported increases in streamflow resulting from fire-caused transpirational reductions had likely masked concentration effects (Helvey and others 1976, Tiedemann and others 1978, Gottfried and DeBano 1990, DeBano and others 1998). Other investigators observed higher concentrations of some chemicals in streamflow from burned watersheds (Snyder and others 1975, Campbell and others 1977). However, these elevated concentrations often return to prefire levels after the first flush of flow.

Dissolved Chemical Constituents

The main sources of dissolved chemical constituents (nutrients) in the water flowing from watersheds are geologic weathering, decompositions of photosynthetic products into inorganic substances, and large storm events. Vegetative communities accumulate and cycle large quantities of nutrients in their biological role of linking soil, water, and atmosphere into a biological continuum (Tiedemann and others 1979, DeBano and others 1998, Brooks and others 2003). Nutrients are cycled in a largely orderly (tight) and often predictable manner until a disturbance alters the form of their distribution. One such disturbance is fire.

The effects of fire on the nutrient capital (status) of a watershed ecosystem are largely manifested by a rapid mineralization and dispersion of plant nutrients

Table 6.4—The pH of water after fire alone or in combination with other treatments (Adapted from Landsberg and Tiedemann 2000).

Treatment	Habitat	Location	Treatment Pre	Treatment Post	Reference
			- - - - - *pH* - - - - -		
Wildfire	Ponderosa pine and Douglas-fir	Washington	—	7.2-8.5	Tiedemann 1973
Wildfire + N			—	7.1-9.5	
Wildfire + N	Mixed conifer	California	6.2-7.0	6.7-7.0	Hoffman and Ferreira 1976
Pile burn	Juniper	Texas			Wright and others 1976
		Slopes			
		3-4%	7.3	7.3	
		8-20%	7.6	7.7	
		37-61%	7.4	7.7	
Wildfire	Pine, spruce	Minnesota	6.2	6.3	Tarapchak and Wright 1977
Rx fire	Ponderosa pine	Arizona	6.2	6.4	Sims and others 1981
Pile burn	Juniper	Texas	7.1	7.3	Wright and others 1982
Slash burn	Hemlock/cedar	British Columbia	6.8	7.8	Feller and Kimmins 1984
Wildfire	Mixed conifer	Wyoming	7.4	7.5	Lathrop 1994

from an intrabiotic to an extrabiotic state (Grier 1975, Tiedemann and others 1979, DeBano and others 1998, Brooks and others 2003). Part of the plant- and litter-incorporated N, P, K, Ca, Mg, copper (Cu), Fe, manganese (Mn), and zinc (Zn) are volatilized and, through this process, evacuated from the system. Metallic nutrients such as Ca, Mg, and K are converted into oxides and deposited as ash layers on the soil surface. Oxides are low in solubility until they react with carbon dioxide (CO_2) and water in the atmosphere and, as a result, are converted into bicarbonate salts. In this form, they are more soluble and vulnerable to loss through leaching or overland flow than they are as oxides, or they are incorporated into plant tissues or litter.

In a postfire situation where, compared to prefire conditions, there are less vegetative cover and lower accumulations of litter and other organic materials, the result is an increase in susceptibility to nutrient loss from a watershed through erosion. With a reduced vegetative cover, soil-plant cycling mechanisms reduce nutrient uptake, further increasing the potential nutrient loss by leaching. Responses of ionic concentrations of the more important nutrients to burning are discussed in the following paragraphs.

Nitrogen

NO_3-N, NH_4-N, and organic-N are the nitrogen forms most commonly studied as indicators of fire disturbance. Most of the attention of hydrologists and watershed managers relative to water quality responses to fire focuses on NO_3-N because it is highly mobile. The potential for increased NO_3-N in streamflow after burning is attributed mainly to accelerated mineralization and nitrification (Vitousek and Melillo 1979, Covington and Sackett 1986, 1992, DeBano and others 1998) and reduced plant demand (Vitousek and Melillo 1979). This increase results from the conversion of organic N to available forms (Kovacic and others 1986), mineralization (Covington and Sackett 1992, Ojima and others 1994), or mobilization by microbial biomass through the fertilizing effect of ash nutrients and improved microclimate (Koelling and Kucera 1965, Hulbert 1969, Ojima and others 1995). These postfire effects are short lived, however, usually lasting only a year or so (Kovacic and others 1986, Monleon and others 1997).

The response of NO_3-N to burning is varied. Some investigators have found no significant change in the postfire levels of NO_3-N in streamflows, while others report increases of NO_3-N in either the soil solution or streamflow (Hibbert and others 1974, Tiedemann and others 1979). Examples of increases in NO_3-N (table 6.5) with fire and followup applications of herbicides causing the largest increases are shown in most studies of forest disturbances such as fire.

The most striking response of NO_3-N concentration in streamflow to fire (and the only case not related to herbicides where the primary water quality standard was exceeded) was observed in southern California, where N loadings from atmospheric deposition are relatively high, and the frequent wildfires in the chaparral shrublands are often high in their fire severities (Riggan and others 1994). Severe burning of a watershed in this Mediterranean-type resulted in a maximum NO_3-N level of 15.3 ppm (mg/L) in streamflow compared to 2.5 ppm (mg/L) in streamflow from an unburned control watershed. Maximum concentration for a moderately burned watershed was 9.5 ppm (mg/L). These results are likely to represent an "unusual response" because the watersheds studied were subject to a chronic atmospheric deposition of pollutants. Regardless of the treatment or treatment combinations on these watersheds, levels of NO_3-N in the streamflows were well below maximum allowable concentrations in most other studies. Beschta (1990) reached the same conclusion in his assessment of streamflow NO_3-N responses to fire and associated treatments.

Table 6.5—Effect of forest disturbances on maximum NO_3-N levels in streamflow (Adapted from Neary and Hornbeck 1994, Neary and Michael 1997, Landsberg and Tiedemann 2000).

Location	Forest type	Treatment	Maximum NO_3-N	Reference
			ppm (mg/L)	
1. Harvesting				
M212 ADIRONDACK-NEW ENGLAND MIXED FOREST PROVINCE				
New Hampshire	Hardwoods	Clearcut	6.1	Hornbeck and others 1987
M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE				
Oregon	Douglas-fir	Clearcut	2.1	Brown and others 1973
2. Herbicides				
M212 ADIRONDACK-NEW ENGLAND MIXED FOREST PROVINCE				
New Hampshire	Hardwoods	Cut, herbicide	17.8	Aubertin and Patric 1974
313 COLORADO PLATEAU SEMIDESERT PROVINCE				
Arizona	Chaparral	Herbicide	15.3	Davis 1984
3. Fire				
231 SOUTHEASTERN MIXED FOREST PROVINCE				
South Carolina	Loblolly pine	Rx fire	<0.1	Richter and others 1982
South Carolina	Loblolly pine	Rx fire	<0.1	Douglass and Van Lear 1983
M242 CASCADE MIXED-CONIFER-MEADOW FOREST PROVINCE				
Oregon	Douglas-fir	Clearcut, burn	0.6	Fredriksen and others 1975
Washington	Ponderosa	Wildfire	<0.1	Tiedemann 1973
Washington	Ponderosa	Wildfire + N	0.3	
Washington	Douglas-fir			
M262 CALIFORNIA COASTAL RANGE WOODLAND-SHRUB-CONIFER PROVINCE				
California	Chaparral	Unburned	2.5	Riggan and others 1994
		Moderate burn	9.5	
		Severe burn	15.3	
	Chaparral	Wildfire	<0.1	Taylor and others 1993
313 COLORADO PLATEAU SEMIDESERT PROVINCE				
Arizona	Chaparral	Herbicide, burn	18.4	Davis 1987
		Rx fire alone	12.0	
M313 AZ-NM MOUNTAINS SEMIDESERT-WOODLANDS-CONIFER PROVINCE				
Arizona	Ponderosa	Wildfire	0.2	Campbell and others 1977
Arizona	Ponderosa	Rx fire	<0.1	Gottfried and DeBano 1990
M313 SOUTHERN ROCKY MOUNTAIN STEPPE-CONIFER-MEADOW PROVINCE				
Idaho	Douglas-fir	Slash burn	<0.1	Clayton and Kennedy 1985
M333 NORTHERN ROCKY MOUNTAIN STEPPE-CONIFER-MEADOW PROVINCE				
Montana	Lodgepole	Wildfire	0.3	Hauer and Spencer 1998

Tiedemann (1973) and Tiedemann and others (1978) show that fertilizer application following fire resulted in higher concentrations of NO_3-N in streamflows than fire alone. It is probably safe to conclude, however, that neither fire nor fertilization after fire will have adverse effects on NO_3-N in drinking water (Tiedemann and others 1978, Landsberg and others 1999, Scatena 2000).

NO_2-N was reported by itself rather than in combination with $N0_3$-N in two studies on the effects of fire on the chemical concentrations of streamflows in California and Washington. At the Lexington Reservoir in southern California, Taylor and others (1993) found NO_2-N levels of 0.03 ppm (mg/L) in streamflows after a wildfire occurred in a subwatershed that drains into the reservoir, while control levels were 0.01 ppm (mg/L). Tiedemann (1973) reported that NO_2-N concentrations were below that of analytical detection after a wildfire occurred in eastern Washington.

Other studies on the effects of wildfire or prescribed burning on both N and other dissolved chemical constituents are summarized in the following paragraphs. The observed elevated postfire ionic concentrations observed in these studies returned to prefire levels after the first flush of flows following the fire in most cases.

The maximum concentrations of NO_3-N in streamflows increased from less than 0.016 to 0.56 ppm (mg/L) in the 3 years after a wildfire destroyed the mixed conifer forest cover on a 1,410 acre (564 ha) watershed in eastern Washington (Tiedemann and others 1978). These increases appeared to be a result of increased nitrification. Concentrations of total P in the streamflow from the burned watershed were 1.5 to three times greater than those from an undisturbed watershed. Despite the lack of prefire information, elevated levels of these magnitudes indicate that the wildfire significantly affected the P levels in streamflow, at least in the short term. The combined concentrations of Ca, Mg, K, and Na in streamflow before the fire ranged from 12.0 to 14.9 ppm (mg/L). The concentrations declined to a range of 7.4 to 10.5 ppm (mg/L) in the second postfire year because of the dilution caused by increased streamflow. These losses were insignificant relative to the total capital of these nutrients, however.

The chemical quality of streamflows from two small watersheds that originally supported ponderosa pine forests in northern Arizona was not greatly affected by a wildfire (Campbell and others 1977). One watershed of 20.2 acres (8.1 ha) was severely burned while a smaller watershed of 10 acres (4 ha) was moderately burned. While concentrations of Ca, Mg, and K increased with the first postfire streamflow events from these watersheds, concentrations of these ions decreased rapidly in subsequent flow events. Sodium (Na) concentrations were largely unaffected by the fire or the observed changes in streamflow discharges. Combined organic-inorganic N concentrations also increased in the initial postfire streamflow event and then quickly decreased to the level that was found in streamflow from the unburned watershed.

The magnitude of stream water quality changes for N after prescribed fire are normally less than those observed after wildfires. It is unlikely that prescribed burning would consume as much of the litter and other organic materials, understory herbaceous vegetation, or overstory trees as severe wildfires (McNabb and Cromack 1990, DeBano and others 1998). Stream chemistry responses to prescribed fire in an undisturbed 1,163 acre (470 ha) Southwestern ponderosa pine forest watershed (Gottfried and DeBano 1990) support the speculation of minimal changes in N following a prescribed fire. Surface fuels were burned on 43 percent of this watershed, and 5 percent of the trees were killed. The prescribed fire resulted in relatively small increases in NO_3-N in the streamflow, the levels of which did not approach the primary water quality standard. Measures taken to protect streams and riparian corridors with unburned buffers could also minimize effects of fire on stream chemistry.

Several studies show that increased streamflow discharges and increased concentrations of N in the streamflow from burned areas can cause an accelerated loss of N from watershed lands in the short term. While these losses in N have not been referenced to the total N capital on a watershed, these losses likely pose little threat to continued onsite productivity (DeBano and others 1998, Brooks and others 2003).

The primary drinking water standard for NO_3-N is 10 ppm (mg/L), and the standard for NO_2-N is 1 ppm (m/L). There is no standard for dissolved organic N, NH_4-N, or urea-N in drinking water. The combined concentrations of these N forms seldom exceed 1 ppm (mg/L), and dissolved organic N is usually the most abundant form.

Phosphorus

Phosphorus is present in several forms in soil solution and streamflow. These are reactive orthophosphate (inorganic phosphate), dissolved complex organic phosphate, particulate organic phosphate, and other inorganic forms (Ice 1996). Total phosphate (PO_4-P) is reported as total P in most of the studies of the P response to fire. PO_4-P is not as readily leached as NO_3-N because it complexes with organic compounds in the soil (Black 1968). Studies of soil leachates have reported increased levels of total phosphorous due to burning, indicating accelerated mobilization of phosphorous after burning (McColl and Grigal 1975, Knighton 1977). PO_4-P concentrations in streamflows prior to fire can range from 0.007 to 0.17 ppm (mg/L)

(Longstreth and Patten 1975, Hoffman and Ferreira 1976, Wright and others 1976, Tiedemann and others 1978, Tiedemann and others 1988). After wildfire, prescribed burning, or the clearcutting of timber followed by broadcast burning of the slash, Longstreth and Patten (1975) reported that PO_4-P concentrations stayed the same or increased only as high as 0.2 ppm (mg/L).

Phosphorus concentrations in overland flow from the hillslopes of a watershed can increase as a result of burning, although these increases are not always sufficient to alter the quality of the watershed's streamflow regime (Longstreth and Patten 1975, Gifford and others 1976). There is no established standard for PO_4-P in drinking water. Phosphorus can be limiting in aquatic habitats, but once in the water, it is taken up quickly by aquatic organisms, especially algae.

Sulfur

Sulfate (SO_4-S) is relatively mobile in soil-water systems (Johnson and Cole 1977). Although not as well studied as those for N, the mineralization processes for S are essentially similar. Observed levels of SO_4-S in the streamflow from most wildland watersheds are inherently low. Prefire concentrations of the ion can range from as low as 1.17 ppm (mg/L) to as high as 66 ppm (mg/L), while postfire values range from 1.7 to 76 ppm (mg/L). All of the SO_4-S concentrations reported by Landsberg and Tiedemann (2000) were below the secondary water quality standard for drinking water of 250 ppm (mg/L) (U.S. Environmental Protection Agency 1999).

Chloride

Chloride ion (Cl) responses to fire have been documented in several studies (Landsberg and Tiedemann 2000). All responses were significantly lower (less than 5 ppm, or mg/L) than the water quality standard of 250 ppm (mg/L) established by the U.S. Environmental Protection Agency (1999). Concentrations of some natural sources of Cl (such as geothermal areas) are reportedly larger than those produced by wildfires.

Bicarbonate

Bicarbonate ions in soil solutions and streamflows are often increased as a consequence of burning. The bicarbonate ion represents the principal anion in soil solution, the end product of root respiration and a product of oxide conversion after fire (McColl and Cole 1968, Tiedemann and others 1979, DeBano and others 1998). Concomitant fluctuations of bicarbonate and cation concentrations indicate that bicarbonate is a main carrier of cations in the soil solution (Davis 1987).

Total Dissolved Solids

In their synthesis of the scientific literature on the quality of drinking water originating from natural ecosystems, Landsberg and Tiedemann (2000) found only two studies that reported concentrations of total dissolved solids (TDS), although the investigators in other studies have measured some of the constituents of TDS but not TDS per se. Hoffman and Ferreira (1976) detected TDS concentrations of about 11 ppm (mg/L) in the streamflow from an unburned area in Kings Canyon National Park of California and 13 ppm (mg/L) in the streamflow from an adjacent burned area, which had been a mixed conifer and shrub stand. Lathrop (1994) found that Yellowstone Lake and Lewis Lake in Yellowstone National Park had prefire TDS concentrations of 65.8 and 70 ppm (mg/L), respectively, and that these concentrations were similar to those observed after the fires. These values were significantly below the secondary standard of TDS for drinking water of 500 ppm (mg/L) recommended by the U.S. Environmental Protection Agency (1999).

Nutrients and Heavy Metals

Heavy metals are a growing concern in some forested areas of the United States. Particular concern exists about the release of heavy metals to the air and eventually to streams by increased prescribed fire programs and large wildfires (Lefevre, personal communication). Information on this aspect of water quality and fire use and management is relatively scarce.

One potentially important source of nutrient and heavy metal loss from a watershed that is often ignored is that transported by sediment particles (Gifford and Busby 1973, Fisher and Minckley 1978, Gosz and others 1980, Brooks and others 2003). Sediment has been reported to transport relatively high levels of nutrients and heavy metals (Angino and others 1974, Potter and others 1975), although most of these investigators have looked at large river basins consisting of numerous combinations of bedrock and vegetation.

Few studies have focused directly on the effects of fire on nutrient and heavy metal losses that are transported by sediment from smaller upland watersheds. One study reported that sediment losses of N, P, and ions in streamflow from burned watersheds in chaparral shrublands of southern California chaparral can substantially exceed those lost in solution (DeBano and Conrad 1978). Nitrogen and P losses of 13.5 and 3.0 lb/acre (5.1 and 3.4 kg/ha), respectively, were found in sediments, as compared to only trace levels in solution. Furthermore, losses of Ca, Mg, Na, and K in solution were only one-fourth of the losses in sediment. However, sediment and solution losses of N, P, K, Mg, Ca, and Na were only a small fraction of the total prefire nutrient capital of plants, litter accumulations, and the upper 4 inches (10 cm) of soil.

Biological Quality of Water

Accumulations of litter and of other decomposed organic matter on the soil surface often function as a filter that removes bacteria and other biological organisms from overland flow. Rainfall-induced runoff and snowmelt that percolates through a litter layer or strip of organic matter can contain fewer bacteria than water that had not passed through the strip (DeBano and others 1998, Brooks and others 2003). It follows, therefore, that the destruction of this layer or strip by burning might result in higher concentrations of bacterial and other biological organisms flowing overland to a stream channel.

Fire Retardants

Fire retardants are frequently used in the suppression of wildfires. Although their effects on the soil-water environment are not a direct effect of fire, their use in the control of wildfires can produce adverse environmental impacts. A brief discussion of fire retardant effects on water quality is provided here to acquaint the reader with the topic. More detailed reviews and studies have been completed by Labat and Anderson Inc. (1994), Adams and Simmons (1999), Kalabokidis (2000), and Gimenez and others (2004). The main environmental concerns with fire retardant use are: (1) effects on water quality and aquatic organisms, (2) toxicity to vegetation, and (3) human health effects.

Ammonium-based fire retardants (diammonium phosphate, monoammonium phosphate, ammonium sulfate, or ammonium polyphosphate) play an important role in protecting watershed resources from destructive wildfire (fig. 6.4; table 6.6). However, their use can affect water quality in some instances, and they can also be toxic to aquatic biota (table 6.7; see also chapter 7).

Nitrogen-containing fire retardants have the potential to affect the quality of drinking water, although the research on the applications of these retardants to streams has largely focused on their impacts on aquatic environments (Norris and Webb 1989). For example, in an in vitro study to determine the toxicity of some retardant formulations to stream organisms, McDonald and others (1996) evaluated the impacts of two nonfoam retardants containing SO_4-S, PO_4-P, and ammonium compounds (Fire-trol GST-R and Phos-Chek D75-F), a retardant containing ammonium and phosphate compounds (Fire-Trol LCG-R), and two foam suppressant compounds that contained neither S, P, nor ammonium compounds (Phos-Chek WD-881 and Silv-Ex). These investigators found that concentrations of NO_3-N in water rose from (0.08) to 3.93 ppm (mg/L) after adding the nonfoam retardants. They also discovered that NO_2-N reached concentrations as high as 33.2 ppm (mg/L), well above the primary water quality standard of 1 ppm (mg/L). However, the solutions they tested were much less concentrated than that which is used in firefighting.

The main chemical of concern in streams 24 hours after a retardant drop is ammonia nitrogen (NH_3 + NH_4^+; table 6.6). Un-ionized ammonia (NH_3) is the principal toxic component to aquatic species. The distances downstream in which potentially toxic conditions persist depend on stream volume, the number of retardant drops, and the orientation of drops to the stream long-axis (table 6.8). While concentrations of

Figure 6.4—Fire retardant drop from an contract P-3 Orion, San Bernardino National Forest, California. (Photo by USDA Forest Service).

Table 6.6—Composition of forest fire retardants (Adapted from Johnson and Sanders 1977).

Trade name and chemical components	Amount of component: Concentrate	Amount of component: Field mix
	- - - - - - - ppm(mg/L) - - - - - -	
FIRE-TROL 100		
$(NH_4)_2SO_4$	635,000	178,624
NH_4	173,353	48,764
FIRE-TROL 931		
Ammonium polyphosphate	930,000	268,122
NH_4	119,235	34,376
PHOS-CHEK 202		
$(NH_4)_2HPO_4$	833,500	114,085
NH_4	241,372	31,168
PHOS-CHEK 259		
$(NH_4)_2HPO_4$	919,500	155,358
NH_4	251,207	42,443

Table 6.7—Fire retardant lethal levels for aquatic organisms (Adapted from Johnson and Sanders 1977).

Organism	24 hour LC50 concentration[1] FIRE-TROL 100	FIRE-TROL 931	PHOS-CHEK 202	PHOS-CHEK 259
	(ppm) mg/L			
Coho salmon				
Yolk-sac fry	160	> 500	175	> 200
Swim-up fry	1,100	1,050	210	175
Fingerling	> 1,500	1,050	320	250
Rainbow trout				
Yolk-sac fry	158	> 500	140	> 200
Swim-up fry	900	780	210	175
Fingerling	> 1,000	> 1,000	230	175
Bluegill	> 1,500	> 1,500	840	600
Fathead minnow	> 1,500	> 1,500	820	470
Largemouth bass	> 1,500	> 1,500	840	720
Scud	> 100	> 100	100	> 100

[1]LC50 concentration is the concentration needed to kill 50 percent of the exposed individuals in a given time period (24 hr).

$NH_3 + NH_4^+$ can reach 200 to 300 ppm (mg/L) within 164 to 328 feet (50-100 m) below drop points, toxic levels may persist for over 3,280 feet (1,000 m) of stream channel.

Inadvertent applications of fire retardants into a stream could have water quality consequences for NO_3-N, SO_4-S, and possibly trace elements. However, information about these potential effects of retardants on water quality is limited.

Another source of concern is fire retardants that contain sodium ferrocyanide (YPS) (Little and Calfee 2002). Photo-enhanced YPS has a low lethal concentration in water for aquatic organisms. Fortunately, YPS, like the other retardant chemicals, is adsorbed onto organic and mineral cation exchange sites in soils. Thus, its potential for leaching out of soils is reduced.

Gimenez and others (2004) concluded that the most significant environmental impact of fire retardants is the toxic effect on aquatic organisms in streams. They noted that the amount of fire retardant used and its placement on the landscape are the two main factors determining the degree of environmental impact. Thus, placement planning and operational control of fire retardant aircraft are critical for minimizing impacts on streams and lakes and their biota.

Table 6.8—Fish mortality related to a hypothetical fire retardant drop orientation (Adapted from Norris and Webb 1989).

Angle to long axis of a stream: Degrees	Position	Distance for 100% mortality: Standard drop	2 standard drops
		ft (m)	
90	Perpendicular	164 (50)	1,575 (480)
67		164 (50)	1,837 (560)
45		328 (100)	3,281 (1,000)
22		787 (240)	>3,281 (>1,000)
0	Over stream	3,281 (1,000)	>3,281 (>1,000)

Rodeo-Chediski Fire, 2002: A Water Quality Case History

The Rodeo-Chediski Fire of 2002 was the largest wildfire in Arizona's history (Neary and Gottfried 2002, Ffolliott and Neary 2003). This fire damaged, destroyed, or disrupted the hydrologic functioning and ecological structure of the ponderosa pine forest ecosystems at the headwaters of the Salt River, a major river supplying the city of Phoenix's main water supply reservoir, Lake Roosevelt. The Rodeo-Chediski Fire was actually two fires that ignited on lands of the White Mountain Apache Nation and merged into one. Arson was the cause of the Rodeo Fire, which began a few miles from Cibecue, a small streamside village, on June 18, 2002. The Chediski Fire was set on the Reservation as a signal fire by a seemingly lost person a few days later. This second fire spread out of control, moving toward and eventually merged with the ongoing and still out of control Rodeo Fire. Burning northeastwardly, the renamed Rodeo-Chediski Fire then burned onto the Apache-Sitgreaves National Forest, along the Mogollon Rim in central Arizona, and into many of the White Mountain tourist communities scattered along the Mogollon Rim from Heber to Show Low. Over 30,000 local people were eventually forced to flee the inferno.

The Rodeo-Chediski Fire had burned 276,507 acres (111,898 ha) of Apache land, and the remainder of the total of 467,066 acres (189,015 ha) were on the Apache-Sitgreaves National Forest. Nearly 500 buildings were destroyed; more than half of the burned structures were houses of local residents or second homes of summer visitors. Rehabilitation efforts began immediately after the fire was controlled and after it was declared safe to enter into the burned area. Two BAER teams operated out of the White Mountain Apache Tribe headquarters at White River, AZ, and the other for the Apache-Sitgreaves National Forest at Show Low, AZ. Watershed protection was of prime importance because of the municipal watershed values of these lands.

Culverts were removed from roads to help mitigate the anticipated flash flooding that is often caused by high-intensity, short-duration monsoonal rainfall events that commonly occur in Arizona from early July through August and, occasionally, into September. Temporary detention dams and other diversions were constructed to divert intermittent water flows initiated by these storms away from critical infrastructures. In a major rehabilitation activity, helicopters ferried bales of straw to burned sites susceptible to erosion, and the straw was spread onto the ground to alleviate the erosive impact of the monsoonal rain. Seeding of rapidly established grasses and other herbaceous plants accompanied this rehabilitative activity, which continued into the late summer and early autumn.

Water-quality constituents of primary interest to hydrologists, land managers, and watershed managers in Southwestern ecosystems are sediment concentrations and dissolved nutrients, specifically nitrogen and phosphorus (DeBano and others 1996). In July, monsoon thunderstorms initiated storm runoff from the wildfire-burned area. While there was not an opportunity to collect samples to determined the quality characteristics of the water flows from several watersheds within the Rodeo-Chediski Fire area (Stermer Ridge watersheds; Ffolliott and Neary 2003), samples were taken of the ash and sediment-laden streamflow farther downstream where the Salt River enters into Lake Roosevelt. The storm runoff streamflows contained large amounts of organic debris, dissolved nutrients (including nitrogen, phosphorus, and carbon), and other chemicals that were released by the fire. Some of the elevated concentrations of nitrogen and phosphorus may have originated from the fire retardants dropped to slow the advancement of the fire.

The sediment- and organic-rich water significantly increased the flow of water into the Salt River, the major tributary to the Theodore Roosevelt Reservoir, a primary source of water for Phoenix and its surrounding metropolitan communities. The Salt River Project provides drinking and irrigation water, as well as power, to 2 million residential, business, and industrial customers in a 2,900 mile^2 (7,511 km^2) service area in parts of Maricopa, Gila, and Pinal Counties, Arizona (Autobee 1993). Water is furnished primarily by the Salt and Verde Rivers, which drain a watershed area of 13,000 miles^2 (33,670 km^2). Four storage reservoirs on the Salt River form a continuous chain of lakes almost 60 miles (96 km) long. An important supplemental supply is obtained from well pumping units. Capacity assigned to flood control is 556,000 acre-feet (685.8 million m^3). Total storage capacity of Salt River reservoirs is more than 2.4 million acre-feet (3.0 trillion m^3). Total hydroelectric generating capacity is 232 megawatts, including power from pumped storage units.

What made this situation more serious than would be expected were the size of the Rodeo-Chediski Fire and the critically low level of the drought-impacted Roosevelt reservoir, which was less than 15 percent of its capacity at the time of the fire. The reservoir has a drainage area of 5,830 miles^2 (3.7 million acres or 1.5 million ha) of which about 12 percent was burned in the Rodeo-Chediski Fire. Even with a small percent burned, there was a concern that the flow of ash and debris might threaten the aquatic life inhabiting the reservoir, leaving it lifeless for months to come. However, this dire situation failed to materialize. Even though some fish died upstream of the reservoir and a few carcasses showed up at the diversion dam above Roosevelt Dam, the reservoir water body itself suffered little permanent environmental damage.

The largest pulses in water quality parameters were for suspended sediment, conductivity, and turbidity. Nutrient levels (particularly P and N) in the water shot off the chart in the first few days of monsoon-induced storm runoff (U.S. Geological Survey 2002, Tecle, personal communication; table 6.9) but fell quickly, and therefore, the large algae blooms that were predicted to form and consume much of the water's oxygen did not form. The postfire debris in the water was never a health risk to people as most of the pollutants were easily removed at water treatment plants. The outcome would be different in a situation where a much larger percentage of a municipal watershed is burned by a wildfire.

Management Implications

A number of management considerations relate to water quality and the use or management of prescribed fires and wildfires. These considerations are tied to both Federal and State regulations and laws such as the National Forest Management Act of 1976, the Clean Water Act of 1972 as amended, the Code of Federal Regulations (36 CFR 219.13), and State laws. The purpose of these laws and regulations is to conserve soil and water resources, minimize serious or long-lasting hazards from flood, wind, wildfire, erosion, or other natural forces, and to protect streams, lakes, wetlands, and other bodies of water. The approach of these regulations is to use Best Management Practices to achieve water quality goals and protection. Special attention is given to land and vegetation in recognizable areas dominated by riparian vegetation.

Several sources of conservation measures guidelines relate to the use of prescribed fire (table 6.10) and the management of wildfire (table 6.11) to maintain water quality. These include the U.S. Environmental Protection Agency (1993), the USDA Forest Service (1988, 1989a,b, 1990a), California Department of Forestry (1998), Georgia Forestry Association (1995), to mention just a few.

In some instances conservation measures may be prohibitive or at least impractical because of local terrain conditions, hydrology, weather, fuels, or fire behavior. The ground rule is to use common sense when applying these guidelines. If conservation measures or actual State-mandated Best Management Practices are to be used, it is important to clearly communicate objectives and goals at briefings and on operational period fire plans. Assistance from staff hydrologists and soil scientists is also crucial in successfully applying these water quality protection guidelines.

Table 6.9—Water quality at the Salt River entrance to Lake Roosevelt, Arizona, Salt River Basin, before and after the Rodeo-Chediski Fire, 2002, compared to existing water quality standards (U.S. Geological Survey 2002).

Water quality parameter	Unit	Drinking water standard	Prefire level	Postfire peak
Arsenic	mg/L	0.05	0.05-0.350	0.685
Bicarbonate	mg/L	380.0	80-250	312.0
Calcium	mg/L	50.0	30-85	144.0
Chloride	mg/L	250.0	100-1,100	2,110.0
Copper	mg/L	1.0	<1	0.375
Iron	mg/L	0.3	<0.3	90.6
Lead	mg/L	0.05	0.0-0.10	0.69
Magnesium	mg/L	20.0	8-40	45.0
Phosphorus	mg/L	0.1	0.0-0.1	39.0
Potassium	mg/L	5.0	2-15	26.0
Sulfate	mg/L	100.0	20-140	170.0
Total nitrogen	mg/L	10.0	<10	220.0
Dissolved oxygen	mg/L	5.0	8.2-8.6	7.4
Sediment	mg/L	500.0	10-500	25,800
Conductivity	usiemens/cm	1,650	800-4,000	6,970
Turbidity	NTUs	1	8-110	51,000

Table 6.10—Suggested conservation measures for prescribed fires.

Fire type	Item	Best Management Practice
Prescribed	1	Carefully plan burning to adhere to weather, time of year, and fuel conditions that will help achieve the desired results and minimize impacts on water quality.
	2	Evaluate ground conditions to control the pattern and timimg of the burn.
	3	Intense prescribed fire for site preparation should not be conducted in Streamside Management Zones (SMZ) except to achieve riparian vegetation management objectives.
	4	Piling and burning for slash removal should not be conducted in SMZs.
	5	Avoid construction of firelines in SMZs.
	6	Avoid conditions requiring extensive blading of firelines by heavy equipment.
	7	Use handlines, firebreaks, and hoselays to minimize blading of firelines.
	8	Use natural or in-place barriers to minimize the need for fireline construction.
	9	Construct firelines in a manner that minimizes erosion and prevents runoff from directly entering watercourses.
	10	Locate firelines on the contour whenever possible, and avoid straight up-downhill placement
	11	Install grades, ditches, and water bars while the line is being constructed.
	12	Install water bars on any fireline running up-down slope, and direct runoff onto a filter strip or sideslope, not into a drainage.
	13	Construct firelines at a grade of 10 percent or less where possible.
	14	Adequately cross-ditch all firelines at time of construction.
	15	Construct simple diversion ditches or turnouts at intervals as needed to direct surface runoff off a plowed line and onto undisturbed forest floor or vegetation for dispersion of water and soil particles.
	16	Construct firelines only as deep and wide as necessary to control the spread of the fire.
	17	Maintain the erosion control measures on firelines after the burn.
	18	Revegetate firelines with adapted herbaceous species. Native plants are preferable when there are adequate sources of seed.
	19	Execute burns with a well-trained crew and avoid high-severity burning.

Table 6.11—Suggested conservation measures for wildfires.

Fire type	Item	Best Management Practice
Wildfire	1	Review and use BMPs listed for prescribed fires when possible.
	2	Whenever possible, avoid using fire-retardant chemicals in SMZs and over watercourses, and use measures to prevent their runoff into watercourses.
	3	Do not clean retardant application equipment in watercourses or locations that drain into waterways.
	4	Close water wells excavated for wildfire suppression activities as soon as practical following fire control.
	5	Provide advance planning and training for firefighters that considers water quality impacts when fighting wildfires. This can include increasing awareness so direct application of fire retardants to waterbodies is avoided and firelines are placed in the least detrimental position.
	6	Avoid heavy equipment use on fragile soils and steep slopes.
	7	Implement Burned Area Emergency Rehabilitation (BAER) team recommendations for watershed stabilization as soon as possible.

Summary

When a wildland fire occurs, the principal concerns for change in water quality are: (1) the introduction of sediment; (2) the potential increasing nitrates, especially if the foliage being burned is in an area chronic atmospheric deposition; (3) the possible introduction of heavy metals from soils and geologic sources within the burned area; and (4) the introduction of fire retardant chemicals into streams that can reach levels toxic to aquatic organisms.

The magnitude of the effects of fire on water quality is primarily driven by fire severity, and not necessarily by fire intensity. Fire severity is a qualitative term describing the amount of fuel consumed, while fire intensity is a quantitative measure of the rate of heat release (see chapter 1). In other words, the more severe the fire, the greater the amount of fuel consumed and nutrients released, and the more susceptible the site is to erosion of soil and nutrients into the stream where they could potentially affect water quality. Wildfires usually are more severe than prescribed fires. As a result, they are more likely to produce significant effects on water quality. On the other hand, prescribed fires are designed to be less severe and would be expected to produce less effect on water quality. Use of prescribed fire allows the manager the opportunity to control the severity of the fire and to avoid creating large areas burned at high severity. The degree of fire severity is also related to the vegetation type. For example, in grasslands the differences between prescribed fire and wildfire are probably small. In forested environments, the magnitude of the effects of fire on water quality will probably be much lower after a prescribed fire than after a wildfire because of the larger amount of fuel consumed in a wildfire. We expect canopy-consuming wildfires to be the greatest concern to managers because of the loss of canopy coupled with the destruction of soil aggregates. These losses present the worst-case scenario in terms of water quality. The differences between wild and prescribed fire in shrublands are probably intermediate between those seen in grass and forest environments.

Another important determinant of the magnitude of the effects of fire on water quality is slope. Steepness of the slope has a significant influence on movement of soil and nutrients into stream channels where it can affect water quality. Wright and others (1976) found that as slope increased in a prescribed fire, erosion from slopes accelerated. If at all possible, the vegetative canopy on steep, erodible slopes needs to be maintained, particularly if adequate streamside buffer strips do not exist to trap the large amounts of sediment and nutrients that could be transported quickly into the stream channel. It is important to maintain streamside buffer strips whenever possible, especially when developing prescribed fire plans. These buffer strips will capture much of the sediment and nutrients from burned upslope areas.

Nitrogen is of concern to water quality. If soils on a particular site are close to N saturation, it is possible to exceed maximum contamination levels of NO_3-N (10 ppm or 10 mg/L) after a severe fire. Such areas should not have N-containing fertilizer applied after the fire. Review chapter 3 for more discussion of N. Fire retardants typically contain large amounts of N, and they can cause water quality problems where drops are made close to streams.

The propensity for a site to develop water repellency after fire must be considered (see chapter 2). Water-repellent soils do not allow precipitation to penetrate down into the soil and therefore are conducive to erosion. Severe fires on such sites can put large amounts of sediment and nutrients into surface water.

Finally, heavy rain on recently burned land can seriously degrade water quality. Severe erosion and runoff are not limited to wildfire sites alone. But if postfire storms deliver large amounts of precipitation or short-duration high intensity rainfalls, accelerated erosion and runoff can occur even after a carefully planned prescribed fire. Conversely, if below-average precipitation occurs after a wildfire, there may not be a substantial increase in erosion and runoff and no effect on water quality.

Fire managers can influence the effects of fire on water quality by careful planning before prescribed burning. Limiting fire severity, avoiding burning on steep slopes, and limiting burning on potentially water-repellent soils will reduce the magnitude of the effects of fire on water quality.

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Chapter 7: Aquatic Biota

Fire Effects on Fish

Prior to the 1990s, little information existed on the effects of wildfire on fishes and their habitats (fig. 7.1). Severson and Rinne (1988) reported that most of the focus of fire effects on riparian-stream ecosystems—that is, habitat for fishes—was on hydrological and erosional responses to vegetation removal and resultant effects on sedimentation and water quality. Most of these studies were conducted in the 1970s (Anderson 1976, Tiedemann and others 1978) and examined water quality and quantity effects, algae, and aquatic micro- and macroinvertebrates. Accordingly, they addressed the potential "indirect effects" of fire on fishes; none addressed the "direct effects" of fire on fishes.

The Yellowstone Fires in 1988 ushered in an extensive effort to examine both the direct and indirect effects of wildfire on aquatic ecosystems (Minshall and others 1989, Minshall and Brock1991). Other studies on fishes were conducted following some of the historically worst wildfires in the Southwestern United States

Figure 7.1—Crowning Clear Creek Fire, Salmon, Idaho. (Photo by Karen Wattenmaker).

(Rinne 1996, Rinne and Neary 1996). The result is that most of the information available on fire effects on fishes and their habitats has been generated in the 1990s and on a regional basis. By the late 1990s, summary and review papers on the topic of fire and aquatic ecosystem, including fishes, were drafted (Reiman and Clayton 1997, Gresswell 1999), which brought 20th century information together and suggested future research and management direction for both corroborating aquatics-fisheries and fire management and conservation of native, sensitive species (fig. 7.2). This state-of-our-knowledge documentation is what we use to base 21st century fisheries and aquatic management relative to both wild and prescription fires.

Because of the increased number, size, and intensity of fires commencing in 2000 in the Western and Southwestern United States, we now see a marked increase in data describing the impact of post-wildfire events on fishes (Rinne and Carter, in press, Rinne 2003a,b). While before the turn of the century we had information on fire effects on salmonid species only, now the data base includes a dozen native fish species and several nonnative fish species (table 7.1). Key messages from these data are discussed below.

Direct Fire Effects

Fire can result in immediate mortalities to fishes, but few studies have documented direct mortality following wildfire (McMahon and de Calesta 1990, Minshall and Brock 1991, Reiman and others 1997). High severity fire and heavy fuel and slash accumulations in the riparian zone are the common predisposing factors for direct fish mortality. Moring and Lantz (1975) found some fish mortality after prescribed fire conducted after harvesting in the Alsea Basin of Oregon. The fish kill was confined to stream heads where slash accumulations were heavy and the prescribed fire reached levels of high severity. Rinne (1996) found no significant reduction in densities of fishes in three streams as a direct result of a large fire (Dude Fire, Arizona, 1990) that burned across the watersheds encompassing them. Although the Dude Fire had large areas of high severity fire, many of the riparian areas either suffered only low severity fire or did not burn at all. Reiman and others (1997) reported both dead fish and reaches of stream with no live fish after a high severity fire burned through two riparian corridors in Idaho. On the other hand, no mortalities of the endangered Gila trout were reported immediately after the Divide Fire in southwestern New Mexico (Propst and others 1992).

Figure 7.2—Wildfire encroaching on a riparian area, Montana, 2002. (Photo courtesy of the Bureau of Land Management, National Interagency Fire Center, Image Portal).

Key factors in immediate postfire fish mortality are the size of the riparian area, the riparian fuel load, fire severity, and stream size. Small streams with high fuel loads and high severity fire are the ones most likely to suffer immediate aquatic organism mortality from fire.

Fire retardants can also be a source of fish mortality (chapter 6 this volume, Van Meter and Hardy 1975). Dead fish have been reported following fire retardant application (Jones and others 1989); however, documentation is poor (Norris and Webb 1989). The number of retardant drops and orientation to the stream are key factors determining fish mortality.

Indirect Fire Effects

The indirect effects of large fires on fishes are better documented and can be significant (Reiman and Clayton 1997). Within 2 weeks after the Divide Fire in New Mexico, a single Gila trout was collected from Main Diamond Creek. Sampling 3 months later suggested extirpation of this endangered species from this headwater stream (Rinne and Neary 1996). Similarly, Rinne (1996) reported dead fishes on streambanks 2 weeks postfire and documented only a single brook trout at spring outflow in Dude Creek remaining 3 months after the Dude Fire. In both cases, "slurry ash flows" appeared responsible for fish mortality within onset of summer monsoons and basically local extirpation after several months of sustaining flooding of stream corridors resulting from heavy monsoon precipitation and vegetation removal (fig. 7.3).

Rinne and Carter (in press a) documented the complete loss of four species and more than 2,000 individual fishes from the fire impacted reaches of Ponil Creek, New Mexico, following the Ponil Complex 2002 wildfire (table 7.2). Drought conditions and stream intermittency combined with ash and flood flows synergistically resulted in this total loss of fishes (Rinne in press b). By comparison, postfire ash and flood flows

Table 7.1—Species that have been affected by wildfire in the Southwestern United States from 1989 to 2003. Species are listed by respective-named fires and locations. Nonnative species are denoted by an asterisk.

Fire	State	Common name	Scientific name	Stream
Divide	AZ	Gila trout	*Oncorhynchus gilae*	Main Diamond
Dude	AZ	*Rainbow trout	*Oncorhynchus mykiss*	Dude and Ellison
		*Brook trout	*Salvelinus fontinalis*	Bonita Creek
Ponil	NM	*Rainbow trout	*Oncorhynchus mykiss*	Ponil
		Blacknose dace	*Rhinichthys atratulus*	Ponil
		Creek chub	*Semotilus atromaculatus*	Ponil
		White sucker	*Catostomus commersoni*	Ponil
Borrego	NM	*Brown trout	*Salmo trutta*	Rio Medio
Cub Mtn.	NM	Desert sucker	*Catostomus clarki*	West Fork Gila
		Sonora sucker	*Catostomus insignis*	West Fork Gila
		Longfin dace	*Agosia chrysogaster*	West Fork Gila
		Speckled dace	*Rhinichthys osculus*	West Fork Gila
		Spikedace	*Meda fulgida*	West Fork Gila
		Loach Minnow	*Rhinichthys cobitis*	West Fork Gila
		Roundtail chub	*Gila robusta*	West Fork Gila
Picture	AZ	Headwater chub	*Gila nigra*	Turkey (T), R, S
		Speckled dace	*Rhinichthys osculus*	T, Rock (R), S
		Desert Sucker	*Catostomus clarki*	T, R, Spring (S)
		*Green sunfish	*Lepomis cyanellus*	R, S
		*Yellow bullhead	*Ameiurus natalis*	R, S
		*Brown trout	*Salmo trutta*	R, S
Dry Lakes	NM	Gila chub	*Gila intermedia*	Turkey

Figure 7.3—Slurry ash flow after the Rodeo-Chediski Fire, Apache-Sitgreaves National Forest, Arizona, 2002. (Photo by Daniel Neary).

Table 7.2—Comparison of fish species abundance and total fish numbers in Ponil Creek, site one, at initial (June) and final autumn (October) sampling, 2002.

	Month	
Species	**June**	**October**
	- - - - Number of fish - - - - -	
Site 1: (Unburned)		
Rainbow trout	18	13
White sucker	8	5
Creek chub	6	2
Longnose dace	15	29
TOTAL: Site 1	47	49
TOTAL: Sites 2-8	1,910	0

in Rio Medio, New Mexico, following the Borrego Fire, reduced brown trout *Salmo trutta* populations by 70 percent from June to October 2002 (table 7.3). Similarly, six native cypriniform (minnow and sucker) species were reduced 70 percent in the West Fork of the Gila River following the Cub Mountain Fire (table 7.4). Both of these streams had summer, base-flushing flows that apparently diluted the impacts of ash and flood flows from July to October 2002 and enabled fishes to survive in fire-impacted reaches of the river. In 2003, the Picture Fire in Arizona impacted three streams and six species of fishes (including three native species). It markedly altered stream habitat on the Tonto National Forest (Carter and Rinne in press). Overall, fish numbers were reduced by 90 percent (fig. 7.4). Finally, two fires, the Lake Complex in New Mexico and the Aspen Fire in Arizona, probably resulted in total loss of the endangered Gila chub (*Gila intermedia)* in two streams. In summary, short-term data suggest from 70 percent to total loss of fishes in wildfire-impacted reaches of streams in the Southwest (Rinne 2003a,b).

Bozek and Young (1994) noted mortalities of four species of salmonids 2 years postfire following heavy precipitation and flooding on a burned watershed in Wyoming. Death was attributed to the increase in suspended sediment loading. In Montana and Idaho, Novak and White (1989) reported that flooding and debris flows extirpated fish from stream reaches below forest fires

Table 7.3—Pre- and postfire comparison of brown trout densities per 50-m reaches of stream, Rio Medio, Santa Fe National Forest. Percent reductions between June and October are in parentheses. Sites 2 and 3 were not sampled in August.

	Month		
Site	**June**	**August**	**October**
	- - - - - - Number of fish per 50 m reach - - - - - -		
1	74	33	21 (72)
2	77	—	19 (75)
3	97	—	18 (86)

Other Anthropogenic Influencing Factors

Watershed disturbances have occurred on the landscapes of National Forests for the past century. Roads to support timber harvest, timber harvest itself, and grazing of watersheds have cumulatively disturbed and altered watersheds. Dams, diversions, and road culverts, in concert with introduction of nonnative species of fishes, have fragmented and isolated native fish populations (Rinne 2003a,b). Fire suppression itself has greatly altered the vegetative and litter component of forested landscapes (Covington and Moore 1992, Covington and others 1994), increasing tree densities and litter loads. These combined have facilitated intense crowning fires that can be more devastating to fishes and aquatic habitats (Rinne and Neary 1996).

Temporal-Spatial Scales

Most fires that burn on the landscape are less than several acres (1 ha) in size because they are normally suppressed through fire management. By comparison, only 1 percent of the wildfires are responsible for more than 90 percent of the landscaped burned (DeBano and others 1998). Most studies of the effects of fire on fishes and aquatic systems are of short term (less than 5 years) (Gresswell 1999). Although attempts have been made to extrapolate the effects of fire on aquatic systems and organisms to a watershed scale, this in reality has not been achieved to date (Gresswell 1999). Attempts to connect the effects of fire 25 to 50 years previous have been made (Rinne and Neary 1996, Rinne 2003a,b, Albin 1979) but are wanting at best. Sampling of two streams impacted by the Dude Fire (1990), a decade after the fire, suggests impacts of fire on stream fish populations may be

Table 7.4—Total numbers of fishes in 50-meter reaches of stream in the West Fork of the Gila River, July and October, 2002, after the Cub Mountain Fire.

Date	**Stream condition**	**Site 1**	**Site 2**
		- - Number of Fish - -	
Early July	Post-Wildfire	168	560
Late July	Post-Wildfire, After 1st Storm	278	481
October	Post-Wildfire, After 2nd Storm	50	118

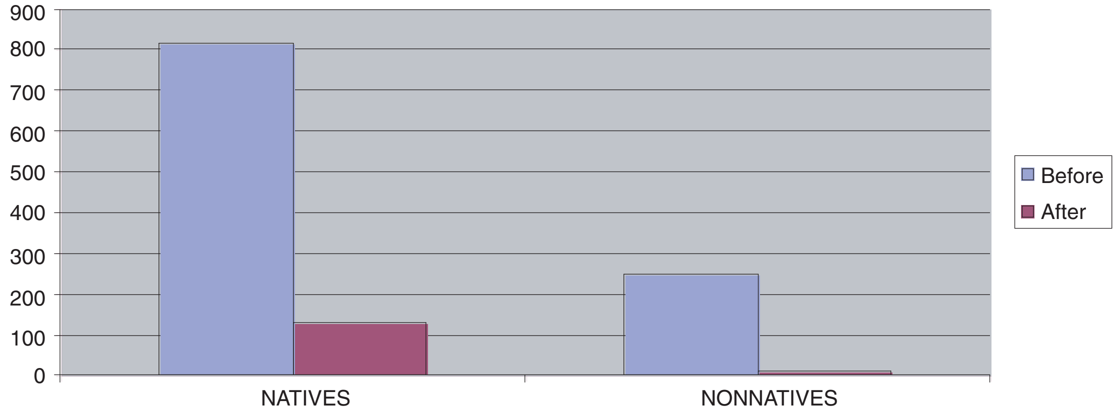

Figure 7.4—Impact of the Picture Fire on fishes in Spring Creek, Tonto National Forest, 2003.

chronic (fig. 7.5). Further, attempting to simplify these effects in time and space is not advisable.

Species Considerations

Recovery of fishes to a stream can occur quickly depending on connectivity of populations and refugia populations to replenish locally extirpated populations (Propst and others 1992, Reiman and Clayton 1997, Gresswell 1999, Rinne and Calamusso in press). In the Southwest, fragmentation of streams as a result of land uses, coupled with fisheries management such as the introduction of nonnative fishes, most often preclude access and replenishing of fishes to a fire-extirpated stream. A big consideration is whether the species is a threatened or endangered one, such as the Gila trout in the Southwest or the Bull or redband trout in the Pacific Northwest. These cases may necessitate quick removal of surviving fishes prior to ash flows or intense flooding resulting from watershed denudation. In case of put and take, introduced sport fishes, these can always be replenished through stocking. Managers should be vigilant of opportunities to restore native stocks or races of fishes to streams where introduced species have been removed through the aftermath of wildfire. This has been successfully completed for the Gila trout in Dude Creek a decade after extirpation of a brook trout population by the Dude Fire of 1990.

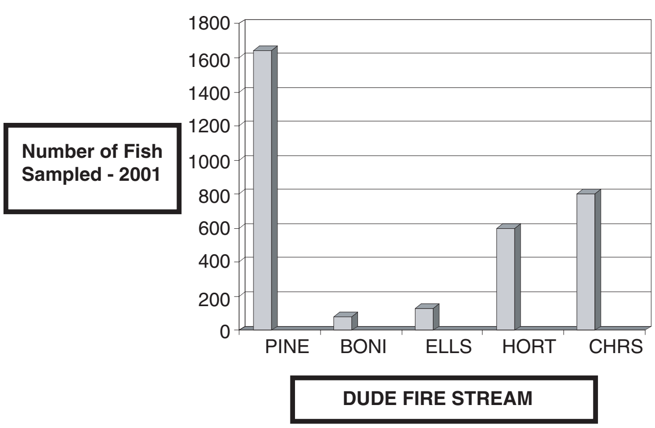

Figure 7.5—Response of fish numbers in two streams, Bonita (BONI) and Ellison (ELLS) Creeks, compared to three nonfire impacted streams, Pine (PINE), Horton (HORT), and Christopher (CHRS) Creeks, 11 years after the Dude Fire, 1990, Tonto National Forest, Arizona.

Summary and General Management Implications for Fish

The effects of wildland fire on fish are mostly indirect, with most studies demonstrating the effects of ash flows, changes in hydrologic regimes, and increases in suspended sediment on fishes. These impacts are marked, ranging from 70 percent to total loss of fishes. There are some documented instances of fires killing fish directly (Reiman and others 1997).

The largest problems arise from the longer term impact on habitat that includes changes in stream temperature due to plant understory and overstory removal, ash-laden slurry flows, increases in flood peakflows, and sedimentation due to increased landscape erosion (fig. 7.6).

Most of information on the effects of wildfire on fishes has been generated since about 1990. The Yellowstone Fires of 1988 resulted in extensive study of fire effects on aquatic ecosystems including fishes. The effects of fire retardants on fishes are observational and not well documented. Further, all information is from forested biomes as opposed to grasslands.

Anthropogenic influences, largely land use activities over the past century, cumulatively influence fire

Figure 7.6—Four-Corners Fire encroaching on a lake near Crane Prairie, Oregon. (Photo by Tom Iraci).

effects on fishes. Fire suppression alone has affected vegetation densities on the landscape, and affected the severity and extent of wildfire and, in turn, its effects on aquatic ecosystems and fishes. Most studies of fire effects on fishes are short term (less than 5 years) and local in nature. A landscape approach to analyses has not been made to date. Fish can recover rapidly from population reductions or loss but can be markedly limited or precluded by loss of stream connectivity because of human-induced barriers. Fisheries management postfire should be based on species and fisheries and their management status. Managers should be vigilant of opportunities to restore native fishes in event of removal of introduced, nonnative, translocated species.

Recent advances in atmospheric, marine, and terrestrial ecosystem science have resulted in the correlation of ocean temperature oscillations, tree ring data, and drought. These newer advances, along with current information on fire effects on fishes and data on climate change, drought, and recent insect infestations, all combine to show that the greatest impacts on fish may lie ahead. For example, new climate change analyses suggest that the Southeastern and parts of the Western United States have a high probability of continuing and future drought into the next 20 to 30 years, and that the potential impact of wildfire on fishes in the Southwest is marked. Also, the overarching global indicators are starting to unfold: tree ring data indicate 2002 was the driest year in the past 1,000 years (Service 2004); Atlantic and Pacific oceanic temperatures are in turn correlated with historic wet and dry cycles.

The cumulative impacts of warmer and drier climate, with increasing insect outbreaks at higher and higher elevations, will potentially increase the risk of wildfire throughout the ranges of Southwestern native fishes (fig. 7.7). Such impacts increase the future risk of loss of fish gene pools and Southwestern native fishes' sustainability.

Birds

The effects of fires on bird populations are covered in volume 1 of this series (Effects of Fire on Fauna, Smith 2000). Aquatic areas and wetlands often provide refugia during fires. However, wetlands such as cienegas, marshes, cypress swamps, spruce and larch swamps, and so forth do burn under the right conditions. The impacts of fires on individual birds and populations in wetlands would then depend upon the season, uniformity, and severity of burning (Smith 2000).

Reptiles and Amphibians

The effects of fires on reptile and amphibian populations are also covered in volume 1 of this series (Effects of Fire on Fauna, Smith 2000). Russell and others (1999) concluded that there are few reports of fire-caused injury to herpetofauna in general, much less aquatic and wetland species. They noted that aquatic and semiaquatic herpetofauna benefit from wetland fires due to vegetation structure improvement and increases in the surface area of open water. Excessive postfire sedimentation of streams or small standing bodies of water could potentially reduce habitat for reptile and amphibian populations. However, the effects are not well documented.

Mammals

Aquatic and wetland dwelling mammals are usually not adversely impacted by fires due to animal mobility and the lower frequency of fires in these areas. Lyon and others (2000a,b) discuss factors such as fire uniformity, size, duration, and severity that affect mammals. However, most of their discussion relates to terrestrial mammals, not aquatic and wetland ones. Aquatic and wetland habitats also provide safety zones for mammals during fires.

Invertebrates

The fauna volume in this series deals mainly with terrestrial invertebrates (Smith 2000). As with fishes, little information existed on the effects of wildfire on aquatic macroinvertebrates prior to 1980, although some studies were conducted in the 1970s (Anderson

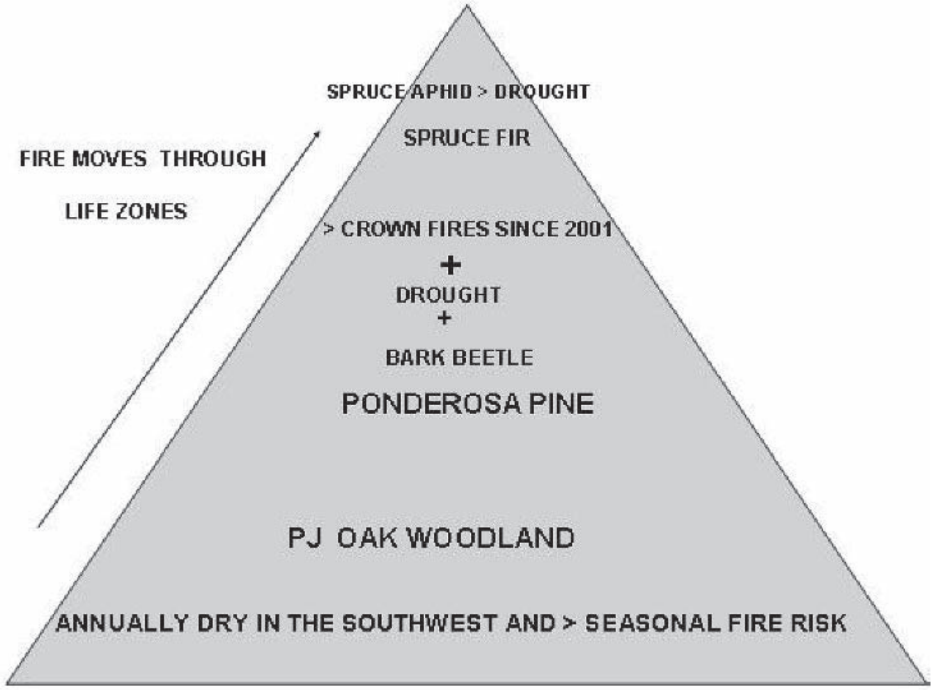

Figure 7.7—Multiple, cumulative impacts are occurring in the West and Southwest to greatly increase the risk of wildfire and further the marked loss of native fishes.

1976). Tiedemann and others (1978) examined water quality and quantity effects, algae, and aquatic micro- and macroinvertebrates. As with fishes, these studies addressed the potential "indirect effects" of fire on aquatic macroinvertebrates rather than any "direct effects."

Parallel with information generation on fishes, the Yellowstone Fires in 1988 ushered in an extensive effort to examine both the direct and indirect effects of wildfire on aquatic ecosystems (Minshall and others 1989, Minshall and Brock 1991). Isolated studies on aquatic macroinvertebrates have been conducted following some of the historically worst wildfires in the Southwestern United States (Rinne 1996). By the late 1990s, a summary and review paper on the topic of fire and aquatic ecosystems, including aquatic macroinvertebrates, was produced (Gresswell 1999). As for fishes, the Gresswell paper is the state-of-the-art reference for the effects of fire on aquatic macroinvertebrates, collates a comprehensive review of information from the 20th century, and suggests future research and management direction.

Response to Fire

Similar to fishes, the direct effects of fire on macroinvertebrates have not been observed or reported. Albin (1979) reported no change in aquatic macro invertebrates abundance during and after fire and no dead macroinvertebrates observable. Rinne (1996) could not document any changes in mean aquatic macroinvertebrate density in three headwater streams from prefire to immediately following the then-worst wildfire Arizona history, the Dude Fire (table 7.5). However, sampling after initial "ash slurry flows" (in the 2 weeks postfire) revealed an

Table 7.5—Aquatic macroinvertebrate densities following the Arizona Dude Fire, Tonto National Forest, 1990 (Adapted from Rinne 1996).

	Date				
Stream	**Prefire**	**07/03/90**	**07/26/90**	**05/20/91**	**06/08/91**
	Number/m^2 x 1,000				
Dude Creek	5.4	5.0	0.1	2.8	2.5
Bonita Creek	8.6	3.8	0.1	6.0	3.8
Ellison Creek	9.7	10.8	0.0	6.5	2.9

80 to 90 percent reduction in mean densities. Further sampling after several significant flood events determined that macroinvertebrate populations were near zero. Sampling over the next 2 years revealed dramatic fluctuations in density and diversity (number of species) of macroinvertebrates in all three streams.

In general, responses to fire by aquatic macroinvertebrates, as with fishes, are indirect and vary widely in response. Studies in the 1970s by Lotspeich and others (1970) and Stefan (1977) suggested the effects of fire were minimal or undetectable. La Point and others (1996) also reported no difference in macroinvertebrate distribution in streams encompassed by burned and unburned sites nor shifts in water chemistry. Changes in functional feeding groups (Cummins 1978) by aquatic macroinvertebrates have been noted and attributed to substrate stability differences between fire impacted and nonimpacted streams. By comparison, Richards and Minshall (1992) reported that in the first 5 years postfire, macroinvertebrate diversity in streams affected by fire exhibited greater annual variation in diversity than in unaffected streams. Annual fluctuations or variation in diversity did decline with time; however, greater species richness was sustained in streams within burned areas.

As indicated in the introduction, the Yellowstone Fires provided an opportunity to study the effects of fire on aquatic ecosystems and to generate most of the available information on aquatic macroinvertebrate response to fire. In the first year postfire, minor declines in macroinvertebrate abundances, species richness, and diversity were recorded (Robinson and others 1994, Lawrence and Minshall 1994, Minshall and others 1995, Mihuc and others 1996). Two years postfire, these indices had increased but less so in smaller order streams (Minshall and others 1997). Food supply, in the form of unburned coarse particulate material, was suggested to be the factor most important to macroinvertebrates.

Jones and others (1993) reported macroinvertebrate abundance fluctuations in four larger (fifth and sixth order streams) in Yellowstone National Park, yet species diversity and richness did not decline. A change in functional feeding groups from shredder-collector species to scraper-filter feeding species occurred (Jones and others 1993).

Temporally, the effects of fire on macroinvertebrates can be sustained for up to a decade (Roby 1989) or possibly longer (greater than three decades; Albin 1979). Roby and Azuma (1995) studied changes in benthic macroinvertebrate diversity in two streams after a wildfire in northern California (fig. 7.8).

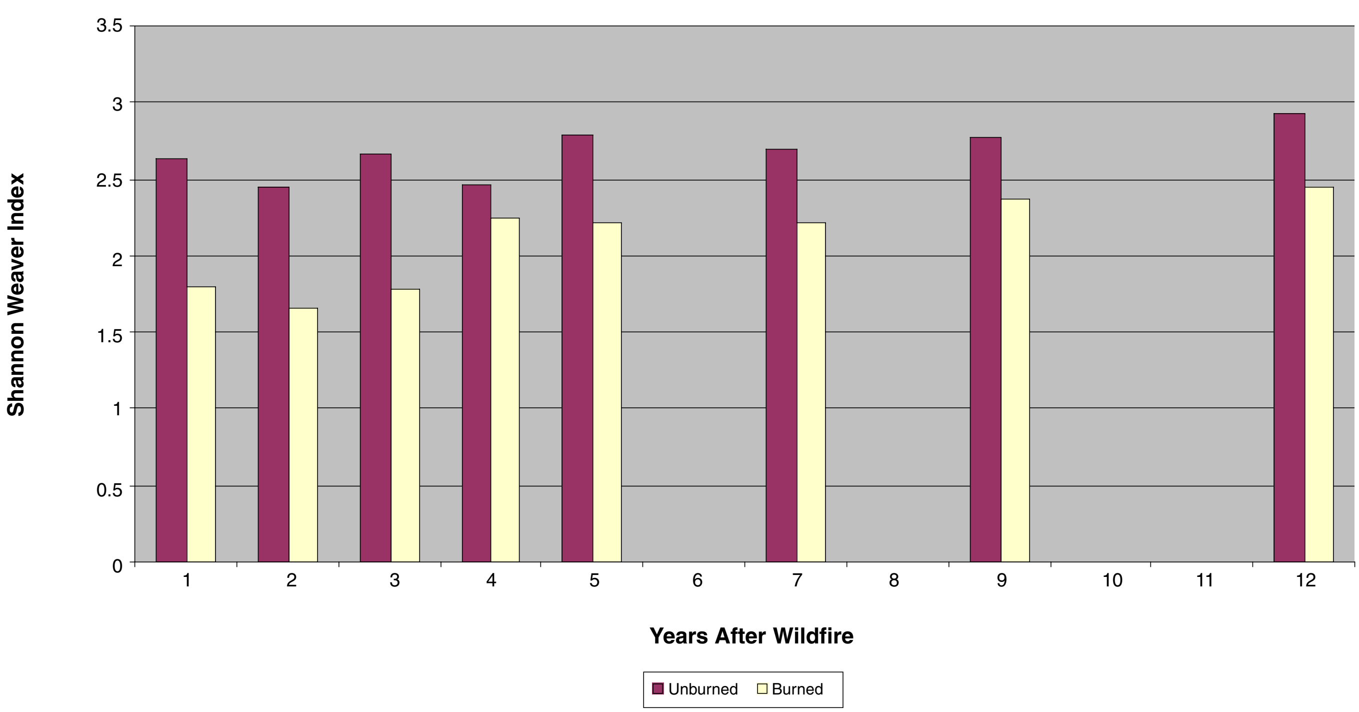

Figure 7.8—Invertebrate diversity changes after a wildfire in northern California. (Adapted from Roby, K.B.; Azuma, D.L. 1995. Changes in a reach of a northern California stream. Figure 2. Environmental Management. Copyright © 1995. With kind permission of Springer Science and Business Media).

Invertebrate diversity, density, and taxa richness in the stream of the burned watershed were low immediately after the fire compared to the unburned watershed. Within 3 years, mean density was significantly higher in the burned watershed (fig. 7.9). A decade after the wildfire, taxa richness and species diversity were still lower in the stream of the burned watershed. Albin (1979) reported lower diversity of macroinvertebrates in reference compared to streams in burned areas. Macroinvertebrate density also was greater in burned streams; however, Chironomidae (immature midges adapted to disturbed areas characterized by fine sediments) were the most abundant group represented in samples. Mihuc and others (1996) suggested changes in physical habitat and availability of food supply are the primary factors affecting postfire response of individual macroinvertebrate populations.

Invertebrate Summary

Similar to fishes, the recorded effects of fire on aquatic macroinvertebrates are indirect as opposed to direct. Most data have been produced during and since the 1990s. Regarding abundance, diversity, or richness, response varies from minimal to no changes, and from considerable to significant changes. Abundance of macroinvertebrates may actually increase in fire-affected streams, but diversity generally is reduced. These differences are undoubtedly related to landscape variability, burn size and severity, stream size, nature and timing of postfire flooding events, and postfire time. Temporally, changes in macroinvertebrate indices in the first 5 years postfire can be different from ensuing years, and long-term (10 to 30 years) effects have been suggested.

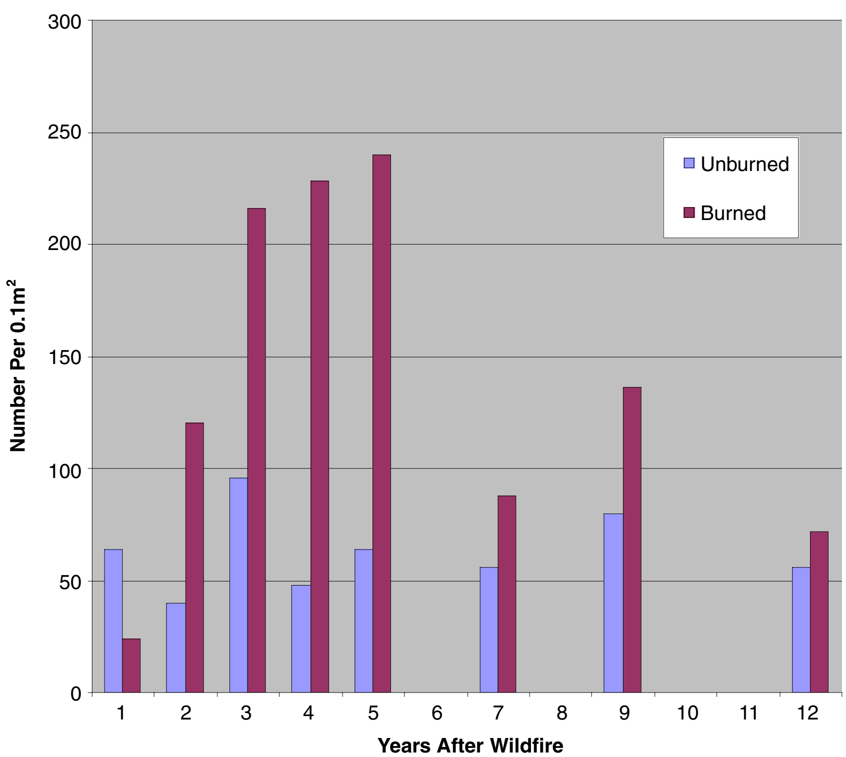

Figure 7.9—Invertebrate density changes in streams after a wildfire in northern California. (Adapted from Roby, K.B.; Azuma, D.L. 1995. Changes in a reach of a northern California stream. Figure 2. Environmental Management. Copyright © 1995. With kind permission of Springer Science and Business Media).

Notes

Part C

Other Topics

Daniel G. Neary

Part C—Other Topics

Chapters 8 through 11 in this part of the volume deal with special topics that were not considered in the original "Rainbow Series" on the effects of fire on soils (Wells and others 1979) and water (Tiedemann and others 1979). These topics include wetlands and riparian ecosystems, fire effects models and soil erosion models, the Burned Area Emergency Rehabilitation (BAER) program with its various treatments and results, and information sources such as databases, Web sites, journals, books, and so forth. Part C concludes with the References section and an appendix of glossary terms.

Wetlands science has advanced considerably since the original "Rainbow Series" was published in 1979, and the role of fire in wetlands ecosystems is now understood more completely. Tiedemann and others (1979) briefly discussed everglades wetlands of southern Florida and cypress wetlands of northern Florida. Chapter 8 of this volume recognizes the extent of wetlands in North America beyond the swamps of Florida. The authors go into considerable detail on the effects of fire in wetlands and their associated riparian areas.

Fire effects models and soil erosion models were in their infancy in the mid 1970s, and researchers and land managers then only dreamed of the compact, high speed desktop and laptop computers we now have that are able to do the millions of calculations needed to run these models. Chapter 9 briefly examines how these models increase our understanding of the effects of fire on soils and water. While this discussion is not intended as inclusive of the complete scope of fire effects models, we do intend to illustrate some of the more commonly available ones.

Rehabilitation of burned landscapes has long been recognized as a necessary step for healing severely burned watersheds to reduce erosion and mitigate adverse changes in hydrology. However, the BAER program did not start until the mid 1970s and was not initially well organized nor well understood. Chapter 10 provides a much needed synopsis of the BAER program based on a Robichaud and others (2000) literature and research review publication.

The numbers of textbooks, research reports, journal articles, users guides, and databases with specific focus on wildland fire have rapidly increased in the past quarter century. And now in the 21st century, we have readily accessible sources of fire effects information such as databases and Web sites that did not exist in the 1970s. The Web sites facilitate rapid dissemination of fire effects information to managers, researchers, and the general public. Their use and importance will continue to grow. Chapter 11 details these information sources.

Chapter 12 is an overall summary of this volume and contains suggestions for research needs and priorities.

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Chapter 8: Wetlands and Riparian Systems

Introduction

Wetlands and riparian ecosystems contain biotic communities that develop because of the regular presence of water (Brooks and others 2003). Undisturbed or well-managed wetlands and riparian ecosystems provide benefits that are proportionally much more important compared to the relatively small portion of the land area that they occupy because these ecosystems support a diversity of plants and animals that are not found elsewhere. Wetlands and riparian ecosystems also play important roles relating to enhanced water quality, flood peak attenuation, and reduced erosion and sediment transport. At the same time, these ecosystems are often fragile and often easily disturbed, and both wildfires and prescribed burns can affect the soil, water, litter, and vegetation in wetlands and riparian ecosystems.

Although the terms "wetland" and "riparian" are sometimes used interchangeably, it is important to distinguish between the two systems both ecologically and in terms of their individual responses to fire. While the presence of water is an essential feature of both ecosystems, wetlands are more typically represented by large accumulations of surface organic matter that are waterlogged for long periods during the year, producing persistent anaerobic soil conditions; riparian areas, on the other hand, are characterized more as communities of water obligate plants that are typically found along river systems where moisture is readily available but the soils are saturated for only short periods. These differences in morphology and hydric environment strongly affect the type of fire that burns in each ecosystem. When the water table is low in wetlands, surface and crown fires often ignite smoldering grond fires in think layers of humus, peat, and muck when the resulting in deep depth of burn. Conversely, when the water table is high, fires exhibit low depth of burn regardless of the fireline intensity. In riparian areas fires often back down hill at low intensity leaving irregular mixed severity burns. Periodically, however, riparian areas experience high intensity crown fires due to the high fuel continuity and the chimney effect of the terrain. Thus mixed severity fire regimes are common in wetland and riparian communities.

Wetlands

Wetlands are widely distributed from tropical to arctic regions and found in both arid and humid environments. They cover approximately 4 to 6 percent of Earth's land area and display a broad ecological range with characteristics of both aquatic and terrestrial ecosystems (Mitsch and Gosselink 1993). The wide distribution and diversity of wetlands make forming generalizations and widely adaptable management solutions difficult (Lugo 1995) (fig. 8.1).

Wetland functions are crucial to environmental quality. They supply the habitat requirements for a diverse range of plants, animals, and other organisms, some of which are threatened and endangered. Wetlands regulate water quality and quantity and are an integral element of global climate control. Improved understanding of wetlands and the relationship between fire and wetlands has become increasingly important for a number of reasons, including concerns about wetland habitat loss, global warming, toxic metal accumulation, and the treatment of hazardous fuels.

While the connection between wetlands and fire appears tenuous, fire plays an integral role in the creation and persistence of wetland species and ecosystems. Changes in climate and vegetation are recorded in the deposition of macrofossils, pollen, and peat, and disturbances such as fire are also recorded in this depositional record (Jasieniuk and Johnson 1982, Cohen and others 1984, Kuhry and Vitt 1996). Paleoecological evidence of fire is derived from a number of sources including pollen analysis and charcoal distribution. Charcoal, which is produced when plant materials are pyrolized, is a record of the influence of fire on the structure of ecosystems (Cope and Chaloner 1985). The distribution of pollen and charcoal within wetland soils reflect both fire history and environmental conditions (Kuhry 1994).

Paleoecological studies show consistent evidence of frequent fire in a wide distribution of wetlands from the Southeastern United States to the boreal regions of Canada and Alaska. For example, peat cores from the Okefenokee Swamp in Georgia exhibited distinct charcoal bands indicating a fire history (Cohen and others 1984, Hermann and others 1991). Patterns in pollen and charcoal distribution in Wisconsin and Minnesota wetlands show changes that reflect variation in environmental conditions and fire frequency (Davis 1979, Clark 1998). Trees from mires of northern Minnesota show fire scars as evidence of fire history (Glaser and others 1981). In boreal regions, peat sediments from sphagnum-dominated wetlands show a fire history as distinct charcoal layers and pollen changes (Kuhry 1994).

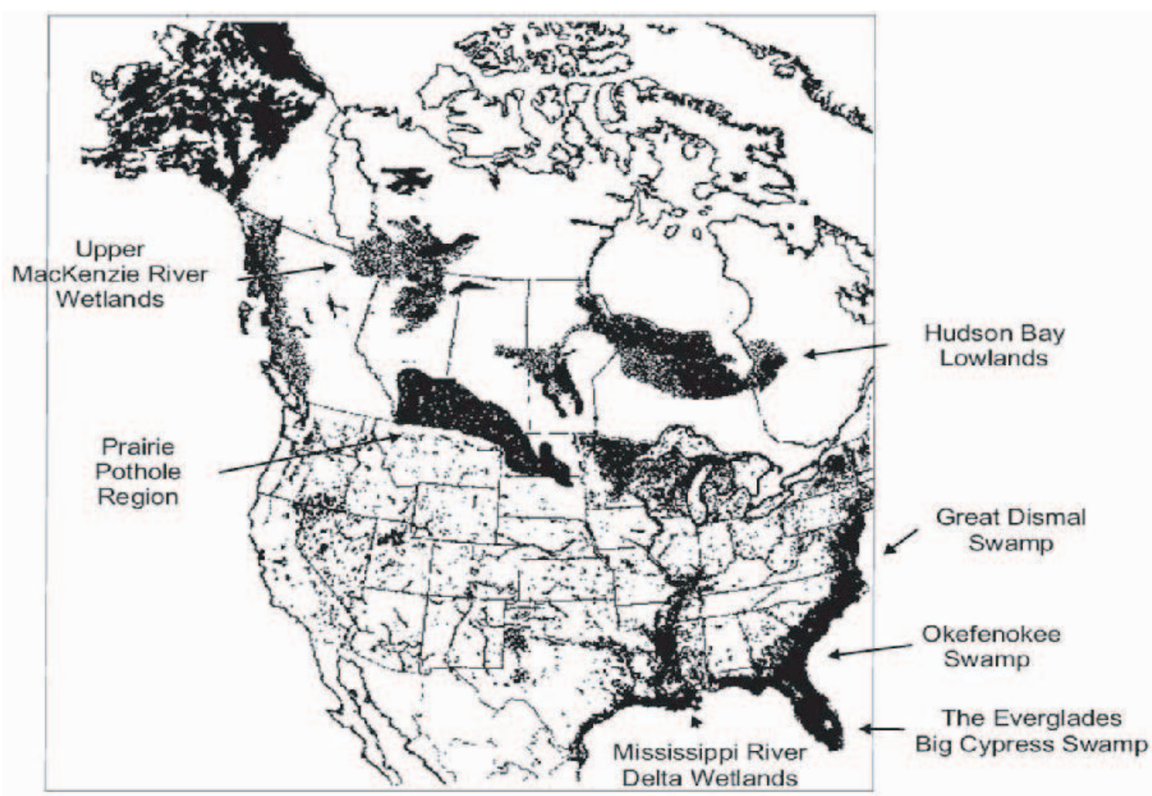


Figure 8.1—Distribution of wetland-dominated regions of North American. Shaded areas have a significant percentage of wetlands.

Wetland and Hydric Soil Classification

Not only are there a confusing number of terms and definitions to describe organic soils and organic matter while classifying wetlands (table 8.1), but during the past three decades several new wetland classification systems have been developed and used by various government agencies. This array of classification systems reflects both our understanding of wetland processes and the missions of the individual agencies.

The system developed and currently in use by the U.S. Fish and Wildlife Service (Cowardin and others 1979; table 8.2) for wetland inventory is well suited for ecological applications (Mitsch and Gosselink 1993). In this classification system, the divisions between the five wetland systems are based on hydrologic similarities in the water source and flow. Wetland systems of interest to fire managers in this classification include:

- Lacustrine wetlands, which are associated with lakes and reservoirs.
- Riverine wetlands, which are associated with flowing water of rivers.
- Estuarine wetlands, which include salt marshes.
- Palustrine wetlands, which include swamps, bogs, fens and other common freshwater wetlands.

Table 8.1—Terms and definitions used to describe organic soils and organic matter (Adapted from Soil Science Society of America 1997, and Mitsch and Gosselink 1993).

Term	Definitions
Bog	A peat-accumulating wetland that has no significant inflows and supports acidophilic mosses, particularly sphagnum.
Fen	A peat-accumulating wetland that receives some drainage from surrounding mineral soils and usually supports marsh-like vegetation. These areas are richer in nutrients and less acidic than bogs. The soils under fens are peat (Histosol) if the fen has been present for a while.
Fibric	A type of organic soil where less than one-third of the material is decomposed, and more than two-thirds of plant fibers are identifiable.
Marsh	A frequently or continually inundated wetland characterized by emergent herbaceous vegetation adapted to saturated soil conditions.
Muck	Organic soil material in which the original plant remains is not recognizable. Contains more mineral matter and is usually darker in color than peat.
Muskeg	Large expanses of peatlands or bogs; particularly used in Canada and Alaska.
Oligotrophic	Describes a body of water (for example, a lake) with a poor supply of nutrients and a low rate of formation of organic matter by phototsynthesis.
Ombrotrophic	True raised bogs that have developed peat layers higher than their surroundings and which receive nutrients and other minerals exclusively by precipitation.
Peat	Organic soil materials in which the original plant remains are recognizable (fibric material).
Peatlands	A generic term for any wetland that accumulated partially decayed plant matter.
Pocosin	Temperate zone evergreen shrub bogs dominated by pond pine, ericaceous shrubs, and sphagnum.
Sapric	Type of organic soil where two-thirds or more of the material is decomposed, and less than one-third of plant fibers are identifiable.
Wet meadow	Grassland with waterlogged soil near the surface but without standing water most of the year.
Wet prairie	Similar to marsh but with water levels usually intermediate between a marsh and a wet meadow.

Table 8.2—U.S. Fish and Wildlife Service classification hierarchy of wetland and deepwater habitats (Cowardin and others 1979). These are wetland classes of particular interest to fire management. Wetlands are shown in bold print.

System	Subsystem	Class
Marine	Subtidal	Rock Bottom, Unconsolidated Bottom, Aquatic Bed, Reef
Marine	Intertidal	Aquatic Bed, Unconsolidated Bottom, Aquatic Bed, Reef
Estuarine	Subtidal	Rock Bottom, Unconsolidated Bottom, Aquatic Bed, Reef
Estuarine	Intertidal	Aquatic Bed, Reef, Streambed, Rocky Shore, Unconsolidated Shore **Emergent Wetland** **Scrub-Shrub Wetland** **Forested Wetland**
Riverine	Tidal	Rock Bottom, Unconsolidated Streambed, Aquatic Bed Streambed, Rocky Shore, Unconsolidated Shore **Emergent Wetland**
Riverine	Low Perennial	Rock Bottom, Unconsolidated Bottom, Aquatic Bed Rocky Shore, Unconsolidated Shore **Emergent Wetland**
Riverine	Upper Perennial	Rock Bottom, Unconsolidated Bottom, Aquatic Bed Rocky Shore, Unconsolidated Shore
Riverine	Intermittent	Streambed
Lacustrine	Limnetic	Rock Bottom, Unconsolidated Bottom, Aquatic Bed
Lacustrine	Littoral	Rock Bottom, Unconsolidated Bottom, Aquatic Bed Rocky Shore, Unconsolidated Shore **Emergent Wetland**
Palustrine		Rock Bottom, Unconsolidated Bottom, Aquatic Bed Unconsolidated Shore **Moss-Lichen Wetland** **Emergent Wetland** **Scrub-Shrub Wetland** **Forested Wetland**

At a finer level, the division of wetland systems and subsystems into wetland classes is based upon the dominant vegetation type or substrate. The major wetland classes of interest in this system include:

- Forested
- Scrub-shrub
- Emergent
- Moss-lichen

Wetland classes are further modified by factors including water regime, water chemistry, and soil factors.

In addition to unique vegetation and hydrologic characteristics, wetlands are also distinguished by the presence of hydric soils. The definition of hydric soils has evolved during the last few decades, and the changes to the definition reflect our increased understanding of wetland soils genesis (Richardson and Vepraskas 2001). Although organic soils (Histisols) are commonly associated with wetlands, poorly drained mineral soils with wet moisture regimes are also common on wetland sites. An additional soil order has recently been added to the U.S soil classification system (Gelisols). This new soil order reflects the influence of low temperatures and soil moisture on soil processes. Thus, wetland soils of the boreal peatlands are now classified as Gelisols (Bridgham and others 2001).

The current definition of hydric soils includes both histisols and mineral soils (table 8.3). The primary factor contributing to the differences between hydric and nonwetland soils is the presence of a water table

Table 8.3—Definitions of hydric soils (Adapted from Mausbach and Parker 2001).

1. All Histosols
2. Mineral Soils
   a. Somewhat poorly drained with a water table equal to 0.0 foot (0.0 m) from the surface during the growing season, or
   b. Poorly drained or very poorly drained soils that have either:

      (1) water table equal to 0.0 foot (0.0 m) from the surface during the growing season if textures are coarse sand, sand, or fine sand in all layers within 20 inches (51 cm), or for other soils, or
      (2) water table at less than or equal to 0.5 foot (0.15 m) from the surface during the growing season if permeability is equal to or greater than 6.0 in/hr (15 mm/hr), or
      (3) water table at less than or equal to 1.0 foot (0.3 m) from the surface during the growing season, or
3. Soils that are frequently ponded for long duration or very long duration during the growing season, or
4. Soils that are frequently flooded for long duration or very long duration during the growing season.

close to the soil surface or frequent long-duration flooding or ponding. The oxygen-limiting conditions found in these waterlogged environments affect micro- and macrofaunal organisms and nutrient cycling processes and result in decreased decomposition rates and different metabolic end products (Stevenson 1986).

Wetland Hydrology and Fire

Wetland hydrology is a complex cycle of water inflows and outflows that are balanced by the interrelationships between biotic and abiotic factors. The presence and movement of water are dominant factors controlling wetland dynamics, nutrient and energy flow, soil chemistry, organic matter decomposition, and plant and animal community composition.

Wetland systems are characterized by annual water budgets comprising storage, precipitation, evapotranspiration, interception, and surface and ground water components. In addition to water budgets, the seasonality and movement of water is characterized by hydroperiod and hydrodynamics. *Hydroperiod* is the time a soil is saturated or flooded, and results from water table movements caused by distinct seasonal changes in the balance between inflows and outflows of a particular wetland. *Hydrodynamics* is the movement of water in wetlands and is an important process affecting soil nutrients and productivity.

Alteration of ground water levels and flow induced by anthropogenic activities such as ditching and fire line construction is a serious concern in wetland fire management (Bacchus 1995). Long-term changes in hydroperiod or hydrodynamics affect productivity, decomposition, fuel accumulation, and the subsidence of organic soils. Changes in these wetland characteristics lead to changes in fire behavior and ground fire potential.

Water budgets, hydroperiods, and hydrodynamics are unique for each wetland type and, as a result, have substantial effect on site productivity and decomposition rates (Mitsch and Gosselink 1993). Significant ground and surface water inputs are characteristic of the water budgets of marshes and fens, while precipitation and evapotranspiration dominate the water budgets of bogs and pocosins. The timing and pattern of these inputs has a distinct influence on fire occurrence. For example, the water budget of pocosin ecosystems in the Southeastern United States is dominated by rainfall and evapotranspiration with minor surface and ground water components. Typically, the highest wildfire danger occurs between March and May because the evapotranspiration rate is higher than precipitation, thereby causing a water deficit in these soils (fig. 8.2).

In contrast to the pocosin wetlands, in the boreal black spruce/feather moss wetlands, frozen soil restricts drainage during spring and early summer, and the organic soil retains a higher percentage of water. Typically, soil drying in these communities is delayed until in June or July (Foster 1983). Early season fires in these wetlands consume little organic soil material, while fires later in the season or during extended droughts can consume significant amounts of organic soil material (Heinselman 1981, Johnson 1992, Duchesne and Hawkes 2000, Kasischke and Stocks 2000).

Although studies have identified interrelationships among hydrology, fire, and soil in wetland esosystems (Kologiski 1977, Sharitz and Gibbons 1982, Frost 1995), the codominant influence of fire and hydrology is not clear. Hydrologic characteristics influence fire severity, and in turn the fire severity influences postburn hydrology and future fire severity. Both

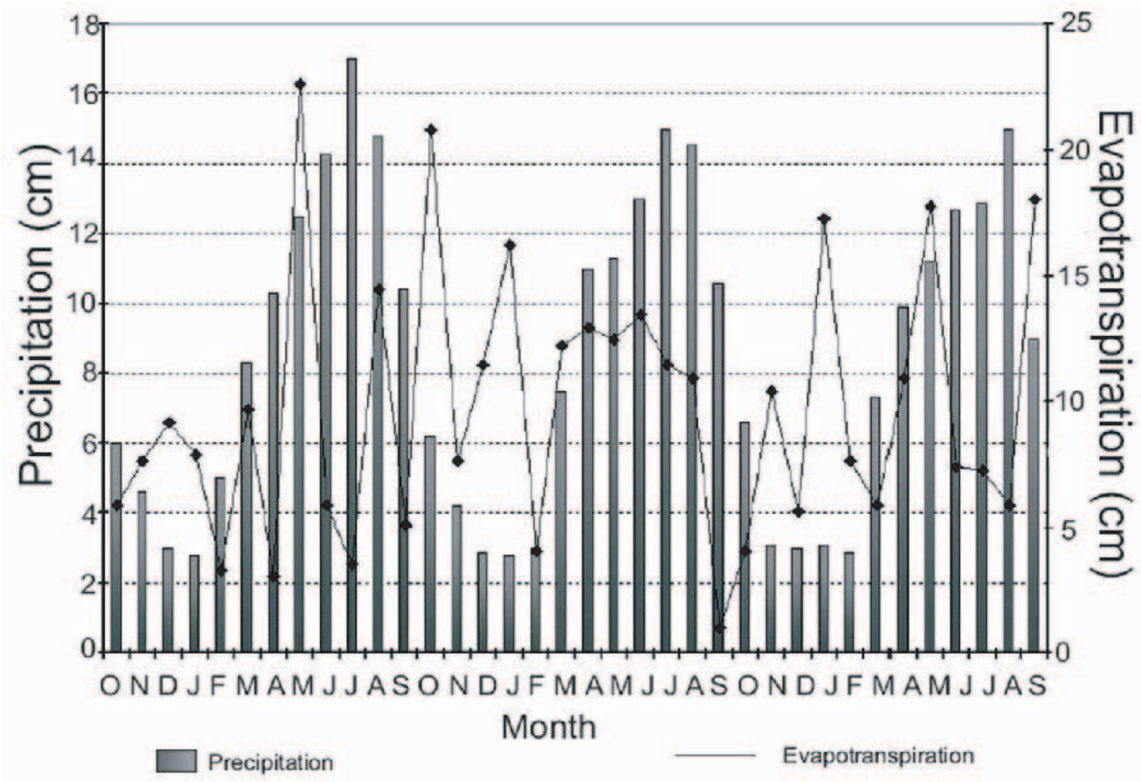


Figure 8.2—An example of precipitation and evapotranspiration dynamics in a pocosin habitat measured for a 3-year period. (Adapted from Daniel 1981, Pocosin Wetlands, Copyright © 1981, Elsevier B.V. All rights reserved).

these factors also affect numerous ecological variables including soil moisture, soil aeration, and soil temperature regime. The separation of disturbance effects from hydrological effects is difficult because wetland response to disturbance is a function of changes in numerous spatial and temporal gradients (Trettin and others 1996).

Wetland community responses to fire frequency, hydroperiod, and organic soil depth were investigated in a North Carolina pocosin (scrub-shrub wetland). The results showed that emergent sedge bogs on deep organic soils with long hydroperiods were associated with frequent fires. Deciduous bay forest communities with shallower organic soils and shorter hydroperiods were associated with decreases in fire frequency, while pine savanna communities with shallow organic soils and short hydroperiods were associated with a decrease in fire frequency (Kologiski 1977) (fig. 8.3). A similar relationship between fire and hydroperiod was found in a study of Florida cypress domes communities. Emergent/persistent wetlands with long hydroperiods were associated with frequent fires, while forested alluvial wetlands with shorter hydroperiods were associated with infrequent fires (Ewel 1990) (fig. 8.4).

Fire regimes are representative of long-term patterns in the severity and occurrence of fires (Brown 2000). They reflect the interdependence of several factors including climate, fuel accumulation, and ignition sources. Several fire regime classification systems have been developed using fire characteristics or effects produced by the fire (Agee 1993, Brown 2000).

Wetland fire regimes vary in frequency and severity. Light, frequent fires maintain the herbaceous vegetation in emergent meadows such as tidal marshes and sedge meadows (Frost 1995) (fig. 8.5). Light surface fires in *Carex stricta* dominated sedge meadows remove surface litter and resulted in increased forb germination and species diversity (Warners 1997). Infrequent stand-replacing fires are needed to regenerate Atlantic white cedar and black spruce/feather moss forested wetlands (Kasischke and Stocks 2000).

Surface fires associated with high water table conditions reduce shrub and grass cover and produce conditions favorable for Atlantic white cedar germination and survival (Laderman 1989). Experimental results have shown that when the organic soil horizons in black spruce/feather moss wetlands are removed by surface fires, the resulting elevated nutrient levels

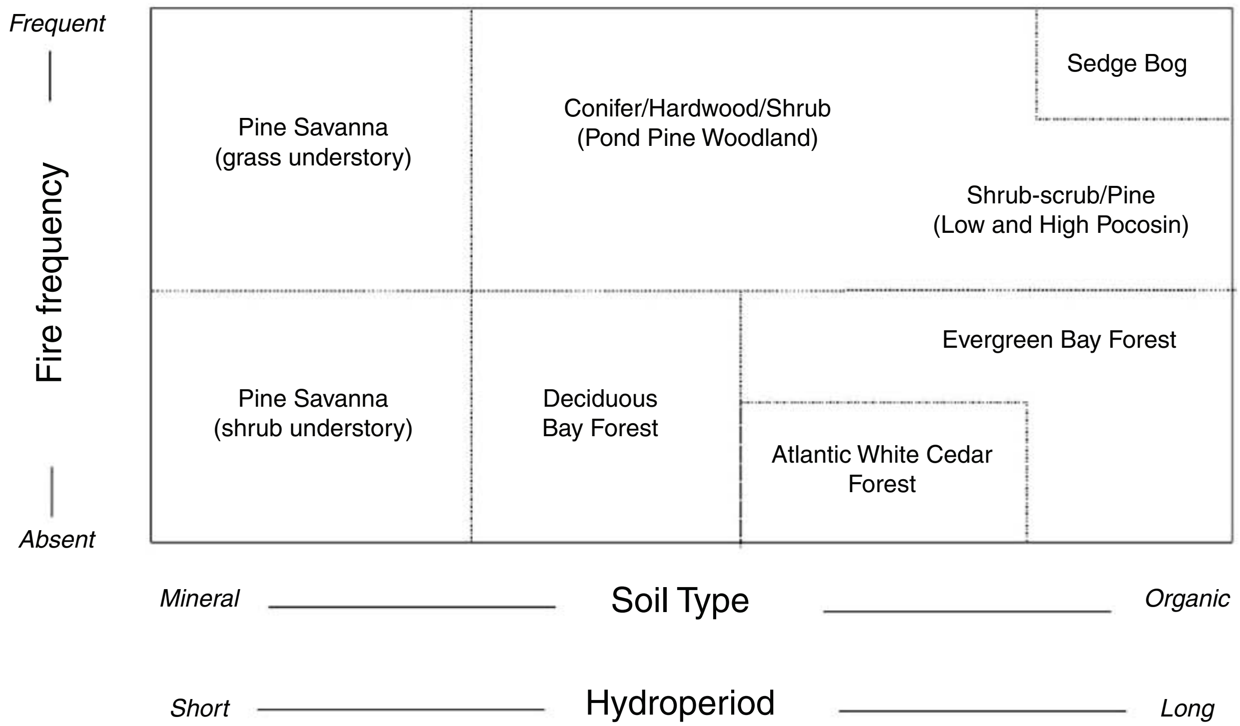
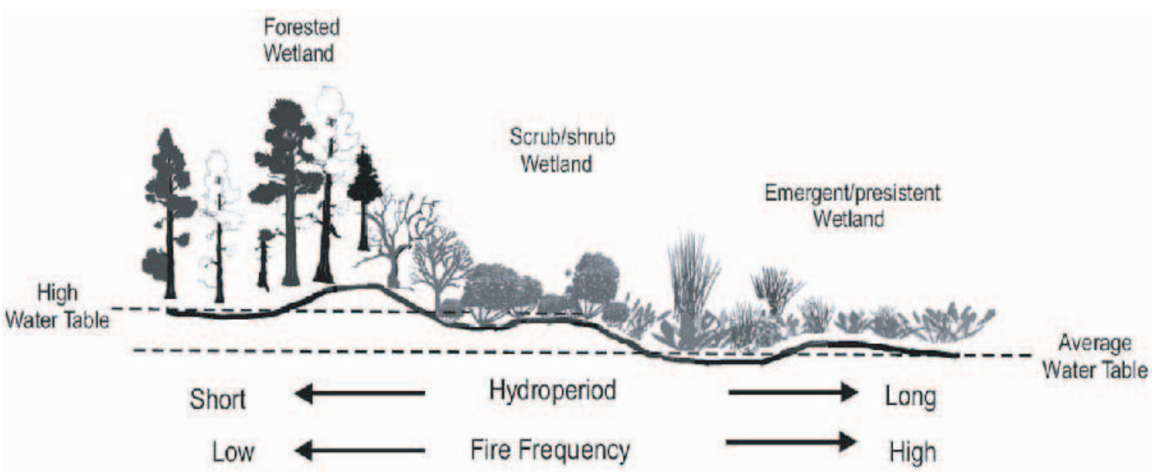


Figure 8.3—Idealized interrelationships between fire frequency, hydroperiod, and organic soil depth in a pocosin wetland in North Carolina. (Adapted from Kologski 1977).



Wetland Type	Forested				Scrub/shrub		Emergent, persistent		
Hydroperiod	Temporarily and seasonally flooded				Temporarily and seasonally flooded; saturated		Temporarily, seasonally and semipermanently flooded; intermittently exposed		
Vegetation Type	Black Spruce	Atlantic White Cedar	Pond Pine Woodland	Pitch Pine	Low Pocosin	High Pocosin	Freshwater Marsh/ Sawgrass	Prairie Pothole	Wet Prairie
Fire Return Interval (yrs)	50-100	25-300	2-10	20-50	13-50	25-50	3-5	5-10	2-6
Citation	Heinselman, 1983; Viereck, 1983	Wade and others 2000	Little, 1946, 1973	Christensen and others, 1988	Sharitz and Gresham, 1998	Sharitz and Gresham, 1998	Wade and others, 1980	Wright and Bailey, 1982	Connelly and Kauffman, 1991

Figure 8.4—Fire frequency, hydrological, and ecological characteristics of palustrine wetlands.

Figure 8.5—Emergent wetland in Agassiz Wildlife Refuge, Minnesota: (top) fall prescribed burning, (bottom) postburn ground surface. (Photos by USDA Forest Service).

and soil temperatures increase site productivity (Viereck 1983) (fig. 8.6).

Fuel accumulation in wetlands is a function of productivity and decomposition. Productivity is a function of hydroperiod and hydrodynamics, which regulate the inflows and outflows of water, nutrients, and oxygen. The seasonal or temporary flooding that is common in tidal marshes and alluvial forest wetlands

Figure 8.6—Black spruce/feather moss forested wetland, Tetlin Wildlife Refuge, Alaska: (top) prescribed burning (7/1993); (middle and bottom) postburn ground surfaces. (Photos by Roger Hungerford and James Reardon).

provides external inputs of water and nutrients and results in high productivity and high decomposition rates. In contrast, the long, stable hydroperiod and dependence on precipitation for water and nutrients common in bogs, pocosins, and other nutrient-limited wetlands result in lower productivity and slower decomposition rates.

On the nutrient-limited wetlands with long, stable hydroperiods, greater amounts of fuel accumulate because of slower decomposition rates in the relatively nutrient-poor litter and fine fuels. This slower decomposition rate leads to more frequent fires on the nutrient-limited sites compared to the productive sites that have inherently less litter and fuels production (Christensen 1985).

Surface Fire in Wetlands—In wetland soils, the effects produced by surface and ground fires are related to the intensity and duration of the energy at either the mineral soil surface or the interface between burning and nonburning organic soil materials. In general, surface fires can be characterized as short duration, variable intensity sources of energy, while ground fires can be characterized as longer duration and lower intensity sources of energy (see chapter 1). Fire severity, which is a measure of the immediate fire effects on plants and soils, is the result of both the intensity and duration of the flaming combustion of surface fires and the smoldering combustion of the organic materials (Ryan and Noste 1985, Ryan 2002).

Following ignition, surface fire behavior depends on weather, topography, and fuels (Albini 1976, Alexander 1982, Finney 1998). Behavior is characterized by parameters that include intensity, rate of spread, and flame geometry. One factor of significance in the prediction of surface fire behavior is fuel compactness (packing ratio). Compactness is the relationship between the amount of fuel in each size class (fuel loading) and the physical volume it occupies (fuel depth), and it is correlated with rate of spread and intensity (Burgan and Rothermel 1984). The compactness ratio of fuel loading to fuel depth reflects the structure and composition of the vegetation (Brown 1981).

Wetland vegetation types range from short sedge meadows to forested wetlands. The surface fuels in graminoid-dominated or scrub-shrub wetlands are primarily vertical, and these fuel types rapidly increase in depth as loading increases. In contrast, the fuels in forested wetlands, which are dominated more by overstory species, are primarily horizontal, and these fuel types slowly increase in depth as fuel loading increases (Anderson 1982). The fuel loading in many wetlands is a product of both the horizontal fuel component of the understory and the vertical component of the overstory.

Physiography and vegetation structure are important factors influencing localized surface fire behavior (Foster 1983). A wide range of fire behavior occurs in wetlands due to the diverse wetland plant communities and their associated fuel types. Fires in hardwood swamps will typically burn with low intensities and low flame lengths, while fires in scrub-shrub wetlands can exhibit extreme behavior (Wade and Ward 1973). Measurements of pocosin scrub-shrub fuel loading and depths show characteristics similar to the chaparral vegetation type with higher loading of fine live and dead materials (Scott 2001). In pocosins, high fuel loads in the small size classes and the presence of volatile oils and resins in the vegetation contribute to high rates of spread and high intensities, making suppression difficult (Anderson 1982).

Ground Fire in Wetlands—Due to the presence of large amounts of organic material, ground fires are a special concern in Histisols, Gelisols, and other soil types with thick organic horizons. Surface fires that result from periodic fluctuations in the water table are correlated with weather cycles. These fires generally consume the available surface fuels but little organic soil material (Curtis 1959). However, fires that occur during longer or more severe water table declines result in significant consumption of both surface fuels and organic soil material (Johnson 1992, Kasischke and Stocks 2000).

Sustained combustion and depth of burn are correlated with soil water content, which is a function of soil water holding capacity, hydroperiod, and evapotranspiration. While soil moisture and aeration in wetland landscapes are controlled by hydrology, the capacity of soil to store water is influenced by the organic matter content and its degree of decomposition. Soil water storage capacity is correlated with physical properties that include bulk density, organic matter content, and hydraulic conductivity. Slightly decomposed fibric materials in soils dominated by sphagnum moss material have low bulk density, high hydraulic conductivity, high porosity, and hold a high percentage of water at saturation. At saturation this fibric material can hold greater than 850 percent of its dry weight as water (USDA Natural Resources Conservation Service 1998). In contrast, the highly decomposed sapric materials found in soils formed in the pocosin wetlands have greater bulk density, finer pore structure, and lower hydraulic conductivity. At saturation, sapric material can hold up to 450 percent of its dry weight in water (USDA Natural Resources Conservation Service 1998).

Following the initial ignition of organic soil material, the probability of sustaining smoldering combustion (that is, ground fire) is a function of moisture and mineral content. Laboratory studies of the relationship between these two factors were conducted using a standardized ignition source and commercial peat moss as an organic soil/duff surrogate. Peat moss was used because of its physical and chemical uniformity.

The results show an inverse relationship between the smoldering moisture limit and moisture and inorganic content. Consequently, these results were used to derive a sustained smoldering probability distribution based upon moisture and mineral content factors. For comparison purposes, the results are reported at the moisture content for a 50 percent probability level of sustained combustion (Frandsen 1987, Hartford 1989) (table 8.4).

Organic soil materials from various wetlands show a similar relationship between smoldering moisture limit and moisture and mineral content. The smoldering limit of low mineral content soils from a North Carolina pocosin site is greater than the smoldering limit of high mineral content soils from a Michigan sedge meadow site. The results also show soil depth differences in smoldering limits. Smoldering limits decline with increasing depth and increasing mineral content in black spruce/sphagnum, sedge meadow, and other sites (Frandsen 1997) (fig. 8.7).

Although the smoldering limits for both peat moss and organic soil materials are the function of mineral and moisture contents, differences exist between the predicted smoldering limits derived from the peat moss surrogate and the limits derived from organic soil material. With the exception of samples from black spruce/feather moss sites in Alaska, at the 50 percent probability level, organic soil materials from other sites smoldered at consistently higher moisture contents than the limits predicted for peat moss. Frandsen (1997) speculated that ignition methods were partially responsible for some of the difference.

Fire line intensity alone does not give the total heat input to the soil surface because it does not take into account the residence time of the flaming combustion or smoldering and glowing combustion (Peter 1992, Albini and Reinhardt 1995). The long duration of smoldering combustion is an important characteristic that differentiates the effects of ground fires and surface fires (Wein 1983).

In laboratory studies, the rate of smoldering combustion depended on both moisture and mineral content (Frandsen 1991a,b), with the laboratory estimates of smoldering rates in agreement with the 1.2 to 4.7 inches/hour (3 to 12 cm/hour) reported by Wein (1983). The results of laboratory studies of the relationship between consumption and moisture content have shown that sustained smoldering of sphagnum moss was possible at up to 130 percent moisture content, and that the percent consumption of organic material was linked to the variability in moisture distribution (Campbell and others 1995).

Table 8.4—Moisture limits of sustained combustion and physical properties of wetland soil materials.

Vegetation type	Wetland class	Average inorganic content	Average organic bulk density	Depth		Moisture content for 50 percent probability of sustained combustion
		Percent	*kg/m³*	*cm*	*in*	*Percent*
Sphagnum (upper)	Forested Wetland	12.4	22	0-5	0-2	118
Sphagnum (lower)	Forested Wetland	56.7	119	10-25	25-64	81
Spruce/Moss	Forested Wetland	18.1	43	0-25	0-64	39
Spruce/Moss/Lichen	Forested Wetland	26.1	56	0-5	0-2	117
Sedge Meadow (upper)	Emergent Wetland	23.3	69	5-15	2-6	117
Sedge Meadow (lower)	Emergent Wetland	44.9	92	15-25	6-64	72
White Spruce Duff	Upland	35.9	122	0-5	0-2	84
Peat	Peat Moss	9.4	222	17-25	7-64	88
Peat Muck	Peat Moss	34.9	203	12-20	5-8	43
Sedge Meadow (Seney)	Emergent Wetland	35.4	183	17-25	7-64	70
Pine Duff (Seney)	Upland	36.5	190	0-5	0-2	77
Spruce/Pine Duff	Upland	30.7	116	0-5	0-2	101
Grass/Sedge Marsh	Wetland	35.2	120	0-5	0-2	106
Southern Pine Duff	Upland	68.0	112	0-5	0-2	39
Pocosin	Shrub-Scrub	18.2	210	10-30	25-12	150

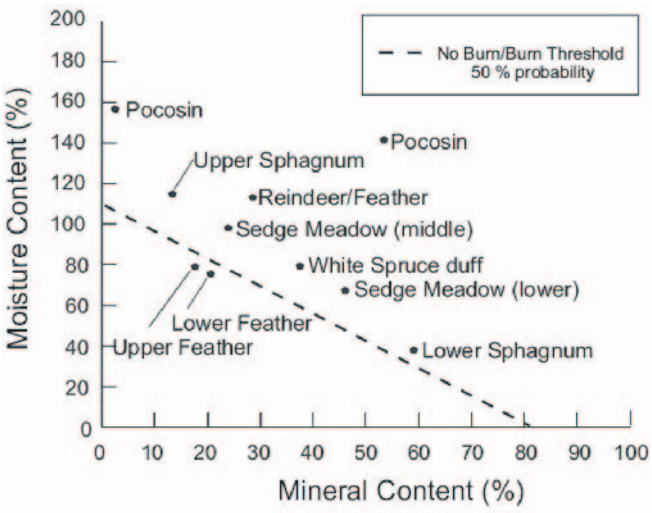


Figure 8.7—Comparison of the predicted moisture smoldering limits of organic soil materials in different wetlands with the smoldering limit of an organic soil surrogate. The dotted line is the 50 percent probability of burn limit. (Adapted from Hungerford and others 1995b).

Heat transfer in organic and mineral soils is a function of the energy produced by combustion and soil thermal properties. The energy produced by flaming and smoldering combustion depends on fuel properties and environmental conditions. Soil thermal properties are affected by soil characteristics and moisture content. Changes in thermal properties are primarily the result of soil drying in advance of the smoldering front (Peter 1992, Campbell and others 1994, 1995).

In addition to fire behavior characteristics, the depth of burn is an important factor in predicting and interpreting soil heating and the effects of fire on wetland soils. In mineral soil, the interface between the burnable organic materials and unburnable mineral soil is significant because of differences in nutrient pools and heat transfer properties. In wetland soils, the depth of burn, which is the interface between burning and nonburning organic soil materials, is important because it reflects the amount of organic material consumed. A number of factors are correlated with depth of burn including nutrient volatilization, ash deposition, and emissions.

Fall prescribed burns were conducted by the Nature Conservancy in pocosin scrub-shrub wetlands in 1997 and 1998 at Green Swamp, North Carolina. The soils were Histosols that were greater than 3 feet (0.9 m) deep. During average years the study site is temporally flooded a number of times, and the soils remain saturated most of the year. The vegetation and surface fuels of the burn units were dominated by shrubs species (*Lyonia lucida* and *Ilex glabra*), scattered pond pine (*Pinus serotina*), and evergreen bays (*Persia borbonia, Gordonia lasianthus*). Fuel and soil moisture variability was influenced by the hummock and depression microtopography of the ground surface. The surface fires in both burns were active to running surface fires with passive crowning that was associated with single or small clusters of pond pine and evergreen bay species (Hungerford and Reardon, unpublished data).

The 1997 fall prescribed burn was conducted during a period of extended drought and dry soil conditions (fig. 8.8, top). At the time of burning, the water table depth was greater than 28 inches (71 cm) from the

Figure 8.8—Prescribed burn of a pocosin scrub-shrub wetland conducted with dry soil conditions, Green Swamp, North Carolina (September 1997): smoldering combustion (Top, photo by Gary Curcio); postburn surface, (Middle, photo by Roger Hungerford); postburn hydrologic and species changes (Bottom, photo by James Reardon).

ground surface of the depression areas. Soil moisture in the upper profile ranged from 85 percent at the surface to 144 percent at 6 inches (15 cm). These moisture contents in the upper soil profile were less than the moisture limits predicted to sustain smoldering in pocosin soils. The moisture contents in the lower soil profile (greater than 18 inches or 46 cm) were greater than 200 percent and exceeded the moisture limits predicted to sustained smoldering (table 8.4).

The surface fire and extended smoldering resulted in significant consumption of soil organic material and surface fuels (fig. 8.8, middle). After the surface fire, the smoldering front advanced both laterally within the upper soil profile and downward. Smoldering in the downward direction was limited by soil moisture of the lower soil profile. The depth of burn varied between 18 and 24 inches (46 to 61 cm) and reflected both the soil moisture distribution within the soil profile and the hummock and depression microtopogaphy common in pocosin wetlands (fig. 8.9).

Soil heating below the burn/no-burn interface was primarily the result of the long duration temperature pulse associated with the smoldering environment. Maximum measured temperatures 0.2 inch (0.5 cm) below the burn/no-burn interface were greater than 680 °F (360 °C), while maximum temperature measured 2.6 inches (6.5 cm) below the burning interface in unburned soil was less than 158 °F (70 °C) (fig. 8.10).

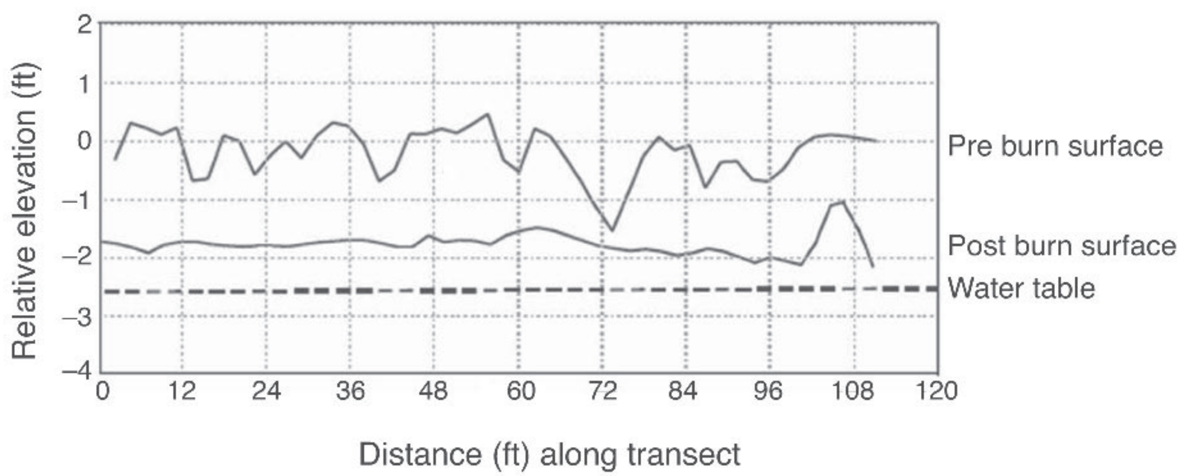


Figure 8.9—Pre- and postburn microelevation transect of a pocosin dry burn study site (Hungerford and Reardon, unpublished data).

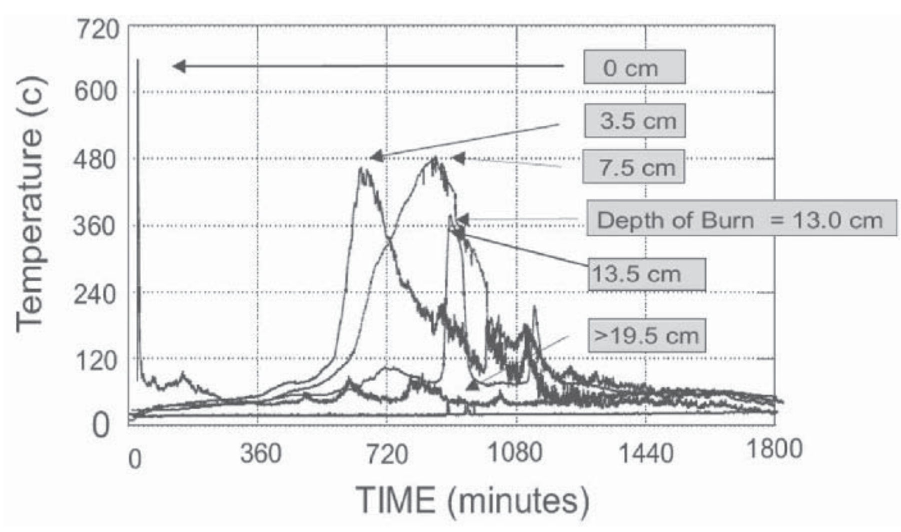


Figure 8.10—Soil profile temperature measurements from a pocosin prescribed burn conducted with dry soil conditions (Hungerford and Reardon, unpublished data).

In areas of high burn severity, the hydrologic changes associated with the consumption of organic soil have led to species composition changes (fig. 8.8, bottom).

The 1998 fall prescribed burn was conducted during a period of saturated soil conditions (fig. 8.11, top). This burn resulted in limited consumption of surface fuels and organic soil materials and limited soil heating (fig. 8.11, middle). At the time of burning, soil moisture in the upper soil profile exceeded 250 percent, and the water table was within 2 to 5 inches (5.1 to 12.7 cm) of the depression ground surface. The depth of burn was limited, and the consumption of organic soil material was restricted to the top of hummocks and microsites with dryer soil conditions. Litter and surface fuels were scorched but not consumed in much of the unit (fig. 8.11, bottom). Soil heating below the burn/no-burn interface was limited and resulted from the flaming combustion primarily associated with the surface fire. The maximum measured temperatures of unburned soil at 2.8 and 4.3 inch (7.0 and 11.0 cm) depths were 171 °F (77 °C) and 189 °F (87 °C), respectively (fig. 8.12). The removal of dead and live fine fuel resulted in no significant species composition changes and regrowth, and fuel accumulation was sufficient to reburn the unit in autumn 2001.

Figure 8.11—Prescribed burn of pocosin scrub-shrub wetland conducted with wet soil conditions, Green Swamp, North Carolina (September 1998): (top) surface fire behavior, (middle and bottom) fuel consumption and postburn ground surface. (Photos by Roger Hungerford and James Reardon)

The fire severity in wetlands is correlated with depth of burn. In addition to the direct effects caused by organic soil combustion, the physical removal of the soil material creates a number of interrelated physical, biological, and hydrological consequences. Removal of this material can result in soil moisture, temperature, and aeration changes that affect microorganism activity changes and ultimately decomposition rate changes (Armentano and Menges 1986).

In boreal wetland systems, the active and permafrost layers depend on the presence of an insulating moss layer (Viereck 1973, Van Cleve and Viereck 1983). A dynamic relationship exists between the depth to permafrost, organic layer thickness, and soil temperature. Removal of this insulating organic layer by burning leads to an increase in soil temperatures, changes in soil moisture regime, and an increase in depth to permafrost. Over time, successional processes and the reestablishment of the organic layer will cause a decrease in soil temperatures and a decrease in the depth of the permafrost layer (Ping and others 1992) (fig. 8.13).

Responses to litter consumption and soil exposure after spring burning of sedge meadow wetlands in Michigan also result in changes in soil temperature

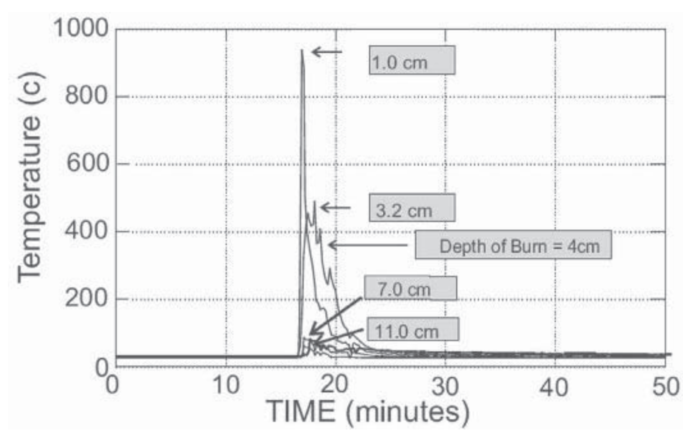


Figure 8.12—Soil profile temperature measurements from a pocosin prescribed burn conducted with wet soil conditions (Hungerford and Reardon, unpublished data).

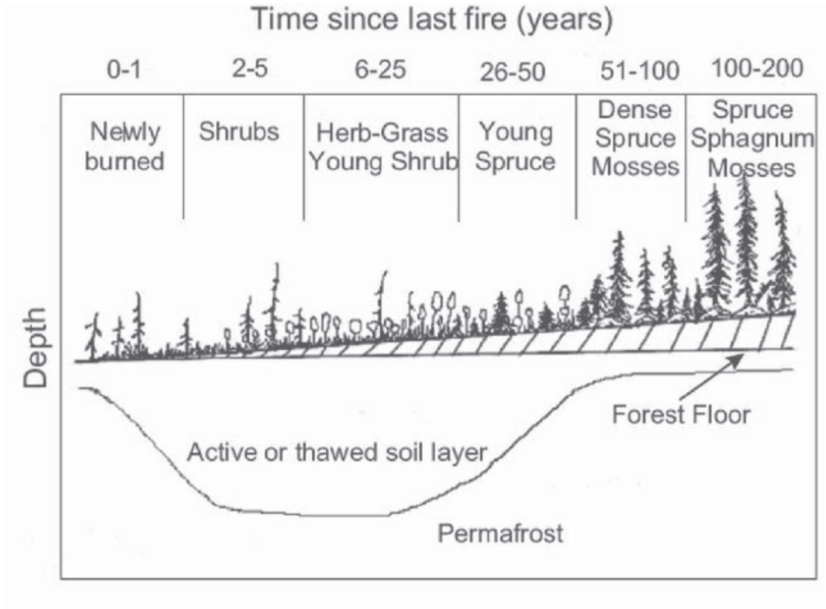


Figure 8.13—Fire cycle effects on soil microclimate, permafrost, and vegetation succession in Alaska. (adapted from Van Cleve and Viereck 1983, Ping and others 1992).

regime. Prescribed burns were conducted over a range of fire intensities and resulted in the removal of the litter and a blackened soil surface. The postburn soil surface had higher soil temperatures and produced increased germination rates of *Carex strica* (Warners 1997).

Wein (1983) suggested that low intensity fires favor bog-forming processes because the thick organic layers that remain after low intensity fires decompose slowly and cause temperature and moisture changes by insulating the soil and retaining moisture. In contrast, deep hot fires resulted in increased soil temperatures and nutrient cycling rates. These conditions favor plant communities dominated by vascular plants.

Wetland Fire Effects and Soil Nutrient Responses

Soil nutrient responses are the result of a number of processes including nutrient volatilization, condensation of combustion products on cool soil surfaces, ash deposition, and soil heating. Soil nutrients exhibit a wide range of sensitivity to temperature changes. Nitrogen-containing compounds are the most heat-sensitive and show changes at temperatures as low as 392 °F (200 °C) while cations such as magnesium (Mg) and calcium (Ca) are less sensitive and show changes at temperatures greater than 1,832 °F (1,000 °C) (see chapter 3; DeBano and others 1998).

Nutrient volatilization is the result of flaming and smoldering combustion and nutrient temperature sensitivity. The amount of nutrients lost to the atmosphere or remaining in the ash layer are dependent on combustion temperatures and preburn nutrient content of the fuels (Raison 1979, Soto and Diaz-Fierros 1993). Laboratory studies of smoldering combustion were conducted with wetland organic soil cores from black spruce/ feather moss (Alaska) and sedge meadow (Michigan) sites. Different depth of burn treatments were simulated by controlling soil core moisture. The results demonstrated that ash differences in total carbon (C) and total nitrogen (N) content produced by smoldering combustion reflected both depth of burn treatment and preburn nutrient levels. The ash and char residue derived from the deep burn treatments showed the largest reductions in total N and C (Hungerford and Reardon unpublished data) (fig. 8.14).

In comparison with the flaming combustion of aboveground biomass, the smoldering of high N content organic soils from pocosin scrub-shrub wetlands in North Carolina produced larger ammonia (NH_3) and hydrocarbon emissions. Smoldering also produced a "tar-like" substance that condensed on cool surfaces. This material had a total N content that was six to seven times greater than the parent material (Yokelson and others 1997).

In addition to nutrient volatilization, combustion temperatures, and preburn fuel nutrient content, the

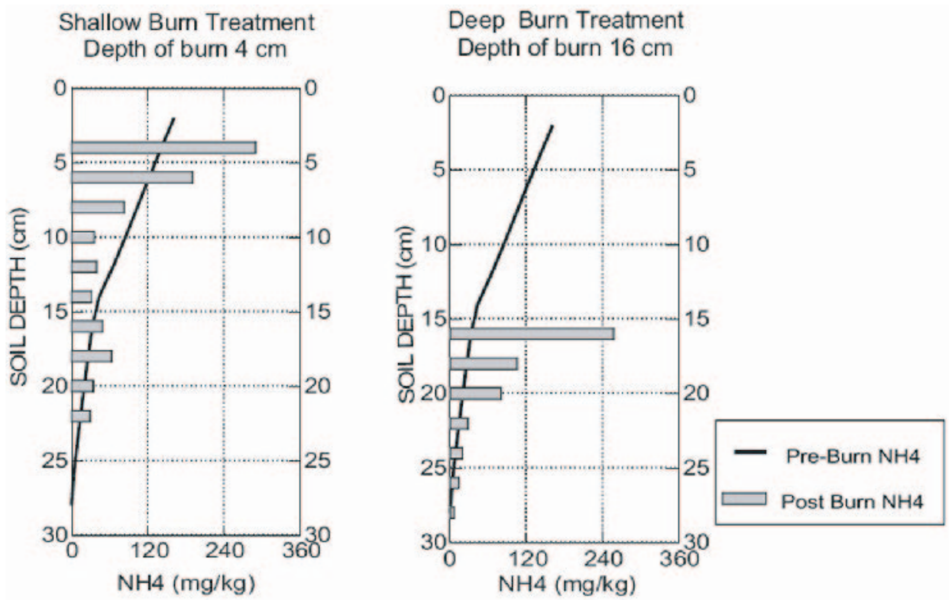


Figure 8.14—Comparison of preburn and postburn NH_4–N distribution from the laboratory burning of spruce/feather moss soil cores (Hungerford and Reardon, unpublished data).

significance of ash deposition is also dependent on the interaction between fire and hydrology. Postburn spatial and temporal water movement dynamics are the result of surface microtopography changes and reflect the extent and depth of burn. In wetlands, the duration and magnitude of the postburn ash-derived nutrient pulse is a function of preburn nutrient levels, burning conditions, and seasonal waterflow.

The significance of the relationship between ash deposition and increased nutrient availability was tested in Michigan *Carex stricta* dominated sedge meadow sites. No significant increase in productivity due to ash additions was found (Warners 1997). The loss of ash material due to water movement was not a factor in this study, and the results suggest that either productivity was not limited by the nutrients present in the ash residue or the ash produced from burnt sedge meadow litter was nutrient poor (Warners 1997).

Nitrogen and phosphorus (P) additions were made to a brackish marsh site in coastal North Carolina at rates intended to simulate ash deposition from burning. The nutrient additions were made along a salinity and hydroperiod gradient. A significant productivity increase to low levels of N and P addition was observed at intermediate salinity levels (Bryant and others 1991). In contrast to the Michigan sedge study (Warners 1997) these findings suggest that nutrient release from burning may stimulate growth.

Soil and tissue nutrient concentrations from *Carex* spp. dominated meadows showed complex interrelationships between nutrient concentrations, water depth, and fire incidence. Fire incidence and water depth were negatively correlated with plant tissue N, P, potassium (K), copper (Cu), manganese (Mn), and zinc (Zn) concentrations. The results suggest that volatilization and loss of these nutrients by snowmelt and runoff after spring fires may result in losses of these elements (Auclair 1977).

The site-specific studies of wetland systems presented in this chapter have shown mixed results in the relationship between burning and nutrient loss and retention. The interelationships of nutrient loss, season of burning, and wetland system are not well understood, and generalizations across wetland systems and classes are limited at this time.

Nutrient Transformations and Cycling—In nutrient deficient wetlands with low decomposition rates, the increased availability of nitrate nitrogen (NO_3-N), ammonium nitrogen (NH_4-N), and phosphate (PO_4) is linked with burning because a high percentage of soil nutrients are stored in organic sediments and plant material (Wilbur and Christensen 1983). Nutrient transformations that result from soil heating are primarily dependent on soil temperatures (DeBano and others 1998). The destructive distillation of organic materials occurs at relatively low temperatures 392 to 572 °F (200 to 300 °C) (Hungerford and others 1991), and soil temperatures above 482 °F (250 °C) result in a decreases in available nutrients (Kutiel and Shaviv 1992).

Wetland soil nutrient cycling processes are similar to those of terrestrial environments, but oxygen (O_2) limitations in the wetland soil environment lead to important differences. Anaerobic reactions require both a supply of organic C and a soil saturated with slow moving or stagnant water. In this environment, aerobic organisms deplete the available O_2, and anaerobic organisms utilize other compounds as electron acceptors to respire and decompose organic tissues (Craft 1999). Soil organisms dominant in aerobic soils do not function as efficiently in low O_2 environments (Stevenson 1986), and soil chemical transformations are dominated by reduction reactions. Dominant reactions taking place in the soil include the denitrification of NO_3-N, the reduction of Mn, iron (Fe), sulfur (S), and methane production. These reactions depend on the presence of electron accepting compounds (such as NO^{-3}, Fe^{+3} Mn^{+4}, SO^{-4}), temperature, pH, and other factors (Craft 1999).

Aerobic conditions may dominate a shallow surface layer of waterlogged soils. The thickness of this layer depends on hydrology and depth of burn. The chemistry of the remaining soil profile is dominated by reduction reactions that result from the anaerobic or O_2 limiting conditions.

Burning has numerous direct influences on nutrient cycling processes. The removal of vegetation and litter material changes the surface soil moisture and temperature dynamics of bare soils, leading to changes in microbial composition and activity. Altered microbial activity can result in nutrient cycling changes in nitrogen mineralization, nitrification rates, and phosphorous mineralization rates.

Wetland Soil Nutrients—Carbon is stored in both living and dead plant materials. In wetland soils the percentage of the total C stored in dead materials is greater than in other terrestrial systems. Because the C sequestered in wetlands contains approximately 10 percent of the global C pool, wetlands play an important role in the global C cycle (Schlesinger 1991). The C balance in wetlands is sensitive to land management and environmental factors (Trettin and others 2001) and is a function of numerous interrelated factors including productivity, decomposition, and fire frequency.

Factors influencing C accumulation rates were studied in sphagnum-dominated boreal peatlands in Western Canada (Kuhry 1994). The results showed that increased frequency of peat surface fires led to decreases in C accumulation rates and organic layer thickness. Kuhry (1994) concludes that in these systems, the increased short-term productivity from postburn nutrient release may not compensate for the peat lost from frequent burning.

In contrast to sphagnum-dominated boreal systems, the burning of forested wetlands in the Southeastern United States can lead to increased C accumulation rates. The loss of overstory vegetation that results from burning produces changes in the balance between transpiration from vegetation and surface evaporation from exposed soil. Evapotranspiration changes leading to higher water levels and reduced soil O_2 limitation may cause decreased decomposition rates and increased C accumulation rates (Craft 1999).

In freshwater wetlands, N is present in both organic and inorganic forms. Organic N is associated with plants, microbes, and sediments, while inorganic N is associated primarily with water and sediments (Craft 1999). Important transformations of available N involve the diffusion of NH_4-N between aerobic layers where mineralization and nitrification dominate N cycling and anaerobic layers where denitrification dominates N cycling (Patrick 1982, Schmalzer and Hinkle 1992).

Laboratory burning of wetland soil cores showed the effects of smoldering combustion on the distribution of available N. Different depth of burn treatments were simulated by manipulating soil core moisture. Comparisons of pre- and postburn available N distributions showed postburn NH_4-N enrichment below the ash deposition layer and the burn/no-burn interface (Hungerford and Reardon unpublished data). The findings support the conclusions of DeBano and others (1976) and suggest that the increase in available N was caused by soil heating and/or the condensation of N-rich combustion products on unburned soil surfaces (fig. 8.15).

Phosphorous limitation is common in wetlands because unlike N and C, it has no significant biologically induced or atmospheric inputs (Paul and Clark 1989). In wetlands the vegetation and organic sediments are the major storage sites for P, and cycling in wetlands is dominated by vegetation and microbes (Craft 1999). Phosphorus is present in wetland soils in a number of forms: organic forms in live and partially decomposed plant materials, in mineral form bound to aluminum (Al), Fe, Ca, Mg, and in orthophosphate compounds (Craft 1999).

The spring burning of pocosin scrub-shrub wetland sites produced immediate increases in NO_3-N, NH_4-N, and PO_4. Surface temperatures were moderate, and only limited smoldering combustion was observed. Postburn NO_3-N levels increased and remained high throughout the 18-month study. Postburn PO_4 concentrations were initially elevated but returned to preburn levels by the end of the first growing season. The NH_4-N levels were high throughout the first growing season following burning but returned to prefire levels by the end of that growing season. The limited soil heating from the surface fire and additional results from laboratory soil incubation studies suggested that postburn N and P nutrient increases were primarily the result of ash deposition (Wilbur 1985).

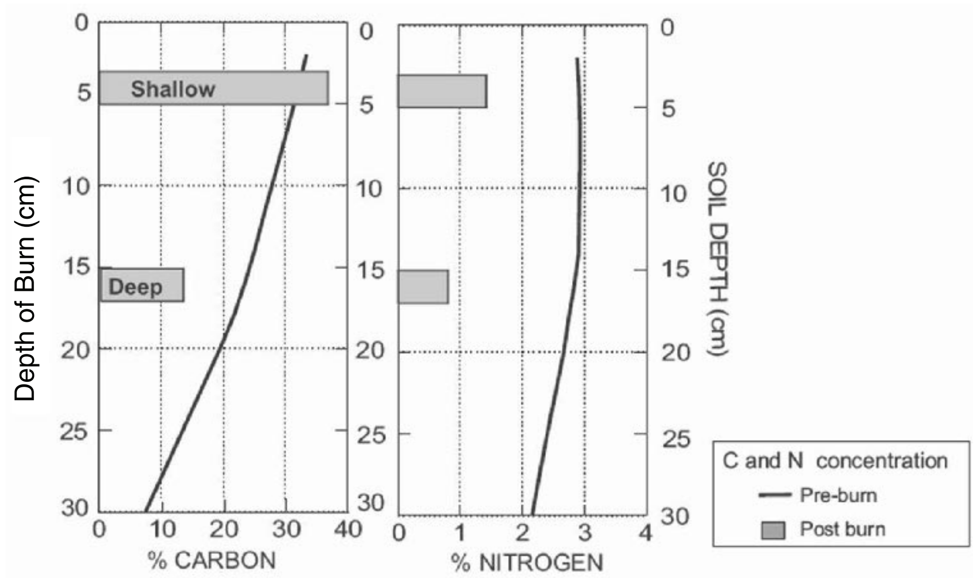


Figure 8.15—Comparison of the preburn and postburn percent C and N in ash and char material from the laboratory burning of black spruce/feather moss soil cores (Hungerford and Reardon, unpublished data).

The spring burning of *Juncus* spp. and *Spartina* spp. marshes in central Florida resulted in significant soil nutrient changes. Increases in PO_4 and soil cations were attributed to ash deposition (Schmalzer and Hinkle 1992). In contrast, the results of winter burning of *Juncus* and *Spartina* marsh communities on the Mississippi Gulf Coast suggested that increases in extractable P were the result of mild heating of organic sediments, and the retention of ash-derived nutrients may be a function of meteorological factors, tidal regimes, and topography (Faulkner and de la Cruz 1982).

Ammonium nitrogen levels of the *Juncus* and *Spartina* Florida marsh study sites were elevated for a period after the burn when the water table declined and soil temperatures increased. The results suggest that the NH_4-N changes were the result of increased soil O_2, soil temperatures, and higher N mineralization rates. Postburn soil NO_3-N levels were high and eventually declined because of a return of anaerobic conditions 6 to 9 months after burning. The authors suggest that NO_3-N declines were the caused by either decreases in nitrification or increases in denitrification rates. Soil heating was limited by standing water during the burning, and the results suggest that available nitrogen nutrient responses were linked with seasonal hydrology.

The role of fire in the biogeochemical cycles of trace metals is not well understood, and much research is currently being conducted in this area. Organic sediments accumulate metals such as mercury (Hg), arsenic, and selenium (Se) as a result of environmental processes including atmospheric deposition and water movement from upstream sources. The origins of these trace metals range from natural processes such as geochemical weathering to anthropogenic processes such as fossil fuel burning and agricultural runoff. These metals are transformed by microorganisms and enter the food web in wetland environments in methylated forms (Stevenson 1986). The accumulation and release of these metals is of concern because they are toxic to many organisms.

Selenium accumulation is of concern in wetlands in a number of Western States due to widespread geological sources. Wetland plants such as cattails and bullrush can accumulate Se, and the decomposition of the dead plant material produces organic sediments with large amounts of the element. The effect of burning on Se volatilization from wetland plants was studied at Benton Lakes National Wildlife Refuge in Montana (Zang 1997). The result showed that burning of wetland plants can volatilize up to 80 percent of the Se in leaves and stems, and the author suggests that fire may be an effective method of reducing the selenium concentrations of wetland sediments.

Mercury accumulation is a growing concern in wetland management. It is the result of atmospheric deposition and fossil fuel combustion. Inorganic Hg in sediments is transformed to methylated forms by microbial activity. The methylated forms of Hg are highly bioaccumlative and are readily incorporated into the food web. The effects of fire and prolonged drying on methylation rates were studied in the Florida Everglades (Krabbenhoft and others 2001). The results showed that burning and prolonged drying changed soil and ground water properties and resulted in Hg methylation rate increases. These rate increases were linked with postburn sulfate availability during reflooding. Increases in the average hydroperiod may result in a decrease in the occurrence and magnitude of conditions linked with high methylation rates.

Wetland Management Considerations

The wide distribution and critical ecological functions that are provided by wetlands make their continued health crucial to environmental quality. Contrary to the general perceptions of wetlands, ecological and paleoecological evidence suggests that creation and maintenance of these important ecosystems depends on fire and other disturbances. Land managers dealing with wetland ecosystems need to understand the ecological role of fire. Experience gained by the renewed use of prescribed fire and prescribed natural fire in wetlands will improve our understanding of important wetland processes and vegetation development. Increased knowledge will enable us to deal more effectively with wetland management challenges such as habitat restoration, threatened and endangered species management, and hazardous fuel reduction.

Riparian Ecosystems

Riparian areas are an integral and important component of watersheds throughout most of the United States (Baker and others 1998, Brooks and others 2003). They occur in both arid and humid regions and include the green-plant communities along the banks of rivers and streams (National Academy of Science 2002) (fig. 8.16). Riparian areas are found not only along most major waterways in arid and humid areas throughout the United States, but they are also important management areas on small perennial, ephemeral, and intermittent streams.

Riparian areas are a particularly unique and important part of the landscape in the Western United States where they represent the interface between aquatic and adjacent terrestrial ecosystems and are made up of unique vegetative and animal communities that require the regular presence of free or unbound water. Riparian vegetation in the United States ranges widely and includes high mountain meadows, deciduous and evergreen forests, pinyon juniper and encinal woodlands, shrublands, deserts, and desert grasslands. In most watersheds, riparian areas make up only a small percentage of the total land area (for example, only about 1 percent or less in the Western United States), yet their hydrologic and biological functions must be considered in the determination of water and other resource management goals for the entire watershed.

Figure 8.16—Examples of riparian areas: Gila River, New Mexico (Top, photo by Daniel Neary); Verde River, Arizona (Middle, photo by Alvin Medina); Black River, Arizona (Bottom, photo by Alvin Medina).

Riparian areas have high value in terms of diversity because they share characteristics with both adjacent upland and aquatic ecosystems (Crow and others 2000). The kinds of biological and physical diversity vary widely, spatially and temporally, and thereby contribute greatly to the overall diversity of numbers, kinds, and patterns in the landscape and waterscape ecosystems along with a multitude of ecological processes associated with these patterns (Lapin and Barnes 1995). Also, riparian areas support a variety of plants and animals that are not found elsewhere, including threatened and endangered species. They play other important, but less obvious, roles relating to enhanced water quality, flood peak attenuation, and reduced erosion and sediment transport.

Riparian Definition and Classification

Riparian areas can be defined in many ways depending upon individual viewpoints or purposes. These approaches can reflect agency concepts, the disciplines involved, or the particular functional role that a riparian area plays in the total ecosystem (Ilhardt and others 2000). For example, a technical definition developed by the Society of Range Management and the Bureau of Land Management (Anderson 1987, p. 70) is:

> A *riparian area* is a distinct ecological site, or combination of sites, in which soil moisture is sufficiently in excess of that otherwise available locally, due to run-on and/or subsurface seepage, so as to result in an existing or potential soil-vegetation complex that depicts the influence of that extra soil moisture. Riparian areas may be associated with lakes; reservoirs; estuaries; potholes; springs; bogs; wet meadows; muskegs; and intermittent and perennial streams. The characteristics of the soil-vegetation complex are the differentiating criteria.

Disciplinary definitions are oriented to the particular disciplines involved, whereas functional definitions focus on using the flow of energy and materials as their basis rather than being based on static state variables.

Classifying riparian systems has involved schemes of hierarchical approaches based on geomorphology, soils, moisture regimes, plant succession, vegetation, and stream channels, or combinations of these parameters. Resource managers are increasingly interested in classifying riparian systems because of their important and unique values. A basic understanding of the ecology of riparian systems is complicated by extreme variation in geology, climate, terrain, hydrology, and disturbances by humans. Geomorphology is especially useful on riparian sites where the natural vegetation composition, soils, or water regimes, or a combination of these, have been altered by past disturbance, either natural or human induced. Other classification schemes are based on stream classification systems such as that developed by Rosgen (1994).

Hydrology of Riparian Systems

Riparian ecosystems are controlled by water, vegetation, soils, and a host of biotic organisms extending in size from microorganisms to large mammals. Water and its hydrologic processes, however, are a central functional force affecting the riparian-watershed system and are the main component that links a particular stream and associated riparian vegetation to the surrounding watershed. The magnitude and direction of the riparian-watershed relationship is further tempered by geology, geomorphology, topography, and climate—and their interactions. This interrelationship between riparian area and surrounding watershed is the most sensitive to natural and human-related disturbances, including fire (DeBano and Neary 1996, DeBano and others 1998).

The important hydrologic processes include runoff and erosion, ground water movement, and streamflow. As the boundary between land and stream habitats, streambanks occupy a unique position in the riparian system (Bohn 1986). Bank and channel profiles affect stream temperature, water velocity, sediment input, and hiding cover and suitable living space for fish. Streamside vegetation on stable banks provides food and shade for both fish and wildlife. As a result, streambank condition and the quality of the fish habitat are closely linked.

Riparian systems in arid environments are hydrologically different from those found in more humid areas. In the arid West, intermittent streamflow and variable annual precipitation are more common than in the more humid areas of the Eastern United States. As a result, riparian systems in humid regions are less dependent on annual recharge by episodic events to sustain flow and are more dependent on a regular and dependable ground water flow accompanied by interflow (subsurface flow) and, to a lesser extent, on overland flow. Watersheds in humid regions also commonly have lakes, ponds, and other systems that sustain flows over much of the year. As a result of variable precipitation, sediment transport is more episodic in the arid West, where pulses of aggradation and degradation are punctuated by periods of inactivity. Unlike on humid watersheds, side-slope erosion in drylands is discontinuous, and there are often long lag periods between watershed events and sediment delivery (Heede and others 1988). Channels in humid regions tend to be more stable over the long term, and the macrotopography of these systems, though still responsive to flooding events, tends to evolve much more slowly.

Riparian Fire Effects

Fire is a common disturbance in both riparian ecosystems and the surrounding hillslopes, and both wildfires and prescribed burns occur in many of these riparian-watershed systems. Fire affects riparian areas both directly and indirectly. The direct effects consist mainly of damage to the vegetation (trees, shrubs, and grasses) that intercepts precipitation, and the partial consumption of the underlying litter layer. The severity of the damage to the riparian vegetation depends upon the severity of the fire, which could consume part or all of the vegetation. Severe wildfires can cause profound damage to plant cover and can increase streamflow velocity, sedimentation rates, and stream water temperatures, as contrasted to low severity, cool-burning prescribed fires, which have less severe consequences. When fire burns the surrounding watershed, the indirect effect on the riparian area is that it decreases basin stability, and in steep erodible topography, debris flows along with dry ravel and small landslides off hillslopes are common. Therefore, the recovery of vegetation following fire reflects the combined disturbance of both the fire and flooding, and together they can impact the time required for revegetation and postfire rehabilitation efforts.

Role of Large Woody Debris

Large organic debris is a major component of watersheds and river systems because of its important role in hydraulics, sediment routing, and channel morphology of streams flowing through riparian systems (Smith and others 1993, DeBano and Neary 1996). An additional benefit of large woody debris is in nutrient cycling and the productivity of forest vegetation. Through time, entire live and dead trees and shrubs, or parts of them, are likely to fall into stream channels within riparian ecosystems. This woody debris increases the complexity of stream habitats by physically obstructing water flow. For example, trees extending partially across the channel deflect the current laterally, causing it to widen the streambed. Sediment stored by debris also adds to hydraulic complexity, especially in organically rich channels that are often wide and shallow and possess a high diversity of riffles and pools in low gradient streams of alluvial valley floors. Stream stabilization after major floods, debris torrents, or massive landslides is accelerated by large woody debris along and within the channel. After wildfire, while the postfire forest is developing, the aquatic habitat may be maintained by large woody debris supplied to the stream by the prefire forests.

Coarse woody debris accumulations are usually larger in the first few years following a wildfire than in prefire years, but they generally decline to below prefire conditions thereafter. The recovery time to prefire conditions ranges from 25 to 300 years depending on the stability of the ecosystem burned. While in stream channels, this added debris can have a beneficial effect on aquatic habitats in the short term by providing structure to the streams. Long-term impacts can be disruptive to stream morphology and the consequent streamflow and sediment transport regimes. Carefully prescribed fire should not affect the accumulations of coarse woody debris either on the watershed or in the channel. Recommended amounts of woody debris to be left on the watershed in Arizona, Idaho, and Montana are given earlier in chapter 4 (see also Graham and others 1994).

Coarse woody debris in the stream channels of burned riparian ecosystems is made up of components of larger individual size, and forms larger accumulations, than that of unburned systems. Postfire debris is also likely to be moved more frequently over longer distances in subsequent streamflow events (Young 1994). Managers frequently debate the merits of leaving burned dead debris in stream channels following fire. Some managers feel that this material could jam culverts and bridges, causing these structures to wash out and cause flooding, and, as a consequence, these managers feel the debris should be removed (Barro and others 1989). However, removal of this debris can also result in changes in channel morphology, a scouring of the channel bed, increases in streamflow velocities and sediment loads, an export of nutrients out of the ecosystem, and a deterioration of biotic habitats. Removal of postfire accumulations of coarse woody debris, therefore, is likely to best be decided on a site-specific basis.

Riparian Management Considerations

Fire is an important factor to consider in the management of riparian ecosystems. Most fire in riparian areas can be intense and cause extensive damage to the vegetation. However, even after severe fires, recovery can be rapid within a couple years to prefire conditions in some environments, but not all. The recovery of vegetation following fire reflects the combined disturbance of both the fire and flooding.

Riparian areas are particularly important because they provide buffer strips that trap sediment and nutrients that are released when surrounding watersheds are burned. The width of these buffer strips is critical for minimizing sediment and nutrient movement into the streams. The best available guidelines for buffer width associated with prescribed fire are that for low intensity fires, less than 2 feet (0.6 m) high, that do not kill stream-shading shrubs and trees. Such fire can be used throughout the riparian area without creating substantial damage. Where fire damages woody vegetation, the width should be proportional to the size of the contributing area, slope, cultural practices in the upslope area, and the nature of the drainage below. A general rule of thumb is that a width of 30 feet (9 m) plus (0.46 x percent slope) be left along the length of the stream to protect the riparian resource (DeBano and others 1998).

Managers need to consider the mixed concerns about leaving downed large woody debris in, or near, channel following fire. On one hand, large woody debris plays an important role in hydraulics, sediment routing, and channel morphology of streams flowing through riparian systems, thereby enhancing these systems. On the other hand, to protect life and property, channel clearing treatments following fire is usually desirable. The USDA Forest Service suggests that the best management practice balances downstream value protection with the environmental implications of the treatment.

Summary

While the connection between wetlands and riparian ecosystems and fire appears incongruous, fire plays an integral role in the creation and persistence of hydric species and ecosystems. Wetland and riparian systems have been classified with various classification systems that reflect both the current understanding of biogeochemical processes and the missions of the individual land management agencies.

This chapter examines some of the complex water inflows and outflows that are balanced by the interrelationships between biotic and abiotic factors. The presence and movement of water are dominant factors controlling the interactions of fire with wetland dynamics, nutrient and energy flow, soil chemistry, organic matter decomposition, and plant and animal community composition.

In wetland and riparian soils, the effects produced by surface and ground fires are related to the intensity of fire at either the soil surface or the interface between burning and nonburning organic soil materials. In general, surface fires can be characterized as short duration, variable intensity ones, while ground fires can be characterized as having longer duration and lower intensity. However, the latter type of fires can also produce high severity fires with profound physical and chemical changes.

Notes

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Chapter 9:

Fire Effects and Soil Erosion Models

Introduction

In many cases, decisions about fire have to be made in short timeframes with limited information. Fire effects models have been developed or adapted to help land and fire managers make decisions on the potential and actual effects of both prescribed fires and wildfires on ecosystem resources (fig. 9.1). Fire effects models and associated erosion and runoff models apply the best fire science to crucial management decisions. These models are undergoing constant revision and update to make the latest information available to fire managers using the state-of-the-art computer hardware and software. Use of these models requires a commitment to understand their assumptions, benefits, and shortcomings, and a commitment to constant professional development.

Figure 9.1—Wildland fires such as the Rodeo-Chediski Fire of 2002 affect the complete range of physical, chemical, and biological components of ecosystems. (Photo by USDA Forest Service).

First Order Fire Effects Model (FOFEM)

FOFEM (First Order Fire Effects Model) is a computer program that was developed to meet the needs of resource managers, planners, and analysts in predicting and planning for fire effects. FOFEM provides quantitative predictions of fire effects for planning prescribed fires that best accomplish resource needs, for impact assessment, and for long-range planning and policy development. FOFEM was developed from long-term fire effects data collected by USDA Forest Service and other scientists across the United States and Canada (fig. 9.2).

Description, Overview, and Features

First order fire effects are those that concern the direct or indirect of immediate consequences of fire. First order fire effects form an important basis for predicting secondary effects such as tree regeneration plant succession, soil erosion, and changes in site productivity, but these long-term effects generally involve interaction with many variables (for example, weather, animal use, insects, and disease) and are not predicted by this program. FOFEM predicts fuel consumption, smoke production, and tree mortality. The area of applicability is nationwide on forest and nonforest vegetation types. FOFEM also contains a planning mode for prescription development.

Applications, Potential Uses, Capabilities, and Goals

FOFEM makes fire effects research results readily available to managers. Potential uses include wildfire impact assessment, development of salvage specifications, design of fire prescriptions, environmental assessment, and fire management planning. FOFEM can also be used in real time, quickly estimating tree mortality, smoke generation, and fuel consumption of ongoing fires.

Figure 9.2—Development of FOFEM and other fire effects models stemmed from long-term fire effects data collected by USDA Forest Service and other scientists across North America. (Photo by USDA Forest Service).

Scope and Primary Geographic Applications

FOFEM—national in scope—uses four geographic regions: Pacific West, Interior West, Northeast, and Southeast. Forest cover types provide an additional level of resolution within each region, and SAF and FRES vegetation types to stratify data and methods. Geographic regions and cover types are used both as part of the algorithm selection key, and also as a key to default input values. FOFEM contains data and prediction equations that apply throughout the United States for most forest and rangeland vegetation types that experience fire.

Input Variables and Data Requirements

FOFEM was designed so that data requirements are minimal and flexible. Default values are provided for almost all inputs, but users can modify any or all defaults to provide custom inputs.

Output, Products, and Performance

FOFEM computes the direct effects of prescribed fire or wildfire. It estimates fuel consumption by fuel component for duff, litter, small and large woody fuels, herbs, shrubs and tree regeneration, and crown foliage and branchwood. It also estimates mineral soil exposure, smoke production of CO, PM10, and PM2.5, and percent tree mortality by species and size class. Alternatively, if the user enters desired levels of these fire effects, FOFEM computes fuel moistures and fire intensities that should result in desired effects.

Advantages, Benefits, and Disadvantages

FOFEM is easy to use, applies to most vegetation types and geographic areas, synthesizes and makes available a broad range of available research results, incorporates planning and prediction modes, and provides a wide range of data in the form of default inputs for different vegetation and fuel types. The main disadvantage is that FOFEM is not currently linked to any other models (fire behavior, smoke dispersion, postfire succession).

System and Computer Requirements

FOFEM version 5.0 is available for IBM-compatible PCs with Windows 98 and Windows 2000 operating systems. FOFEM is supported by the Fire Effects Research Work Unit, Intermountain Fire Sciences

Lab, Missoula, MT 59807. Additional information can be obtained from:

http://www.fire.org/
http://www.firelab.org/

FOFEM includes embedded help and user's information. The current version (5.21) can be downloaded for use with WINDOWS@ 98, 2000, and XP at:

http://www.fire.org/index.php?option=com_content&task=view&id=58&Itemid=25

POWERPOINT@ tutorials provide a FOFEM overview and information for basic and advanced users. FOFEM 4.0 should be replaced with FOFEM 5.21. For more detailed information contact Elizabeth Reinhardt at: ereinhardt@fs.fed.us

Models for Heat and Moisture Transport in Soils

Transfer of heat into the soil beneath a fire produces a large number of onsite fire effects to the physical, chemical, and biological properties of soils (Hungerford 1990) that include:

- plant mortality and injury
- soil organism mortality and injury
- thermal decomposition of organic matter
- oxidation or volatilization of chemical components of the upper soil profile
- other physiochemical changes

To predict the nature and extent of these effects, we need to understand temperature profiles within the soil beneath burned areas (Albini and others 1996). Temperature profiles are rarely measured in actual fires, so some type of model is needed to predict soil temperatures and the response of soils to the thermal input. Albini and others (1996) reviewed a number of existing models to determine their applicability and recommend future development goals.

The Albini and others (1996) review of heat transfer models from the soil science, engineering, and geophysics fields concluded that the only useful models for describing heat transfer phenomena for wildland fires come from the soil science arena. The models of Campbell and others (1992, 1995) seem to function well in predicting temperature histories and profiles of soils heated at rates and temperatures consistent with wildland fires. Their model did not perform as well with soil moisture contents as with temperatures. Because many of the heating effects are a function of soil moisture, this is an important ability for heat transfer prediction models.

Albini and others (1996) identified the omission of a number of important features in the soil science models. These include diffusive transport of water as a vapor or liquid, momentum equations, predictions of the transient movement of phase-change boundaries, lateral nonhomogeneity of soils, and the rapid decline of wetting attraction of liquid water to quartz near 149 °F (65 °C).

Finally, Albini and others (1996) made recommendations for further model development and simplifications of the existing models. They believed that some simplification would improve the use of the existing models without much sacrifice in the fidelity of their predictions.

WEPP, WATSED, and RUSLE Soil Erosion Models

Following a fire, it is often necessary to use some standard prediction technology to evaluate the risk of soil erosion. For forests that tend to regenerate rapidly, the risk of erosion decreases quickly after the first year, at a rate of almost 90 percent each year. For example, the year following a fire may experience 0.4 to 0.9 tons/acre (1 to 2 Mg/ha) erosion, the second year less than 0.04 to 0.10 tons/acre (0.1 to 0.3 Mg/ha), and the third year may be negligible (Robichaud and Brown 1999). The erosion rate depends on the climate, topography, soil properties (including hydrophobicity), and amount of surface cover. Surface cover may include unburned duff, rock, and needle cast following fire.

Three models are commonly used after soil erosion. In USDA Forest Service Regions 1 and 4, the WATSED and similar models have frequently been used (USDA Forest Service 1990b). The Universal Soil Loss Equation (USLE) has been used widely for many years, and more recently, the Revised USLE, or RUSLE, has become common (Renard and others 1997). The Water Erosion Prediction Project (WEPP) model has recently been parameterized for predicting erosion after fire, and an interface has been developed to aid in that prediction. Improvements in the usability of both the RUSLE and WEPP prediction technologies are ongoing. The WATSED model is intended to be a cumulative affects model, to be applied at watershed scale. RUSLE and WEPP are hillslope models. WEPP has a watershed version under development, but it has received little use outside of research evaluation.

WATSED is intended as a watershed model to combine the cumulative effects of forest operations, fires, and roads on runoff and sediment yield for a given watershed. Factors that account for burned area within the watershed, soil properties, topography, and delivery ratios are identified, and an average sediment delivery is calculated. This sediment delivery is reduced over a 15-year period following a fire before the impact is assumed to be zero. Within the Western geographical territory of the Forest Service Regions, some of the factors have been adjusted to calibrate the model for local conditions, leading to the development of models such as NEZSED and BOISED. The erosion predictions are based on observations in the mountains in Regions 1 and 4, and are not intended for use elsewhere. Table 9.1 provides the erosion rates predicted in WATSED, corrected for a USLE *LS* factor of 11.2 (Wischmeier and Smith 1978). These rates are adjusted for topography, landscape, and soil properties before arriving at a final prediction. A variation of the technology in WATSED has been adopted by the State of Washington for its Watershed Analysis procedure (Washington Forest Practices Board 1997).

The Revised USLE was developed not only for agriculture, but also included rangeland conditions. The RUSLE base equation is:

$$A = R\ K\ LS\ C\ P \qquad (1)$$

Where A is the average annual erosion rate, R is the rainfall erosivity factor, K is the soil erodibility factor, LS is the slope length and steepness factor, C is the cover management factor, and P is the conservation management factor. Although it has not been widely tested, the RUSLE values appear to give reasonable erosion values for rangelands (Renard and Simanton 1990, Elliot and others 2000) and will likely do the same for forests. There are no forest climates available in the RUSLE database. Table 9.1 provides some assumptions about rate of vegetation regeneration and typical erosion rates estimated for burned and recovering forest conditions based on those assumed cover values. The RUSLE *LS* factor was about 6.54, almost half of the USLE C-factor used for WATSED. The RUSLE *LS* factor is based on more recent research and the analysis of a greater number of plots (McCool and others 1987, 1989), so it should probably be used with the WATSED technology to adjust for slope length and steepness. The RUSLE R factor was estimated as 20 from the documentation (Renard and others 1997). This is a relatively low value because much of the precipitation in the Northern Rockies comes as snowfall, and snowmelt events cause much less erosion than rainfall events.

The most recent erosion prediction technology is the Water Erosion Prediction Project (WEPP) model (Flanagan and Livingston 1995). WEPP is a complex process-based computer model that predicts soil erosion by modeling the processes that cause erosion. These processes include daily plant growth, residue accumulation and decomposition, and daily soil water balance. Each day that has a precipitation or snow melt event, WEPP calculates the infiltration, runoff, and sediment detachment, transport, deposition, and yield.

WEPP was released for general use in 1995, with an MS DOS text-based interface. Currently a Windows interface is under development and is available for general use (USDA 2000). Elliot and Hall (1997) developed a set of input templates to describe forest conditions for the WEPP model, for the MS DOS interface. The WEPP model allows the user to describe the site conditions with hundreds of variables, making the model extremely flexible, but also making it difficult for the casual user to apply to a given set of conditions. To make the WEPP model run more easily for forest conditions, Elliot and others (2000) developed a suite of interfaces to run WEPP over the Internet using Web browsers. The forest version of WEPP can be found at:

http://forest.moscowfsl.wsu.edu/fswepp

One of the interfaces is Disturbed WEPP, which allows the user to select from a set of vegetation conditions that describe the fire severity and recovering conditions. The Disturbed WEPP alters both the soil and the vegetation properties when a given vegetation treatment is selected. Table 9.2 shows the vegetation treatment selected for each of the years of recovery. In all cases, the cover input was calibrated to ensure that WEPP generated the desired cover given

Table 9.1—Erosion rates observed and predicted by WATSED, RUSLE, and WEPP for the cover shown, for a 30 percent steepness, 60-m long slope.

Year after fire	Estimated cover	Observed erosion rate	Predicted erosion rate WATSED	RUSLE	WEPP
	Percent	*Mg/ha*	*- - - - - - - - - - Mg/ha - - - - - - - - - -*		
1	50	2.2	1.92	3.35	1.74
2	65	0.02	1.64	1.30	0.37
3	80	0.01	0.96	0.54	0.02
4	95	0.00	0.48	0.20	0.00
5	97	0.00	0.29	0.16	0.00
6	99	0.00	0.15	0.09	0.00
7	100	0.00	0.06	0.07	0.00

[1] From Robichaud and Brown (1999)

Table 9.2—Vegetation treatment selected for each year of recovery with the Disturbed WEPP interface.

Years since fire	Disturbed WEPP vegetation treatment
0	High severity fire
1	Low severity fire
3	Short grass
4	Tall grass
5	Shrubs
6	5-year-old forest (99 percent cover)
7	5-year-old forest (100 percent cover)

in table 9.1. The Disturbed WEPP interface has access to a database of more than 2,600 weather stations to allow the user to select the nearest station to the disturbed site. The values in table 9.3 were predicted for Warren, ID, climate. Warren climate is similar to the climate for Robichaud and Brown's study, and also near the site where the WATSED base erosion rates were developed in central Idaho.

An important aspect of soil erosion following a fire is that the degree of erosion depends on the weather the year immediately following the fire. Table 9.1 shows the rapid recovery of a forest in the years after fire. If the year after the fire has a number of erosive storms, then the erosion rate will be high. If the year after the fire is relatively dry, then the erosion rate will be low. The values presented in table 9.1 are all average values. There is a 50 percent chance that the erosion in this most susceptible year will be less than the average value. To allow managers to better evaluate the risk of a given level of erosion following a fire, the Disturbed WEPP interface includes some probability analyses with the output, giving the user an indication of the probability associated with a given level of erosion. Table 9.3 shows that there is a one in 50, or 2 percent, chance that the erosion rate from the specified hill will exceed 3.18 tons/acre (7.12 Mg/ha), and the sediment delivery will exceed 2.88 tons/acre (6.45 Mg/ha). There is a one in 10, or 10 percent, chance that the erosion and sediment delivery rate will exceed 2.11 tons/acre (4.72 Mg/ha), and so forth. This feature will allow users to evaluate risks of upland erosion and sediment delivery to better determine the degree of mitigation that may be justified following a given fire. In California, for example, erosion is often estimated for a 5-year condition, which in this case is 1.47 tons/acre (3.3 Mg/ha). Disturbed WEPP also predicted that there was an 80 percent chance that there would be erosion on this hillslope the year following the fire.

The variability of erosion following a fire due to the climate makes any measurements difficult to evaluate. Note in table 9.1 the large drop from year 1 to year 2 in erosion rate. This decline was likely due not only to regeneration but also to the lower precipitation in 1996. In the nearby Warren, ID, climate, the average precipitation is 696 mm; the year following the fire it was 722 mm, and the second year after the fire only 537 mm. These variations from the mean also help explain why the Disturbed WEPP predicted erosion rates in table 9.1 for "average" conditions were below the observed value the first year but above the observed value the second year.

The variability in erosion observations and predictions is influenced not only by climate but also by spatial variability of soil and topographic properties. In soil erosion research to determine soil properties, it is not uncommon to have a standard deviation in observations from identical plots greater than the mean. A rule of thumb in interpreting erosion observations or predictions is that the true "average" value is likely to be within plus or minus 50 percent of the observed value. In other words, if a value of 0.9 tons/acre (2 Mg/ha) is observed in the field from a single observation, the true "average" erosion from that hillside is likely to be between 0.4 and 1.3 tons/acre (1 and 3 Mg/ha). Following this rule leads to the conclusions that WATSED, RUSLE, and WEPP predictions in table 9.1 are not different from the observed erosion rates.

Table 9.3—Exceedance probabilities associated with different levels of precipitation, runoff, and soil erosion for the year following a severe wild fire in central Idaho.

Return period	Precipitation		Runoff		Erosion		Sediment	
Years	*mm*	*in*	*mm*	*in*	*Mg/ha*	*tons/ac*	*Mg/ha*	*tons/ac*
50.0	973.60	38.33	31.82	1.25	7.12	3.18	6.45	2.88
25.0	892.80	35.15	31.79	1.25	6.45	2.88	6.32	2.82
10.0	811.50	31.95	27.65	1.09	4.72	2.11	4.72	2.11
5.0	756.10	29.77	20.56	0.81	3.30	1.47	3.30	1.47
2.5	671.80	26.45	14.74	0.58	1.80	0.80	1.80	0.80
Average	670.92	26.41	12.47	0.49	1.74	0.78	1.74	0.78

In the years of regeneration, it appears that both WATSED and RUSLE are overpredicting observed erosion rates, whereas the Disturbed WEPP predictions are nearer to the observed values. WATSED, as a cumulative effects model, is considering the impact of the disturbance on a watershed scale. Frequently eroded sediments following a disturbance may take several years to be routed through the watershed, whereas WEPP is only considering the hillslope in its predictions. RUSLE is also a hillslope model but considers only the upland eroding part of the hillside and does not consider any downslope deposition. This means that RUSLE values will frequently be overpredicted unless methods to estimate delivery ratio are considered. A RUSLE2 model currently under development addresses downslope deposition and sediment delivery.

Model Selection

Managers must determine which model most suits the problem at hand. The WATSED technology is geographic specific, as is the Washington Forest Practices model. These models should not be used outside of the areas for which they were developed. The WATSED technology is intended to assist in watershed analysis and not necessarily intended for estimating soil erosion after fires. RUSLE is intended to predict upland erosion and is best suited for estimating potential impacts of erosion on onsite productivity. It is less well suited for predicting offsite sediment delivery. The WEPP technology provides estimates of both upland erosion for soil productivity considerations and sediment delivery for offsite water quality concerns. The WEPP DOS and Windows technology requires skill to apply and should be considered only by trained specialists. The Disturbed WEPP interface requires little training, and documentation with examples is included on the Web site, making it available to a wider range of users.

DELTA-Q and FOREST Models

Two other models warrant brief mentioning. They can assist fire managers in dealing with watershed scale changes in water flow and erosion. These are DELTA-Q and FOREST. Both programs require an ESRI Arc 8.x license. Further documentation can be found at:

http://www.cnr.colostate.edu/frws/people/faculty macdonald/model.htm.

One of the difficult tasks facing land managers, fire managers, and hydrologists is quantifying the changes in streamflow after forest disturbances such as fire. The changes of interest are alteration of peak, median, and low flows as well as the degradation of water quality due to increased sediment delivery to channels or channel degradation.

DELTA-Q is a model designed to calculate the cumulative changes in streamflow on a watershed scale from areas subjected to the combination of harvesting and road construction. Flow changes due to forest cover removal by wildfire can also be calculated. A current data limitation in the model is that it evaluates only changes due to vegetation combustion, not the possible effects of alterations to runoff and streamflow generation processes. The objective of DELTA-Q is to provide fire and watershed managers with a GIS-based tool that can quickly approximate the sizes of changes in different flow percentiles. The model does not estimate the increases in streamflow from extreme events (see chapters 2 and 5). The model was designed to be used for planning at watershed scales of 5 to 50 mi^2 (3,200 to 32,000 acres, or 1,300 to 13,000 ha).

The FOREST (FORest Erosion Simulation Tools) model functions with DELTA-Q. It calculates changes in the sediment regime due to forest disturbances. It consists of a hillslope model that uses a polygon GIS layer of land disturbances to calculate sediment production. Road-related sediment is treated separately because roads are linear features in the landscape. Input values for the road segment can be generated by several means including WEPP.Road. FOREST does not deal with changes in channel stability.

Models Summary

This chapter is not meant as a comprehensive look at simulation models. Several older modeling technologies commonly used estimate fire effects during and after fire (FOFEM, WATSED, WEPP, RUSLE, and others). New ones such as DELTA-Q, FOREST have been recently developed, and others are under construction. These process-based models provide managers with additional tools to estimate the magnitude of fire effects on soil and water produced by land disturbance. FOFEM was developed to meet needs of resource managers, planners, and analysts in predicting and planning for fire effects. Quantitative predictions of fire effects are needed for planning prescribed fires that best accomplish resource needs, for impact assessment, and for long-range planning and policy development. FOFEM was developed to meet this information need. The WATSED technology was developed for watershed analysis. The RUSLE model was developed for agriculture and rangeland hillslopes and has been extended to forest lands. The WEPP model was designed as an improvement over RUSLE that can either be run as a stand-alone computer model by specialists, or accessed through a special Internet interface designed for forest applications, including wild fires.

All of these models have limitations that must be understood by fire managers or watershed specialists before they are applied. The models are only as good as the data used to create and validate them. Some processes such as extreme flow and erosion events are not simulated very well because of the lack of good data or the complexity of the processes. However, they do provide useful tools to estimate landscape changes to disturbances such as fire. Potential users should make use of the extensive documentation of these models and consult with the developers to ensure the most appropriate application of the models.

Notes

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Chapter 10: Watershed Rehabilitation

Recent large, high severity fires in the United States, coupled with subsequent major hydrological events, have generated renewed interest in the linkage between fire and onsite and downstream effects (fig. 10.1). Fire is a natural and important disturbance mechanism in many ecosystems. However, the intentional human suppression of fires in the Western United States, beginning in the early 1900s, altered natural fire regimes in many areas (Agee 1993). Fire suppression can allow fuel loading and forest floor material to increase, resulting in fires of greater intensity and extent than might have occurred otherwise (Norris 1990).

Figure 10.1—Flood flow on the Apache-Sitgreaves National Forest, Arizona, after the Rodeo-Chediski Fire of 2002. (Photo by Dave Maurer).

High severity fires are of particular concern because the potential affects on soil productivity, watershed response, and downstream sedimentation often pose threats to human life and property. During severe fire seasons, the USDA Forest Service and other Federal and State land management agencies spend millions of dollars on postfire emergency watershed rehabilitation measures intended to minimize flood runoff, onsite erosion, and offsite sedimentation and hydrologic damage. Increased erosion and flooding are certainly the most visible and dramatic impacts of fire apart from the consumption of vegetation.

Burned Area Emergency Rehabilitation (BAER)

The first formal reports on emergency watershed rehabilitation after wildfires were prepared in the 1960s and early 1970s, although postfire seeding with grasses and other herbaceous species was conducted in many areas in the 1930s, 1940s, and 1950s (Christ 1934, Gleason 1947). Contour furrowing and trenching were used when flood control was a major concern (Noble 1965, DeByle 1970b). The Forest Service and other agencies had no formal emergency rehabilitation program. Funds for fire suppression disturbance were covered by fire suppression authorization. Watershed rehabilitation funding was obtained from emergency flood control programs or, more commonly, restoration accounts. Prior to 1974, the fiscal year had ended June 30 of each year, allowing year-end project funds to be shifted to early season fires. After July 1, fires were covered by shifts in the new fiscal year funding. The shift to an October 1 to September 30 fiscal year made it difficult to provide timely postfire emergency treatments or create appropriated watershed restoration accounts.

In response to a Congressional inquiry on fiscal accountability, in 1974 a formal authority for $2 million in postfire rehabilitation activities was provided in the Interior and Related Agencies appropriation. Called Burned Area Emergency Rehabilitation (BAER), this authorization was similar to the fire fighting funds in that it allowed the Forest Service to use any available funds to cover the costs of watershed treatments when an emergency need was determined and authorized. Typically, Congress reimbursed accounts used in subsequent annual appropriations. Later, annual appropriations provided similar authorities for the Bureau of Land Management and then other Interior agencies. The occurrence of many large fires in California and southern Oregon in 1987 caused expenditures for BAER treatments to exceed the annual BAER authorization of $2 million. Congressional committees were consulted and the funding cap was removed. The BAER program evolved, and policies were refined based on determining what constituted a legitimate emergency warranting rehabilitation treatments.

The BAER-related policies were initially incorporated into the Forest Service Manual (FSM 2523) and the Burned Area Emergency Rehabilitation (BAER) Handbook (FSH 2509.13) in 1976. These policies required an immediate assessment of site conditions following wildfire and, where necessary, implementation of emergency rehabilitation measures. These directives delineated the objectives of the BAER program as:

1. Minimizing the threat to life and property onsite and offsite.
2. Reducing the loss of soil and onsite productivity.
3. Reducing the loss of control of water.
4. Reducing deterioration of water quality.

As postfire rehabilitation treatment increased, debates arose over the effectiveness of grass seeding and its negative impacts on natural regeneration. Seeding was the most widely used individual treatment, and it was often applied in conjunction with other hillslope treatments, such as contour-felled logs and channel treatments.

In the mid 1990s, a major effort was undertaken to revise and update the BAER handbook. A steering committee, consisting of regional BAER coordinators and other specialists, organized and developed the handbook used today. The issue of using native species for emergency revegetation emerged as a major topic, and the increased use of contour-felled logs (fig. 10.2) and mulches caused rehabilitation expenditures to escalate. During the busy 1996 fire season, for example, the Forest Service spent $11 million on BAER projects. In 2000, 2001, and 2002 the average annual BAER spending rose to more than $50 million.

Improvements in the BAER program in the late 1990s included increased BAER training and funding review. Increased training needs were identified for BAER team leaders, project implementation, and on-the-ground treatment installation. Courses were developed for the first two training needs but not the last. Current funding requests are scrutinized by regional and national BAER coordinators to verify that funded projects are minimal, necessary, reasonable, practicable, cost effective, and a significant improvement over natural recovery.

In the late 1990s, a program was initiated to integrate national BAER policies across different Federal agencies (Forest Service, Bureau of Land Management, National Park Service, Fish and Wildlife Service, and Bureau of Indian Affairs) as each agency had different authorities provided in the Annual Appropriations Acts.

Figure 10.2—Installing contour-felled logs for erosion control after a wildfire. (Photo by Peter Robichaud).

The U.S. Department of Agriculture and Department of the Interior approved a joint policy for a consistent approach to postfire rehabilitation in 1998. The new policy broadened the scope and application of BAER analysis and treatment. Major changes included:

1. Monitoring to determine if additional treatment is needed and evaluating to improve treatment effectiveness.
2. Repairing facilities for safety reasons.
3. Stabilizing biotic communities.
4. Preventing unacceptable degradation of critical known cultural sites and natural resources.

BAER Program Analysis

Early BAER efforts were principally aimed at controlling runoff and consequently erosion. Research by Bailey and Copeland (1961), Christ (1934), Copeland (1961, 1968), Ferrell (1959), Heede (1960, 1970), and Noble (1965) demonstrated that various watershed management techniques could be used on forest, woodland, shrub, and grassland watersheds to control both storm runoff and erosion (fig. 10.3). Many of these techniques were developed from other disciplines (such as agriculture and construction) and refined or augmented to form the set of BAER treatments in use today (table 10.1).

In spite of the improvements in the BAER process and the wealth of practical experience obtained over the past several decades, the effectiveness of many emergency rehabilitation methods have not been systematically tested or validated. Measuring erosion and runoff is expensive, complex, and labor intensive (fig. 10.4). Few researchers or management specialists have the resources or the energy to do it. BAER team leaders and decisionmakers often do not have information available to evaluate the short- and long-term benefits (and costs) of various treatment options.

Figure 10.3—Straw bale check dams placed in channel by the Denver Water Board after the Hayman Fire, 2002, near Deckers, CO. (Photo by Peter Robichaud).

In 1998, a joint study by the USDA Forest Service Rocky Mountain Research Station and the Pacific Southwest Research Station evaluated the use and effectiveness of postfire emergency rehabilitation methods (Robichaud and others 2000).

The objectives of the study were to:

1. Evaluate the effectiveness of rehabilitation treatments at reducing postwildfire erosion, runoff, or other effects.
2. Assess the effectiveness of rehabilitation treatments in mitigating the downstream effects of increased sedimentation and peakflows.
3. Investigate the impacts of rehabilitation treatments on natural processes of ecosystem recovery, both in the short and long term.
4. Compare hillslope and channel treatments to one another and to a no-treatment option.
5. Collect available information on economic, social, and environmental costs and benefits of various rehabilitation treatment options, including no treatment.
6. Determine how knowledge of treatments gained in one location can be transferred to another location.
7. Identify information gaps needing further research and evaluation.

Robichaud and others (2000) collected and analyzed information on past use of BAER treatments in order to determine attributes and conditions that led to treatment success or failure in achieving BAER goals. Robichaud and others (2000) restricted this study to USDA Forest Service BAER projects in the continental Western United States and began by requesting Burned Area Report (FS-2500-8) forms and monitoring reports from the Regional headquarters and Forest Supervisors' offices. The initial efforts revealed that information collected on the Burned Area Report forms and in the relatively few existing postfire monitoring reports was not sufficient to assess treatment effectiveness, nor did the information capture the knowledge of BAER specialists. Therefore, interview questions were designed to enable ranking of expert opinions on treatment effectiveness, to determine aspects of the treatments that lead to success or failure, and to allow for comments on various BAER-related topics.

Interview forms were developed after consultation with several BAER specialists. The forms were used to record information when BAER team members and regional and national leaders were interviewed. Onsite interviews were conducted because much of the supporting data were located in the Forest Supervisors' and District Rangers' offices and could be

Table 10.1—Burned Area Emergency Rehabilitation (BAER) treatments (From Robichaud and others 2000).

Hillslope	Channel	Road and trail
Broadcast seeding	Straw bale check dams	Rolling dips
Seeding plus fertilizer	Log grade stabilizers	Water bars
Mulching	Rock grade stabilizers	Cross drains
Contour-felled logs	Channel debris clearing	Culvert overflows
Contour trenching	Bank/channel armoring	Culvert upgrades
Scarification and ripping	In-channel tree felling	Culvert armoring
Temporary fencing	Log dams	Culvert removal
Erosion fabric	Debris basins	Trash racks
Straw wattles	Straw wattle dams	Storm patrols
Slash scattering	Rock gabion dams	Ditch improvements
Silt fences		Armored fords
Geotextiles		Outsloping
Sand or soil bags		Signing

retrieved during the interviews. Because much of the information was qualitative, attempts were made to ask questions that would allow for grouping and ranking results.

BAER program specialists were asked to identify treatments used on specific fires and what environmental factors affected success and failure. For each treatment, specific questions were asked regarding the factors that caused the treatment to succeed or fail, such as slope classes, soil type, and storm events (rainfall intensity and duration) affecting the treated areas. They were also asked questions regarding implementation of treatments and whether any effectiveness monitoring was completed. For cases where monitoring was conducted (either formal or informal), interviewees were asked to describe the type and quality of the data collected (if applicable) and to give an overall effectiveness rating of "excellent," "good," "fair," or "poor " for each treatment.

Figure 10.4—During a short duration high intensity rain event, this research sediment trap was filled. Using pre- and postsurveys, Hydrologist Bob Brown and Engineer Joe Wagenbrenner measure the sediment collected with the help of a skid-steer loader. Research plots are in high severity burned areas of the 2002 Hayman Fire, Pike-San Isabel National Forest near Deckers, CO. (Photo by J. Yost).

This evaluation covered 470 fires and 321 BAER projects, from 1973 through 1998 in USDA Forest Service Regions 1 through 6. A literature review, interviews with key Regional and Forest BAER specialists, analysis of burned area reports, and review of Forest and District monitoring reports were used in the evaluation. The resulting report, Evaluating the Effectiveness of Postfire Rehabilitation Treatments (Robichaud and others 2000), includes these major sections:

1. Information acquisition and analysis methods.
2. Description of results, which include hydrologic, erosion and risk assessments, monitoring reports, and treatment evaluations.
3. Discussion of BAER assessments and treatment effectiveness.
4. Conclusions drawn from the analysis.
5. Recommendations.

This chapter provides a synopsis of the findings in that report, as well as new information that has been determined since the report was published.

Postfire Rehabilitation Treatment Decisions

The BAER Team and BAER Report

As soon as possible (even before a fire is fully contained), a team of specialists is brought together to evaluate the potential effects of the fire and to recommend what postfire rehabilitation, if any, should be used in and around the burned area. Hydrology and soil science are the predominant disciplines represented on nearly all BAER teams. Depending on the location, severity, and size of the fire, wildlife biologists, timber, range, and fire managers, engineers, archeologists, fishery biologists, and contracted specialists may be included on the team.

The Burned Area Report filed by the BAER team describes the hydrologic and soil conditions in the fire area as well as the predicted increase in runoff, erosion, and sedimentation. The basic information includes the watershed location, size, suppression cost, vegetation, soils, geology, and lengths of stream channels, roads, and trails affected by the fire. The watershed descriptions include areas in low, moderate, and high severity burn categories as well as areas with water repellent soils. The runoff, erosion, and sedimentation predictions are then evaluated in combination with both the onsite and downstream values at risk to determine the selection and placement of emergency rehabilitation treatments. The BAER team uses data from previous fires, climate modeling, erosion prediction tools, and professional judgment to make the BAER recommendations.

Erosion Estimates from BAER Reports

Robichaud and others (2000) found a wide range of potential erosion and watershed sediment yield estimates in the Burned Area Report forms. Some of the high values could be considered unrealistic (fig. 10.5). Erosion potential varied from 1 to 6,913 tons/acre (2 to 15,500 Mg/ha), and sediment yield varied over six orders of magnitude. Erosion potential and sediment yield potential did not correlate well ($r = 0.18$, $n = 117$). Different methods were used to calculate these estimates on different fires, making comparisons difficult. Methods included empirical base models such as Universal Soil Loss Equation (USLE), values based on past estimates of known erosion events, and professional judgment. In recent years, considerable effort has been made to improve erosion prediction after wildfire through the development and refinement of new models (Elliot and others 1999, 2000). These models are built on the Water Erosion Prediction Project (WEPP) technology (Flanagan and others 1994), which has been adapted for application after wildfire. The model adaptation includes the addition

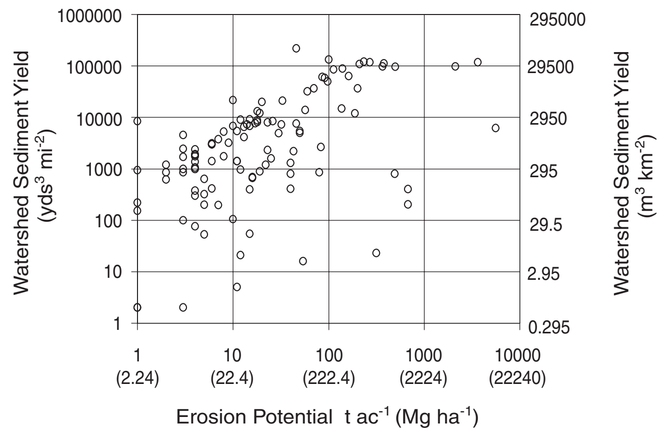

Figure 10.5—Estimated hillslope erosion potential and watershed sediment yield potential (log scale) for all fires requesting BAER funding. (From Robichaud and others 2000).

of standard windows interfaces to simplify use and Web-based dissemination for general accessibility at:

http://forest.moscowfsl.wsu.edu/fswepp/

and

http://fsweb.moscow.rmrs.fs.fed.us/fswepp.

Hydrologic Response Estimates

Evaluating the potential effects of wildfire on hydrologic responses is an important first step in the BAER process. This involves determining storm magnitude, duration, and return interval for which treatments are to be designed. Robichaud and others (2000) found that the most common design storms were 10-year return events (fig. 10.6, 10.7). Storm durations were

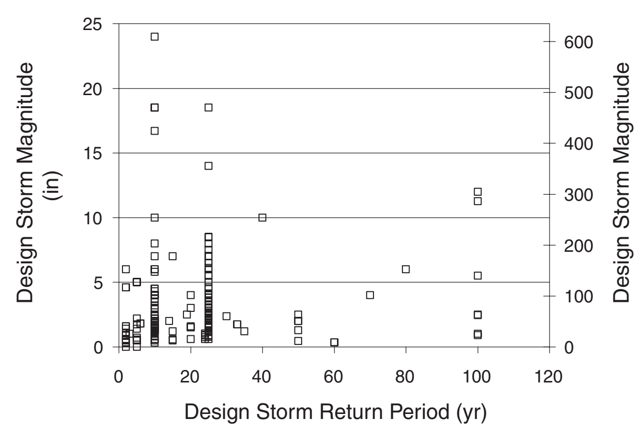

Figure 10.6—Design storm duration by return period for all fires requesting BAER funding. (From Robichaud and others 2000).

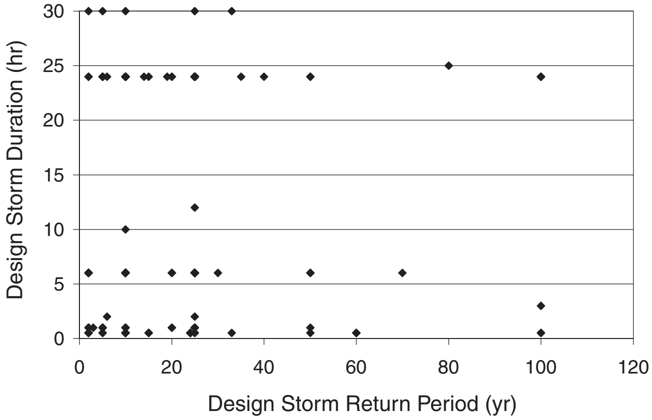

Figure 10.7—Design storm magnitude and return period for all fires in the Western United States requesting BAER funding. (From Robichaud and others 2000).

usually less than 24 hours, with the common design storm magnitudes from 1 to 6 inches (25 to 150 mm). Five design storms were greater than 12 inches (305 mm), with design return intervals of 25 years or less. The variation in estimates reflects some climatic differences throughout the Western United States.

The Burned Area Report also contains an estimate of the percentage of burned watersheds that have water repellent soil conditions. Soils in this condition are often reported after wildfires and are expected to occur more commonly on coarse-grained soils, such as those derived from granite (fig. 10.8). However, no statistical difference was found in the geologic parent material and the percent of burned area that was water repellent. Robichaud and Hungerford (2000) also found no differences in the water repellant conditions with various soil types. BAER teams estimate a percentage reduction in infiltration capacity as part of the Burned Area Report. Comparison of reduction in infiltration rate to percentage of area that was water repellent showed no statistically significant relationship (fig. 10.9). However, Robichaud (2000) and Pierson and others (2001a) showed a 10 to 35 percent reduction in infiltration after the first year. Factors other than water repellent soil conditions, such as loss of the protective forest floor layers, obviously affect infiltration capacity.

Estimation methods for expected changes in channel flow due to wildfire were variable but primarily based on predicted change in infiltration rates. Thus, a 20 percent reduction in infiltration resulted in an estimated 20 percent increase in channel flows. Various methods were used to determine channel flow including empirical-based models, past U.S. Geological Survey records from nearby watersheds that had a flood response, and professional judgment. Some reports show a large percent increase in design flows (fig. 10.10).

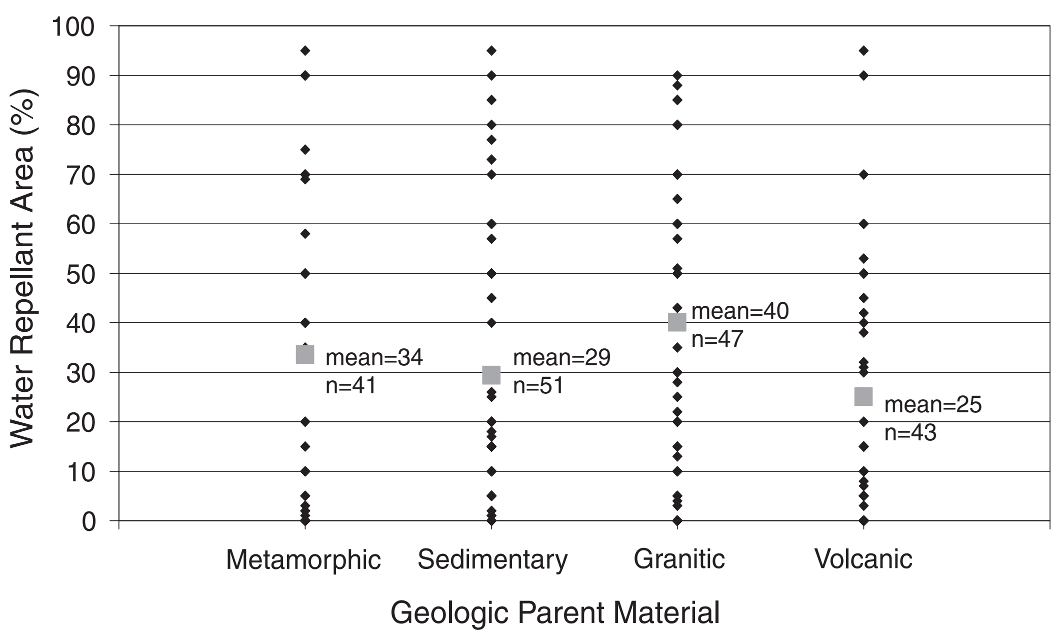

Figure 10.8—Fire-induced water repellent soil areas and their geologic parent material for all fires requesting BAER funding. Fire-induced water repellency was not significantly different by parent material (t-test, alpha = 0.05). (From Robichaud and others 2000).

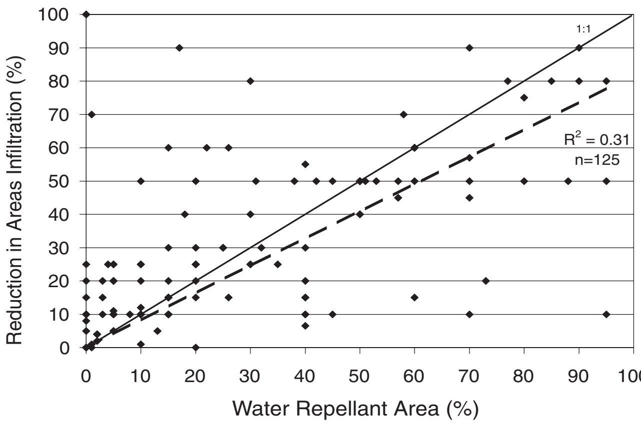

Figure 10.9—Fire-induced water repellent soil areas compared to the estimated reduction in infiltration for all fires requesting BAER funding. (From Robichaud and others 2000).

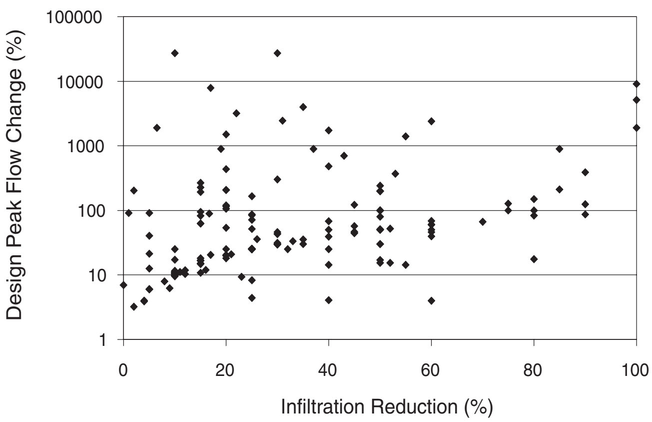

Figure 10.10—Estimated design peakflow change (log scale) due to wildfire burned areas relative to the estimated reduction in infiltration for all fires requesting BAER funding. (From Robichaud and others 2000).

Hillslope Treatments and Results

Hillslope Treatments

Hillslope treatments are intended to reduce surface runoff and keep postwildfire soil in place on the hillslope and thereby prevent sediment deposition in unwanted areas. These treatments are regarded as a first line of defense against postfire erosion and sediment movement. Hillslope treatments comprise the greatest portion of time, effort, and expense in most BAER projects. Consequently, more information is available on hillslope treatments than on channel or road treatments.

Broadcast Seeding— The most common BAER practice is broadcast seeding. Grass seeding after fire for range improvement has been practiced for decades, with the intent to gain useful products from land that will not return to timber production for many years (Christ 1934, McClure 1956). As an emergency treatment, rapid vegetation establishment has been regarded as the most cost-effective method to promote rapid infiltration of water and keep soil on hillslopes (Noble 1965, Rice and others 1965, Miles and others 1989).

Grasses are particularly desirable for this purpose because their extensive, fibrous root systems increase water infiltration and hold soil in place. Fast-growing nonnative species have typically been used. They are inexpensive and readily available in large quantities when an emergency arises (Barro and Conard 1987, Miles and others 1989, Agee 1993). Legumes are often added to seeding mixes for their ability to increase available nitrogen in the soil after the postfire nutrient flush has been exhausted, aiding the growth of seeded grasses and native vegetation (Ratliff and McDonald 1987). Seed mixes were refined for particular areas as germination and establishment success were evaluated. Most mixes contained annual grasses to provide quick cover and perennials to establish longer term protection (Klock and others 1975, Ratliff and McDonald, 1987). However, nonnative species that persist can delay recovery of native flora and alter local plant diversity. Native grass seed can be expensive and hard to acquire in large quantities or in a timely manner compared to cereal grains or pasture grasses. When native seed is used, it should come from a nearby source area to preserve local genetic integrity. When native seed is not available, BAER specialists have recommended using nonreproducing annuals, such as cereal grains or sterile hybrids that provide quick cover and then die out to let native vegetation reoccupy the site.

Application of seed can be done from the air or on the ground. In steep areas and in areas where access is limited, aerial seeding is often the only option. Effective application of seed by fixed-wing aircraft or helicopter requires global positioning system (GPS) navigation, significant pilot skill, and low winds for even cover. Ground seeding, applied from all-terrain vehicles or by hand, assures more even seed application than aerial seeding. Seeding is often combined with other treatments, such as mulching and scarifying, as these additional treatments help anchor the seeds and improve seed germination.

Effectiveness of seeding depends on timeliness of seed application, choice of seed, protection from grazing, and luck in having gentle rains to stimulate seed germination before wind or heavy rains blow or wash soil and seed away. Proper timing of seed application depends on location. In some areas, it is best to seed directly into dry ash, before any rain falls, to take advantage of the fluffy seedbed condition, while in other areas, seed is best applied after the first snow so that it will germinate in the spring. Both conditions also reduce loss to rodents. The potential advantage of seeded grass to inhibit the growth and spread of noxious weeds also depends on timely application and germination.

Mulch—Mulch is any organic material spread over the soil surface that functions like the organic forest floor that is often destroyed in high and moderate severity burn areas. Both wet mulch (hydromulch) and dry mulch (wheat straw, jute excelsior, rice straw, and so forth) are available; however, mulches have only recently been used as a postfire rehabilitation treatment. Mulch is applied alone or in combination to reduce raindrop impact and overland flow and, thereby, to enhance infiltration and reduce soil erosion. It is often used in conjunction with grass seeding to provide ground cover in critical areas. It also intercepts precipitation for subsequent infiltration. Mulch protects the soil and improves moisture retention underneath it, benefiting seeded plants in hot areas but not always in cool ones. Use of straw from pasture may introduce exotic grass seed or weeds, so BAER projects are now likely to seek "weed-free" mulch such as rice straw.

Mulches can be applied from the air or from the ground. Aerial dry mulching uses helicopters with attached cargo net slings carrying the straw mulch, which is released over the treatment area (San Dimas Technology Development Center 2003). Hydromulch can be applied from the air using helicopters fitted with hydromulch slurry tanks or buckets, which are released in controlled drops over the treatment areas. Both of these aerial applications are expensive. Ground application of dry mulch is done by hand using all terrain vehicles to carry the straw from a staging area into the treatment area. Ground application of hydromulch is done from spray trucks and is limited to an area 200 feet (61 m) of either side of a road. Given its expense, mulch is usually used in high value areas, such as above or below roads, above streams, or below ridge tops.

Mulching is most effective on gentle slopes and in areas where high winds are not likely to occur. Wind either blows the mulch off site or piles it so deeply that seed germination is inhibited. On steep slopes, rain can wash some of the mulch material downslope. Use of a tackifier or felling small trees across the mulch may increase onsite retention. Hydromulches often have tackifiers that help bind the mulch in the soil. Both hydromulch and dry mulch were used to stabilize soils on the Cerro Grande Fire of 2000 and Rodeo-Chediski and Hayman Fires of 2002. However, use of these treatments escalated the BAER treatment costs to $10 to $20 million per fire.

Contour Log Structures (Contour Log Basins, Log Erosion Barriers, Log Terraces, Terracettes)— This treatment involves felling logs on burned-over hillsides and laying them on the ground along the slope contour to provide mechanical barriers to water flow, promote infiltration, and reduce sediment movement. Contour-felled logs reduce water velocity, break up concentrated flows, induce hydraulic roughness to burned watersheds, and store sediment. The potential volume of sediment stored is highly dependent on slope, the layout design, the size and length of the felled trees, and the degree to which the felled trees are adequately staked and placed into ground contact. In some instances contour-felled log barriers have filled with sediment following the first several storm events after installation, while others have taken 1 to 2 years to fill (Robichaud 2000).

This treatment was originally designed to provide the same function as contour trenches and furrows. The primary function of the Contour Log Basins or Contour Log Terraces was to detain and infiltrate runoff from a design storm. To accomplish this, logs ranging generally from 6 to 12 inches (15 to 30 cm) in diameter were felled on the contour and staked in place. The treatment was begun at the top of the slope because each course of contour logs depends on the design spacing and capacity of the upslope courses to be effective. The spacing depends on the capacity of the structure to contain runoff according to the formula:

$$S = RO/12 \times C$$

Where: S = spacing of log courses down slope measured horizontally in feet.

RO = Storm runoff in inches.
C = Basin capacity in cubic feet/lineal foot of log.

Basins were created behind each log by scraping soil against the log to seal it. Earthen end sills and baffles complete the structure. To contain 1.0 inch (25 mm) of runoff typically requires spacing of less than 20 feet (9.6 m) between courses. Contour placement is vital, and eliminating long, uninterrupted flow paths by "brick coursing" provides additional effectiveness. The treatments detain storm runoff on site, thereby eliminating transport of eroded soils. If the design capacity is exceeded, the structure provides some secondary benefit by reducing slope length, which interrupts concentrated flows and sediment movement. Because of their small size, the effective life of properly installed treatments is only a few years at most. Undesigned and underdesigned treatments with wide spacing and lacking runoff storage capacity can effectively concentrate runoff and cause damage that might conceivably be greater than no treatment. In high rainfall areas of the West Coast, contour log basins may be infeasible. In these cases, contour logs are placed in the same manner as above, but the exception is that they will provide only secondary benefits. It should be kept in mind that these structures are intended to detain runoff. If they immediately fill with sediment, they were likely underdesigned.

Shallow, rocky soils that are uneven are problematic for anchoring, so care must be taken to ensure that logs are adequately secured to the slope. Overly rocky and steep slopes should be avoided because benefits gained from contour-felling treatment can be easily offset by the extra implementation time required and the limited capacity to detain runoff or provide stabilization of small amounts of soil. Gentler slopes and finer textured soils (except clayey soils) lead to better installation and greater runoff control efficiency. In highly erosive soils derived from parent material such as granitics or glacial till, so much sediment can be mobilized that it might overwhelm small contour-felled logs. Availability of adequate numbers of straight trees must be considered when choosing this treatment.

Straw Wattles—Straw wattles main purpose is to break up slope length and reduce flow velocities of concentrated flow. Straw wattles are 9 to 10 inches (23 to 25 cm) in diameter and made of nylon mesh tubes filled with straw. They are permeable barriers used to detain surface runoff long enough to reduce flow velocity and provide for sediment storage. With end sills, baffles, and on the proper design spacing, straw wattles can provide runoff detention.

Straw wattles have been used in small, first order, drainages or on side slopes for detaining small amounts of sediment. They should never be placed in main or active drainages. Straw wattles function similarly to contour-felled logs. The sediment holding capacity can be increased by turning 2 feet at each end of the wattle upslope. Straw wattles are a good alternative in burned areas where logs are absent, poorly shaped, or scarce. Straw wattles are relatively inexpensive, but they can be disturbed by grazing animals or decompose or catch fire. Although the wattle netting is photodegradable, there are concerns that it persists long enough to pose hazards for small animals.

Contour Trenching and Terraces—Full-scale contour trenches are designed to totally detain the runoff from a design storm on site. The treatment must progress from the top of the slope downward as each trench course is dependent on the next one upslope. Smaller "outside" trenches can be constructed on slopes less than 30 percent. For slopes greater than 30 percent an "inside" trench must be built. This requires building a "full bench" platform for bulldozers to operate on. In subsequent passes, the trench is tipped into the slope, forming a basin. On the final pass, bulldozers back out and push up baffles that segment the trench and allow flows to equalize into other cells. The formula for digging trenches is:

$$S = RO/12 \times C$$

Where: S = spacing of trench courses down slope measured horizontally in feet.

RO = Storm runoff in inches.

C = Basin capacity in cubic feet/lineal foot of trench.

The practical upper limit of capacity is about 3 inches (76 mm) of runoff. Contour trenches require a minimum of 4 feet (1.2 m) of soil above bedrock for adequate construction. They work best in gravelly loams and have been applied in granitic soils and clay soils with less success (Schmidt Personal Communication 2004). Granitic soils do not maintain a structural shape well because of their coarseness and difficulty to get regenerated with cover. Clay soils can become plastic with the addition of water, and in landslide-prone topography, contour trenches can activate localized mass failures. Contour trenching has proven to be effective in a number of localities in the past, but concerns about visual effects and cultural heritage values have limited their use in the past three decades.

More recently, smaller scale contour trenches have been used to break up the slope surface, to slow runoff, to allow infiltration, and to trap sediment. These trenches or terraces are often used in conjunction with other treatments such as seeding. They can be constructed with machinery (deeper trenches) or by hand (generally shallow). Width and depth vary with design storm, spacing, soil type, and slope. When installed with heavy equipment, trenches may result in considerable soil disturbance that can create immediate erosion problems. In addition, erosion problems can occur many years after installation when runoff cuts through the trench embankment. Trenches have high visual impact when used in open areas. Shallow hand trenches tend to disappear with time as they are filled with sediment and covered by vegetation. On the other hand, large trenches installed several decades ago are still visible on the landscape. Because contour trenching and terraces are ground-disturbing activities, cultural clearances are required, and these may significantly slow the installation process.

Scarification and Ripping—Scarification and ripping are mechanical soil treatments aimed at improving infiltration rates in water repellent soils. Tractors and ATVs can be used to pull shallow harrows on slopes of 20 percent or less. Hand scarification uses steel rakes (McLeods). These treatments may increase the amount of macropore space in soils by the physical breakup of dense or water repellent soils, and thus increase the amount of rainfall that infiltrates into the soil. In addition, scarification can provide a seedbed for planting that improves germination rates. Shallow soils, rock outcrops, steep slopes, incised drainages, fine-textured soils, and high tree density create significant problems for scarification and ripping. These treatments work best where there is good soil depth, the soils are coarse textured, slopes are less than 30 percent, and woody vegetation density is low.

Silt Fences—Silt fences are installed to trap sediment in swales, small ephemeral drainages, or along hillslopes where they provide temporary sediment storage. Given the labor-intensive installation, they are used as treatment only when other methods would not be effective. They work best on gentler slopes, such as swales, but can be effective on steeper, rocky slopes where log erosion barriers would not achieve good ground contact. Silt fences are also installed to monitor sediment movement as part of effectiveness monitoring and can last several years before UV breakdown of the fabric (Robichaud and Brown 2002).

Geotextiles and Geowebbing—Polymer textiles and webbings are used to cover ground and control erosion in high-risk areas, such as extremely steep slopes, above roads or structures, or along streambanks. This material is often used in conjunction with seeding. Geotextiles come in different grades with ultraviolet inhibitors that determine how long they will last in the field. Geotextiles must be anchored securely to remain effective, especially along streambanks. The complete cover provided by some geotextiles can reduce native plant establishment.

Sand, Soil, or Gravel Bags—Sand, soil, or gravel bags are used on hillslopes or in small channels or to trap sediment and interrupt water flow. Various seed mixes or willow wands may be added to the bags to help establish vegetation. The bags are often placed in staggered rows like contour-felled logs in areas where there are no trees available. Rows of bags break water flow and promote infiltration. They store sediment temporarily, then break down and release it. They are not appropriate for use in V-shaped channels.

Temporary Fencing—Temporary fencing is used to keep grazing livestock and off-highway vehicles (OHVs) out of burned areas and riparian zones during the recovery period. Resprouting onsite vegetation and seeded species attract grazing animals and require protection to be successful.

Slash Spreading—Slash spreading covers the ground with organic material, interrupting rain impact and trapping soil. It is a common practice after timber sales, but it can also be used on burned slopes where dead vegetation is present. Slash is often used to rehabilitate firebreaks and dozer firelines. It is also used in moderately burned areas where there is more material available to spread, or below an intensely burned slope or area of water repellent soils. To be effective, slash needs to be cut so it makes good contact with the ground.

Needle Cast—Needle cast commonly occurs after low and moderate severity burns in coniferous forests. The dead needles that fall to the ground provide surface cover that functions as a naturally occurring mulch. Pannkuk and Robichaud (2003) indicate that 50 percent ground cover can reduce interrill erosion by 60 to 80 percent and rill erosion 20 to 40 percent. Although needle cast is not an applied treatment, its presence may reduce or eliminate the need for other treatments.

Hillslope Treatment Effectiveness

Increasing infiltration of rainfall and preventing soil from leaving the hillslope are considered the most effective methods to slow runoff, reduce flood peaks, retain site productivity, and reduce downstream sedimentation. Many of these hillslope treatments may be appropriate in critical areas of high risk. Monitoring of treatment effectiveness is needed to determine which treatments will work in specific settings as well as their cost effectiveness (General Accounting Office 2003, Robichaud and others 2003).

BAER Expert Rating—Hillslope treatments are implemented to keep soil in place and comprise the greatest effort in most BAER projects (fig. 10.2). Mulching and geotextiles were rated the most effective hillslope treatments because they provide immediate ground cover to reduce raindrop impact and overland flow as well as to hold soil in place. Mulching was rated "excellent" in 67 percent of the evaluations, and nobody considered it a "poor" treatment. Aerial seeding, the most frequently used BAER treatment, was rated about equally across the spectrum from "excellent" to "poor." Nearly 82 percent of the evaluations placed ground seeding effectiveness in the "good" category. Evaluations of seeding plus fertilizer covered the spectrum from "excellent" to "poor," although most responses were "fair" or "poor." After seeding, contour-felled logs are the next most commonly applied hillslope treatment. The rating for contour-felled logs was "excellent" or "good" in 66 percent of the evaluations. The remainder of the hillslope treatments received only three evaluations each, so no conclusions are offered beyond the fact that they were generally rated "excellent," "good," or "fair," and none were evaluated as "poor."

Research and Monitoring Results

Broadcast Seeding—Robichaud and others (2000) reviewed published studies of seeding effectiveness after 1 and 2 years on 34 burned sites across the Western United States. Erosion was not measured at 16 sites (47 percent); only plant cover was determined. At another 15 of the seeding sites (44 percent), seeding did not significantly reduce erosion when compared to sites that were burned but not seeded. Soil erosion was reduced by seeding on only three sites (16, 31, and 80 percent less sediment). Of 23 monitoring reports that contained some quantitative data on broadcast seeding, 15 (65 percent) did not determine soil loss, and one reported no difference in erosion due to seeding. Three reports noted reductions in sediment yield of 30 to 36 percent (2.6 to 6.2 tons/acre or 5.8 to 13.8 Mg/ha). Four monitoring reports documented increases of sediment yield on seeded areas of 118 to 386 percent of that of burned and untreated areas.

In the studies and monitoring reports examined by Robichaud and others (2000), a wide variety of grass species, mixes, and application rates were used, making generalizations difficult. However, in some of the reported studies it is noted that grass seeding does not assure increased plant cover (or any associated erosion reduction) during the first critical year after fire. In the mid-1900s, southern California foresters were urged to caution the public not to expect significant first-year sediment control from postfire seeding (Gleason 1947). Krammes (1960), in southern California, found that as much as 90 percent of first-year postfire hillslope sediment movement can occur as dry ravel before the first germination-stimulating rains even occur. Amaranthus (1989) measured the most first-year sediment movement on his Oregon study site during several storms in December, before the seeded ryegrass had produced much cover. In the reported studies, erosion was decreased by seeding in only one out of eight first-year studies (12.5 percent). However, several studies showed a trend toward lower sediment movement on seeded plots that was not statistically significant (Amaranthus 1989, Wohlgemuth and others 1998). One report suggested that measures other than seeding should be used in places where first-year control of sediment movement is critical (Ruby 1997).

Better cover and, consequently, greater erosion control may occur by the second postfire year. Amaranthus (1989) reported that in the second year after fire, seeded sites had greater total cover (plant and litter) than unseeded 42 percent of the time. Seeded species are expected to be of greatest value during the second and third rainy seasons (Esplin and Shackleford 1978), when plant litter produced by the first year's growth covers the soil. However, after the Bobcat Fire in the Colorado Front Range, Wagenbrenner (2003) found that seeding had no significant effect on sediment yields at the hillslope scale in either the first or second years. In addition, seeding had no significant effect on percent of vegetative cover compared to untreated areas (Wagenbrenner 2003).

Seeding is often most successful where it may be needed least—on gentle slopes and in riparian areas. Janicki (1989) found that two-thirds of plots with more than 30 percent annual ryegrass cover were on slopes of less than 35 percent. He also noted that grass plants concentrated in drainage bottoms indicating that seed washed off the slopes during the first two storm events. Concentration of seeded species at the base of slopes was also observed by Loftin and others (1998).

Little evidence suggests that fertilizer applied with seeded grass is effective in increasing cover or reducing erosion after fire. Several studies found no significant effect of fertilizer on plant cover or erosion (Cline and Brooks 1979, Esplin and Shackleford 1980, Tyrrel 1981).

Retention of soil onsite for productivity maintenance is an important rehabilitation treatment objective, but almost no evidence indicates whether seeding is effective in meeting this goal. Although some nutrients are inevitably lost in a fire, natural processes tend to replenish the soil over time (DeBano and others 1998).

Mulch—Straw mulch applied at a rate of 0.9 ton/acre (2 Mg/ha) significantly reduced sediment yield on burned pine-shrub forest in Spain over an 18-month period with 46 rainfall events (Bautista and others 1996). Kay (1983) tested straw mulch laid down at rates of 0.5, 1.0, 1.5, and 4 tons/acre (1.1, 2.2, 3.4, and 9.0 Mg/ha) against jute excelsior, and paper for erosion control. Straw was the most cost-effective mulch, superior in protection to hydraulic mulches and comparable to expensive fabrics. Excelsior was less effective but better than paper strip synthetic yarn. The best erosion control came from jute applied over 1.5 tons/acre (3.4 Mg/ha) straw. Miles and others (1989) studied the use of wheat straw mulch on the 1987 South Fork of the Trinity River Fire, Shasta-Trinity National Forest in California. Wheat straw mulch was applied to fill slopes adjacent to perennial streams, firelines, and areas of extreme erosion hazard. Mulch applied at rates of 1.0 to 2.0 tons/acre (2.2 to 4.5 Mg/ha) on large areas, reduced erosion significantly 4.6 to 8.0 yard^3/acre (11 to 19 m^3/ha). They considered mulching highly effective in controlling erosion. Edwards and others (1995) examined the effects of straw mulching at rates of 0.9, 1.8, 2.6, and 3.6 tons/acre (2, 4, 6, and 8 Mg/ ha) on 5 to 9 percent slopes. They reported a significant reduction in soil loss at 0.9 ton/acre (2 Mg/ha) mulch, but increases in mulch thickness provided no additional reduction in soil loss. When comparing all the treatments used after the 2000 Cerro Grande Fire in New Mexico, mulching provided the best rehabilitation results. Although precipitation during the two study years was below normal, the plots treated with aerial seed and straw mulch yielded 70 percent less sediment than the no-treatment plots in the first year and 95 percent less in the second year. Ground cover transects showed that aerial seeding without added straw mulch provided no appreciable increase in ground cover relative to untreated plots (Dean 2001). In a 2-year postfire study in the Colorado Front Range, Wagenbrenner (2003) found that mulching reduced the erosion significantly from storms with return periods of up to 2 years. In the first study year, sediment yields from a high intensity (1.9 inches/hour, 48 mm/hour) rainfall event overwhelmed the silt fence sediment traps on both the treated and untreated study sites. However, in the second year, sediment yields from mulched hillslope sites were significantly less than the sediment yields from untreated slopes and the slopes that were seeded without mulch (Wagenbrenner 2003).

Contour Structures—Contour structures provide immediate benefits after installation in that they trap sediment during the first postfire year, which usually has the highest erosion rates. The ability of contour structures to reduce runoff and rilling, increase infiltration, and decrease downstream time-to-peak (slowing velocities) has not been documented, even though these are reasons often given for doing contour felling. If contour structures slow or eliminate runoff, sediment movement may not occur.

Logs were contour-felled on 22 acres (9 ha) of the 1979 Bridge Creek Fire, Deschutes National Forest in Oregon (McCammon and Hughes 1980). Trees 6 to 12 inches (150 to 300 mm) d.b.h. were placed and secured on slopes up to 50 percent at intervals of 10 to 20 feet (3 to 6 m). Logs were staked and holes underneath were filled. After the first storm event, about 63 percent of the contour-felled logs were judged effective in trapping sediment. The remainder were either partially effective or did not receive flow. Nearly 60 percent of the storage space behind contour-felled logs was full to capacity, 30 percent was half-full, and 10 percent had insignificant deposition. Common failures included flow under the log and not placing the logs on contour (more than 25 degrees off contour caused trap efficiency to decrease to 20 percent). More than 1,600 yard^3 (1,225 m^3) of material was estimated to be trapped behind contour-felled logs on the treated area, or about 73 yard^3/acre (135 m^3/ha). Less than 1 yard^3 (1 m^3) of sediment was deposited in the intake pond for a municipal water supply below.

The few monitoring studies done on contour-felled log treatments did not evaluate runoff, infiltration, or sediment movement changes after treatment installation; they only reported sediment storage. For example, Miles and others (1989) monitored contour-felling on the 1987 South Fork Trinity River Fires, Shasta-Trinity National Forest in California. The treatment was applied to 200 acres (80 ha) within 50,000 acres (20,240 ha) of a burned area. Trees less than 10 inches (250 mm) d.b.h. spaced 15 to 20 feet (4.5 to 6 m) apart were felled at rate of 80 to 100 trees/acre (200 to 250 trees/ha). The contour-felled logs trapped 0 to 0.07 yard^3 (0 to 0.05 m^3) of soil per log, retaining 1.6 to 6.7 yard^3/acre (3 to 13 m^3/ha) of soil onsite. Miles and others (1989) considered sediment trapping efficiency low, and the cost high for this treatment. Sediment deposition below treated areas was not measured. McCammon and Hughes (1980), on the other hand, estimated storage at just over 10 times the amounts reported by Miles and others (1989) using a higher density of logs. Depending on log barrier density and erosion rates, this treatment could trap 5 to 47 percent annual sediment production from high severity burn areas. This wide range of effectiveness indicates the need for proper estimation techniques of the erosion potential and for properly designing contour-felled log installations in terms of log numbers and spacing. If 60 percent or more of the expected sediment production can be trapped, then contour-felled logs are probably cost effective.

Dean (2001) found that plots treated with contour-felled logs as well as aerial seed and straw mulch yielded 77 percent less sediment in the first year and 96 percent in the second year as compared to untreated areas; however, these results were not significantly different from the straw mulch with seed treatment alone. Recent postfire rehabilitation monitoring efforts for six paired watersheds have indicated that contour-felled logs can be effective for low to moderate rainfall intensity storm events. However, during high intensity rainfall events, their effectiveness is greatly reduced. The effectiveness of contour-felled logs decreases over time. Once the sediment storage area behind the log is filled, the barrier can no longer trap sediment that is moving downslope (Robichaud 2000, Wagenbrenner 2003).

Contour Trenching—Contour trenches have been used as a rehabilitation treatment to reduce erosion, increase infiltration, trap sediment, and permit revegetation of fire-damaged watersheds. Although they do increase infiltration rates, the amounts are dependent on soils and geology (DeByle 1970b). Contour trenches can significantly improve revegetation by trapping more snow, but they do not affect water yield to any appreciable extent (Doty 1970, 1972). This treatment can be effective in altering the hydrologic response from short duration, high intensity storms typical of summer thunderstorms. However, this hillslope treatment does not significantly change peakflows resulting from low intensity, long duration rainfall events (DeByle 1970a).

Doty (1971) noted that contour trenching in the sagebrush (*Artemisia* spp.) portion (upper 15 percent with the harshest sites) of a watershed in central Utah did not significantly change streamflow and stormflow patterns. Doty did not discuss changes in sediment yields. There were no observable changes in seasonal flows or the total flow volumes. However, the storm peakflows were substantially reduced. Costales and Costales (1984) reported on the use of contour trenching on recently burned steep slopes (40 to 50 percent) with clay loam soils in pine stands of the Philippines. Contour trenching reduced sediment yield by 81 percent of comparable, burned, but untreated areas.

Other Treatments—Straw wattles may detain surface runoff, reduce velocities, store sediment, and provide a seedbed for germination. They are a good alternative in burned areas where logs are absent, poorly shaped, or scarce. Although the wattle netting is photodegradable, there are concerns that it persists long enough to pose hazards for small animals.

Cattle exclusion with temporary fencing can be important for the first two postfire years.

Ripping and scarification is effective on roads, trails, and firebreaks with slopes less than 35 percent. Slash spreading is effective if good ground contact is maintained.

Channel Treatments and Results

Channel Treatments

In general, channel treatments need to be coupled with hillslope treatments to be really effective. Channel treatments are implemented to modify sediment and water movement in ephemeral or small-order channels to prevent flooding and debris torrents that may affect downstream values at risk. Some in-channel structures are placed and secured to slow water flow and allow sediment to settle out; sediment will later be released gradually as the structure decays. Channel clearing is done to remove large objects that might become mobilized in a flood. Much less information has been published on channel treatments than on hillslope methods.

Straw Bale Check Dams—These structures are used to prevent or reduce sediment inputs into perennial streams during the first winter or rainy season following a wildfire. Straw bales function by decreasing water velocity and detaining sediment-laden surface runoff long enough for coarser sediments to drop out and be deposited behind the check dams. The decreased water velocity also reduces downcutting in ephemeral channels. Straw bale check dams are temporary in-channel grade control structures constructed of commercially available straw or hay bales. They are inexpensive, easy to install, and effective at trapping sediment but eventually deteriorate due to climatic conditions, streamflows, or cattle and wildlife disturbance. Straw bale check dams tend to fail in large storms. Failure can occur if the dams are poorly installed or put in locations where they cannot contain runoff. Straw bales are often used where materials are not available on site to construct check dams.

Log Check Dams—Log check dams are another type of temporary in-channel grade control structure similar in function to straw bale check dams. They are used to prevent or reduce sediment inputs into perennial streams during the first winter or rainy season following a wildfire. Log dams are constructed of more durable material than straw bale dams, usually small diameter fire-killed tree stems that are available nearby. Log check dams function by decreasing water velocity and detaining sediment-laden surface runoff long enough for coarser sediments to deposit behind check dams. Decreased water velocity also reduces downcutting in ephemeral channels. Log check dams require more effort and skill to install, but will last longer than straw bale check dams.

Rock Dams and Rock Cage Gabions—Also known as rock fence check dams, these structures are used in intermittent or small perennial channels to replace large woody debris that may have been burned out during a wildfire. The rock cage dams provide a degree of grade stability and reduce flow velocities long enough to trap coarse sediments. Properly designed and installed rock check dams and rock cage (gabion) dams are semipermanent structures capable of halting gully development and reducing sediment yields by controlling channel grade and stopping head cutting of gullies. The rock cage dams must be properly sited, keyed in, and anchored to stay in place during runoff events. The dam cages should be filled with angular rock that will interlock, preventing rock from mobilizing and pounding itself apart in the cages. Downslope energy dissipaters are recommended because they reduce the risk of the rock cage dams being undercut. Construction of these structures is dependent on the availability of adequate amounts and sizes of rocks. Rock cage dams usually need to be cleaned out periodically if they are to maintain their effectiveness.

Straw Wattle Dams—Straw wattle dams work on the same principle as straw bale check dams. They trap sediment on side slopes and in the upper ends of ephemeral drainages by reducing channel gradient. Straw wattles are easy to place in contact with the soil—a distinct advantage over rigid barriers like logs—and provide a low risk barrier to soil movement. The closer together straw wattles are placed in steep terrain, the more effective they are in detaining sediment. Wattles can be used quite effectively in combination with straw bale check dams. However, they should not be placed in the channels of first order or greater drainages and swales because of high failure rates. They are most effective on hillslopes.

Log Grade Stabilizers—The purpose of log grade stabilizers is much the same as log dams, except that the emphasis is on stabilizing the channel gradient rather than trapping sediment. Numerous small log grade stabilizers are preferable to a few larger ones. In some locations, there might not be adequate, straight, woody material left after a fire to build log grade stabilizers with onsite resources.

Rock Grade Stabilizers— Rock grade stabilizers function the same as log grade stabilizers, except that they are made of rock. The emphasis is on stabilizing the channel gradient rather than trapping sediment although some sediment will be trapped by these structures. Effectiveness is impacted by (1) the use of rocks that are large enough to resist transport during runoff events and (2) placement of screening to collect and hold organic debris or sediment on the upstream side of the grade stabilizer.

Channel Debris Clearing—Channel clearing is the removal of logs, organic debris, or sediment deposits to prevent them from being mobilized in debris flows or flood events. This treatment has been done to prevent creation of channel debris dams, which might result in flash floods or increase flood heights or peakflows. Organic debris can lead to culvert failure by blocking inlets or reducing channel flow capacity. Excessive sediments in stream channels can compromise in-channel storage capacity and the function of debris basins.

Streambank Armoring and Channel Armoring—Streambank and channel armoring is done to prevent erosion of channel banks and bottoms during runoff events. In some hydrologic systems, streambanks are a major source of sediment. Factors that contribute to the success of these treatments include proper sized materials, use of geotextile fabric, avoiding overly steep areas, and the use of energy dissipaters.

In-Channel Felling—This rehabilitation channel treatment is designed to replace woody material in drainage bottoms that have been consumed by wildfire. It is intended to trap organic debris and temporarily detain or slow down storm runoff. Woody material felled into channels will ultimately alter channel gradient and may cause sediment deposition and channel aggradation. This treatment is in conflict with channel removal of woody debris as its objectives are totally different.

Debris Basins— Debris basins are constructed in stream systems that, under normal conditions, carry high sediment loads. They are intended to control runoff and reduce deterioration of water quality and threats to human life and property. Debris basins are considered a last resort because they are extremely expensive to construct and require a commitment to annual maintenance until they are abandoned. In order for debris basins to function, they must be able to trap at least 50 percent and preferably 70 to 80 percent of 100-year flows. A spillway needs to be constructed in the debris basin to safely release flow in excess of the design storage capacity. The downstream channel should be lined to prevent scour (fig. 10.11. In some instances, excavated pits in ephemeral channels have been used as debris basins. These must be large enough to trap 50 to 90 percent of flood flow. Maintenance is a key factor in effectiveness of this treatment. Although protection is immediate, maintaining debris basins is a long-term commitment.

Channel Treatment Effectiveness

BAER Expert Ratings—Effectiveness ratings for straw bale check dams and log grade stabilizers ranged relatively evenly from "excellent" to "poor" (table 10.2). While most interviewees (71 percent) thought that channel debris clearing effectiveness fell into the "good" category, 29 percent rated it "poor." Log dams and straw wattle dams were rated "excellent" or "good" in effectiveness and better than rock grade stabilizers. No one considered the effectiveness of these rehabilitation treatments to be "poor."

Figure 10.11—Rocky Mountain Research Station Engineer Joe Wagenbrenner surveys a channel scour, following the Hayman Fire, 2002, Pike-San Isabel National Forest near Deckers, CO. (Photo by A. Covert).

Table 10.2—Rehabilitation treatment effectiveness ratings from individual fires as provided by BAER program specialists. Total responses are listed as percentages in four classes. Only treatments that received three or more evaluations are included (From Robichaud and others 2000).

BAER treatment	Fires	Excellent	Good	Fair	Poor
	Number	- - - - - - - -	*Percent*	- - - - - - - -	
Hillslope treatment					
Aerial seeding	83	24.1	27.7	27.7	20.5
Contour felling	35	28.6	37.1	14.3	20.0
Mulching	12	66.8	16.6	16.6	0.0
Ground seeding	11	9.1	81.8	9.1	0.0
Silt fence	8	37.5	62.5	0.0	0.0
Seeding and fertilizer	4	25.0	0.0	50.0	25.0
Rock grade stabilizers	3	0.0	33.3	67.7	0.0
Contour trenching	3	67.7	33.3	0.0	0.0
Temporary fencing	3	0.0	67.7	33.3	0.0
Straw wattles	3	33.3	33.3	33.3	0.0
Tilling/ripping	3	33.3	33.3	33.3	0.0
Channel treatments					
Straw bale check dams	10	30.0	30.0	30.0	10.0
Log grade stabilizers	10	30.0	30.0	10.0	30.0
Channel debris clearing	7	0.0	71.4	0.0	28.6
Log dams	5	40.0	60.0	0.0	0.0
Rock grade stabilizers	3	0.0	33.3	67.7	0.0
Straw wattle dams	3	33.3	67.7	0.0	0.0
Road treatments					
Culvert upgrading	6	6.7	66.6	0.0	16.7
Trash racks	4	50.0	0.0	25.0	25.0

Channel Treatment Research and Monitoring Results—Here we look at various results of interest to researchers and land managers.

Straw bale check dams: Straw bale check dams are designed to reduce sediment inputs into streams. They often fill in the first few storms, so their effectiveness diminishes quickly, and they can blow out during high flows. Thus, their usefulness is short-lived. Miles and others (1989) reported on the results of installing 1,300 straw bale check dams after the 1987 South Fork Trinity River Fires, Shasta-Trinity National Forest in California. Most dams were constructed with five bales. About 13 percent of the straw bale check dams failed due to piping under or between bales or undercutting of the central bale. Each dam stored an average 1.1 yard^3 (0.8 m^3) of sediment. Miles and others (1989) reported that filter fabric on the upside of each dam and a spillway apron would have increased effectiveness. They considered straw bale check dams easy to install and highly effective when they did not fail.

Collins and Johnston (1995) evaluated the effectiveness of straw bales on sediment retention after the Oakland Hills fire. About 5,000 bales were installed in 440 straw bale check dams and 100 hillslope barriers. Three months after installation, 43 to 46 percent of the check dams were functioning. This decreased to 37 to 43 percent by 4.5 months, at which time 9 percent were side cut, 22 percent were undercut, 30 percent had moved, 24 percent were filled, 12 percent were unfilled, and 3 percent were filled but cut. Sediment storage amounted to 55 yard^3 (42 m^3) behind all the straw bale check dams and another 122 yard^3 (93 m^3) on an alluvial fan.

Goldman and others (1986) recommended that the drainage area for straw bale check dams be kept to less than 20 acres (8 ha). Bales usually last less than 3 months, flow should not be greater than 11feet^3/sec (0.3 m^3/sec), and bales should be removed when sediment depth upstream is one-half of bale height. More damage can result from failed barriers than if no barrier were installed (Goldman and others 1986).

Log check dams: Log dams can trap sediment by decreasing velocities and allowing coarse sediment to drop out. However, if these structures fail, they usually aggravate erosion problems. Logs 12 to 18 inches (300 to 450 mm) diameter were used to build 14 log check dams that retained from 1.5 to 93 yard^3 (mean 29 yard^3) (1.1 to 71 m^3; mean 22 m^3) of sediment after the 1987 South Fork Trinity River Fires on the Shasta-Trinity National Forest in California

(Miles and others 1989). While log check dams have a high effectiveness rating and 15 to 30 years of life expectancy (Miles and others 1989), they are costly to install.

Rock dams and rock cage gabions: Heede (1970, 1976) reported that these structures can reduce sediment yields by 60 percent or more. Although these cross-channel structures are relatively expensive, they can be used in conjunction with vegetation treatments to reduce erosion by 80 percent and suspended sediment concentrations by 95 percent (Heede 1981). While vegetation treatments, such as grassed waterways, augment rock check dams and are less expensive, their maintenance costs are considerably greater.

On mild gradients, these structures work well. Some failures occurred on steeper slopes when high velocity flows are greater than 3 feet^3/sec (1 m^3/sec). This is a common theme for all channel treatments. Most of the failures occur where treatments are imposed on steep gradient sections of ephemeral or first to second order perennial channels. Rock cage dams often last long enough and trap enough fine sediments to provide microsites for woody riparian vegetation to get reestablished.

Check dams constructed in Taiwan watersheds with annual sediment yields of 10 to 30 yard^3/acre (19 to 57 m^3/ha) filled within 2 to 3 years. Sediment yield rates decreased upstream of the check dams but were offset by increased scouring downstream (Chiun-Ming 1985).

Other channel treatments: No published information was found on the effectiveness of straw wattle dams, log grade stabilizers, rock grade stabilizers, in-channel debris basins, in-channel debris clearing, streambank armoring, or other channel rehabilitation treatments. However, several considerations have been related to the effectiveness of these treatments.

Log and rock grade stabilizers emphasize channel stabilization rather than storing sediment. They tend to work for low and moderate flows, not high flows.

Channel clearing (removing logs and other organic debris) was rated "good" 71 percent of the time because it prevents logs from being mobilized in debris flow or floods. Nonetheless, use of this treatment has declined since the early 1990s (and in-channel felling has increased) because in-stream woody debris has been clearly linked to improved fish habitat. Despite the lack of effectiveness data for rock cage dams, they do provide grade stability and reduce velocities enough to drop out coarse sediment. Debris basins are designed to store runoff and sediment and are often used to prevent downstream flooding and sedimentation in the Southwestern United States. They are usually designed to trap 50 to 70 percent of the expected flows.

Road and Trail Treatments and Results

Road and Trail Treatments

Road rehabilitation treatments consist of a variety of practices aimed at increasing the water and sediment processing capabilities of roads and road structures, such as culverts and bridges, in order to prevent large cut-and-fill failures and the movement of sediment downstream. The functionality of the road drainage system is not affected by fire, but the increased stream and storm flows in a burned-over watershed can exceed the functionality of that system. Road treatments are not designed to retain water and sediment but rather to manage water's erosive forces and avoid damage to the road structures.

Rolling Dips/Waterbars/Cross Drain/Culvert Overflow/Bypass—These treatments are designed to provide drainage relief for road sections or water in the inside ditch to the downhill side of roads especially when the existing culvert is expected to be overwhelmed. Rolling dips are easily constructed with road grader, dozer, or backhoe. Rolling dips or waterbars need to be deep enough to contain the expected flow and location carefully assessed to prevent damages to other portions of the road prism. Waterbars can be made out of rocks or logs, but they are not as effective as earthen bars placed diagonally across roads to divert runoff away from the road surface. Armoring of dip and fillslope at the outlet is often needed to prevent incision and gullying.

Culvert Upgrades—Culvert improvements increase the flow capacity, which may prevent road damage. Upgraded culverts need to be sized and installed (approaches, exits, and slope) to handle expected increased flows. Flexible down spouts and culvert extensions often are needed to keep exiting water from highly erodible slopes.

Culvert Inlet/Outlet Armoring/Risers—These treatments reduce scouring around the culvert entrance and exit. They allow heavy particles to settle out of sediment-laden water and reduce the chance of debris plugging the culvert. Culvert risers allow for sediment accumulation while allowing water to flow through the culvert. Sometimes culvert risers can clog and may be difficult to clean.

Culvert Removal—This procedure is a planned removal of undersized culverts that would probably fail due to increased flows. After culvert removal, armoring the stream crossing will allow for continued use of the road. If the road is not needed, culvert removal is done in conjunction with road obliteration.

Trash Racks—Trash racks are installed to prevent debris from clogging culverts or down-stream structures. These structures are generally built out of logs but occasionally from milled lumber or metal, and they are anchored to the sides and bottom of the channel. Trash racks are sized to handle expected flows and protect downstream structures. Most trash rack designs allow debris to ride up and to the side of the cage. Trash racks generally perform better in smaller drainages and need to be cleared after each storm to be effective.

Storm Patrols—Patrols during storms provide immediate response for assessment of flood risk, clearing of blocked culvert entrances and drainage ditches, and closing areas that are at risk for floods, landslides, and so forth. This treatment can include early warning systems, such as radio-activated rain gauges or stream gauge alarms, that signal potential flooding conditions.

Ditch Cleaning and Ditch Armoring—Cleaning and armoring provides adequate water flow capacity and prevents downcutting of ditches. Without this treatment, high water levels can overtop roadways leading to gully development in roadbeds.

Armoring Ford Crossing—Armored crossings provide low-cost access across stream channels that are generally capable of producing large flows that flood the road surface. Large riprap is placed upstream and downstream of actual road crossing areas. Armored crossings are most often used for gravel roads.

Outsloping—Outsloping prevents concentration of flow on road surfaces that produces rilling, gullying, and rutting. This is one of the few rehabilitation treatments that have both immediate and long-term facility and resource benefits. Given that roads are a major source of sediment in forests, road improvements that reduce erosion from roads are beneficial. Sometimes after regrading, compaction does not occur due to low traffic volume, which may produce some short-term erosion. Traffic should be curtailed during wet road conditions to prevent rutting and road subgrade damages.

Trail Work—The purpose of rehabilitation treatments on trails is to provide adequate drainage and stability so trails do not contribute to concentrated flows or become sources of sediment. This treatment is labor intensive, as all the work must be done by hand with materials that can be hand carried or brought in on ATVs. Water bars need to be installed correctly at proper slopes and depths to be effective.

Other Treatments—A variety of other minor treatments are available as solutions to specific problems. They include wetting agents to reduce water repellency on high erosion hazard areas, gully plugs to prevent headcutting in meadows, flood signing installation to warn residents and visitors of flooding potential, and removal of loose rocks above roadways that were held in place by roots, forest debris, and duff consumed by fire.

Road Treatment Effectiveness

Road treatments are designed to move water to desired locations and prevent washout of roads. There is little quantitative research evaluating and comparing road treatment effectiveness. A recent computer model, X-DRAIN, can provide sediment estimates for various spacings of cross drains (Elliot and others 1998), and the computer model, WEPP-Road, provides sedimentation estimates for various road configurations and mitigation treatments (Elliot and others 1999). Thus, effectiveness of various spacings of rolling dips, waterbars, cross drains, and culvert bypasses can be compared.

BAER Expert Ratings—Only two road treatments—culvert upgrading and trash racks—received more than three effectiveness evaluations. The responses covered the range from “excellent” to “poor,” although 73 percent of the BAER experts rated culvert upgrading “excellent” or “good” in effectiveness (table 10.2). Evaluations of trash racks were evenly split as “excellent,” “fair,” or “poor.”

Research and Monitoring Results—Furniss and others (1998) developed an excellent analysis of factors contributing to road failures at culverted stream crossings. These locations are important because 80 to 90 percent of fluvial hillslope erosion in wildlands can be traced to road fill failures and diversions of road-stream crossings that are unrelated to wildfires (Best and others 1995). Because it is impossible to design and build all stream crossings to withstand extreme stormflows, they recommended increasing crossing capacity and designing to minimize the consequences of culvert exceedence as the best approaches for forest road stream crossings.

Comprehensive discussions of road-related treatments and their effectiveness can be found in Packer and Christensen (1977), Goldman and others (1986), and Burroughs and King (1989). Recently the USDA Forest Service’s San Dimas Technology and Development Program developed a Water/Road Interaction Technologies Series (Copstead 1997) that covers design standards, improvement techniques, and evaluates some surface drainage treatments for reducing sedimentation.

Summary, Conclusions, and Recommendations

Spending on postfire emergency watershed rehabilitation has steadily increased since about 1990. An evaluation of USDA Forest Service burned area emergency rehabilitation treatment effectiveness was completed jointly by the USDA Forest Service Research and Development and National Forest System staffs. The resulting study by Robichaud and others (2000) analyzed BAER treatment attributes and conditions that led to success or failure in achieving BAER goals after wildfires in the continental Western United States. The study found that spending on rehabilitation had risen sharply during the previous decade because the perceived threat of debris flows and floods had increased where fires were closer to the wildland-urban interface. Existing literature on treatment effectiveness is limited, thus making treatment comparisons difficult; however, the amount of protection provided by any treatment is limited—especially during short-duration, high-intensity rainfall events.

Relatively little monitoring of postfire rehabilitation treatments had been conducted between 1970 and 2000. During that time there were at least 321 fires that received BAER treatment, which cost the Forest Service around $110 million. Some level of monitoring occurred on about 33 percent of the fires that received BAER treatments. Since 2000, the number, size, and severity of fires in the Western United States have dramatically increased, with a concurrent increase in BAER spending to about $80 million annually. However, monitoring efforts continue to be short-term and inconsistent. Analysis of the literature, Burned Area Report forms, interview comments, monitoring reports, treatment effectiveness ratings, and the authors' continuing work in the area of postfire rehabilitation have led to the conclusions and recommendations discussed in this section.

Recommendations: Models and Predictions

- Rainfall intensity as well as rainfall amount and duration affect the success of rehabilitation treatment. Rehabilitation treatments are least ineffective in short-duration, high-intensity rainstorms (that is, convective thunderstorms), particularly in the first 2 years after burning.
- Quantitative data are needed to guide future responses to postfire rehabilitation and to build, test, and refine predictive models for different burned forest environments. Accurate climate, runoff, and erosion models for burned forest environments is dependent on (1) improved mapping of burn severity and better characterization of postfire soil water repellency, (2) improved prediction of runoff responses at different spatial scales, including short-duration high-intensity thunderstorms, (3) quantitative data for the relative magnitudes and consequences of hillslope verses channel erosion, and (4) refined sediment deposition and routing models for various drainages.

Recommendations: Postfire Rehabilitation Treatment

- Rehabilitation should be done only if the risk to life and property is high since significant resources have to be invested to ensure improvement over natural recovery. In most watersheds, it is best not to do any treatments. If treatments are necessary, then it is more effective to detain runoff and reduce erosion on site (hillslope treatment) than to collect it downstream (channel treatment).
- Seeding treatment may not be needed as often as previously thought. Seeding has a low probability of reducing erosion the first wet season after a fire when erosion rates are highest. Thus, it is often necessary to do other treatments in critical areas.
- Mulching can be an effective treatment because it provides immediate protection from raindrop impact and overland flow. Mulching rates that provide at least 70 percent ground cover are desirable in critical areas.
- Contour-felled log structures have limited benefit as an effective treatment compared to other hillslope treatments if: (a) the density and size of the felled logs are matched to the expected erosion, (b) the logs, basins, and sills are properly located, and (c) the treated area is not likely to be subjected to short-duration high-intensity storms. This is considered to be true for areas where runoff and erosion rates are expected to be high. These treatments provide storm-by-storm protection during the first year postfire where runoff and erosion rates are highest. In areas that lack available trees, straw wattles can provide an alternative. However, the overall effectiveness of properly designed and constructed contour-felled logs and straw wattles needs adequate study and documentation in the scientific literature.
- Channel treatments, such as straw bale check-dams, should be viewed as secondary mitigation treatments. Sediment has already been transported from the hillslopes and

will eventually be released though the stream system unless it is physically removed from the channel. Most channel treatments hold sediment temporarily so that the release is desynchronized from the storm flow event.

- To reduce the threat of road failure, road treatments such as rolling dips, water bars, and relief culverts, properly spaced, provide a reasonable method to move water past the road prism. Storm patrol attempts to keep culverts clear and close areas as needed. This approach shows promise as a cost effective technique to reduce road failure due to culvert blockage.
- The development and launch of the Web-based database of past and current BAER projects should be expedited so that future decisions are based on the best data available. This database will include treatment design criteria and specifications, contract implementation specifications, example Burned Area Report calculations, and monitoring techniques. This database needs to be kept current as new information is obtained.

Recommendations: Effectiveness monitoring

- The need for improved effectiveness monitoring has gained momentum. In April 2003, the Government Accounting Office published a report entitled *Wildland Fires: Better Information Needed on Effectiveness of Emergency Stabilization and Rehabilitation Treatments,* which clearly stated that neither the USDA Forest Service nor the Department of the Interior's Bureau of Land Management could determine whether emergency stabilization and rehabilitation treatments were achieving their intended results. Although treatment monitoring is required, there is no agreed-on protocol for how and what to collect and analyze for determining effectiveness. Such protocol needs to be pursued.
- Effectiveness monitoring needs to be initiated as quickly as possible after treatments are applied, as the first storms typically pose the greatest risk to downstream resources. Effectiveness monitoring should include quantifying reductions in erosion, sedimentation, and/or downstream flooding. It may also include measurement of changes in infiltration, soil productivity, ecosystem recovery, and water quality parameters. Burned but untreated areas must be available to provide a control, or baseline, from which to assess both short- and long-term effectiveness of treatments as well as ecosystem response to the fire and natural recovery rates. Recently published techniques are now available that may aid in the development of monitoring protocols (Robichaud and Brown 2002).
- Funding for effectiveness monitoring must be part of the BAER treatment funding request and extend for at least 5 years after the fire. This necessitates a change in BAER funding protocol, which currently is limited to 2 years.
- Policy and funding mechanisms should be established to take advantage of the overlap between research and monitoring by supporting activities that can accomplish the goals of both programs. This should include testing of new rehabilitation technologies as they become available.

Notes

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Chapter 11: Information Sources

Introduction

New research and development result in continually improved information on the effects of fire on soils and water. Nevertheless, as the wildfire seasons of the recent past point out, there is need for additional work in this area. Also, expansion of populations in the Western United States have placed greater demands on forested watersheds to provide stable supplies of water for municipalities. And more physical resources are now at risk from postfire streamflow events.

This volume updates the information available on impacts of fire on soils and water, to a given point in time. Barring more frequent updates of this volume (originally published in 1979; Wells and others 1979, Tiedemann and others 1979), future information retrieval must be dynamic and current.

This chapter outlines additional sources of information, particularly those that are likely to be easily updated and accessible. We have attempted to identify some of the more common places where fire personnel may search for general and specific fire effects and associated fire environment information. Examples shown are meant to be illustrative and not inclusive of all sources.

Databases

U.S. Fire Administration

The U.S. Fire Administration Federal Fire Links database is an online, searchable database containing links to Web sites in a variety of categories that are related to fire and emergency services. Location:

http://www.usfa.fema.gov/applications/fflinks/

Current Wildland Fire Information

Up-to-date information on current wildland fire situations, statistics, and other information is maintained by the National Interagency Fire Center's Current Wildland Fire Information site. It can be found at:

http://www.nifc.gov/information.html

Fire Effects Information System

The Fire Effects Information System (FEIS) provides up-to-date information about fire effects on plants and animals. The FEIS database contains nearly 900 plant species, 100 animal species, and 16 Kuchler plant communities found in North America. Each synopsis emphasizes fire and how it affects each species. Synopses are documented with complete bibliographies. Several Federal agencies provide maintenance support and updating of the database. It is located at:

http://www.fs.fed.us/database/feis/

National Climatic Data Center

The National Climatic Data Center (NCDC) database contains the largest active archive of weather data. NCDC provides numerous climate publications and provides access to data from all NOAA Data Centers through the National Virtual Data System (NVDS). Its Web site is located at:

http://lwf.ncdc.noaa.gov/oa/ncdc.html

PLANTS

The PLANTS Database is a single source of standardized information about vascular plants, mosses, and lichens of the United States and its Territories. PLANTS includes names, checklists, identification information, distributional data, references, and so forth. The database will have threatened and endangered plant status for States.Web site is located at:

http://plants.usda.gov/home_page.html

Fire Ecology Database

The E. V. Komarek Fire Ecology Database contains a broad range of fire-related information. Literature on control of wildfires and applications of prescribed burning is included. Citations include reference books, chapters in books, journal articles, conference papers, State and Federal documents. The database contains more than 10,000 citations and is updated on a continuous basis, with both current and historical information. The Tall Timbers Fire Ecology Thesaurus, a guide for doing keyword searches, is also available to be downloaded. This Web site is located at:

http://www.ttrs.org/fedbintro.htm

Wildland Fire Assessment System

The Wildland Fire Assessment System (WFAS) generates daily national maps of selected fire weather and fire danger components of the National Fire Danger Rating System (NFDRS). It was developed by the USDA Forest Service Fire Sciences Laboratory in Missoula, MT. Maps available include Fire Danger, Fire Weather Observations and Next Day Forecasts, Dead Fuel Moisture, Live Fuel Moisture-Greenness, Drought, Lower Atmosphere Stability Index, and Lightning Ignition Efficiency. Its location is:

http://www.fs.fed.us/land/wfas/welcome.htm

National FIA Database Systems

The National Forest Inventory and Analysis (FIA) Database Retrieval System produces tables and maps for geographic areas of interest based on the national forest inventory conducted by the USDA Forest Service. The Timber Product Output (TPO) Database Retrieval System describes for each county the round-wood products harvested, logging residues left behind, and wood/bark residues generated by wood-using mills. The Web site is:

http://ncrsz.fs.fed.us/4801/fiadb/rpa_tpo/wc_rpa_tpo.asp

Web Sites

A number of sites on the World Wide Web contain information on fire effects. These include research, fire coordination centers, and other related sites.

USDA Forest Service, Rocky Mountain Research Station Wildland Fire Research Program, Missoula, Montana

Fire Behavior Research Work Unit RMRS-4401:

http://www.fs.fed.us/rm/main/labs/miss_fire/rmrs4401.html

Fire Effects Research Work Unit RMRS-4403:

http://www.fs.fed.us/rm/main/labs/miss_fire/rmrs4403.html

Fire Chemistry Research Work Unit RMRS-4404:

http://www.fs.fed.us/rm/main/labs/miss_fire/rmrs4404.html

USDA Forest Service, Rocky Mountain Research Station Watershed, Wildlife, and Wildland-Urban Interface Fire Research Program, Flagstaff, Arizona

Watershed Research Unit RMRS-4302

http://www.rmrs.nau.edu/lab/4302/

Wildland-Urban Interface Research Unit RMRS-4156

http://www.rmrs.nau.edu/lab/4156/

Wildlife Research Unit

http://www.rmrs.nau.edu/lab/4251/

USDA Forest Service, Pacific Northwest Research Station Pacific Wildlands Fire Science Laboratory, Seattle, Washington

http://www.fs.fed.us.nw/pwfsl/

USDA Forest Service, Pacific Southwest Research Station, Fire Science Laboratory, Riverside, California

http://www.fs.fed.us/psw/rfl/

Fire and Fire Surrogate Program

USDA Forest Service Fire and Fire Surrogate Treatments for Ecosystem Restoration:

http://www.fs.fed.us/ffs/

National Fire Coordination Centers

National Interagency Coordination Center:
http://www.nifc.gov/news/nicc.html

Alaska Coordination Center:
http://fire.ak.blm.gov

Eastern Area Coordination Center:

http://www.fs.fed.us/eacc

Eastern Great Basin Coordination Center:

http://gacc.nifc.gov/egbc/

Northern California Geographic Area Coordination Center:

http://gacc.nifc.gov/oncc/

Northern Rockies Coordination Center:

http://gacc.nifc.gov/nrcc/

Northwest Interagency Coordination Center:

http://www.nwccweb.us/

Rocky Mountain Coordination Center:

http://www.fs.fed.us/r2/fire/rmacc.html

Southwest Coordination Center:

http://gacc.nifc.gov/swcc/

Southern Area Coordination Center:

http://gacc.nifc.gov/sacc/

Southern California Coordination Center:

http://gacc.nifc.gov/oscc/

Western Great Basin Coordination Center:

http://gacc.nifc.gov/wgbc/

USDA Forest Service Fire & Aviation Management Web site:

http://www.fs.fed.us/fire/

Bureau of Land Management Office of Fire and Aviation:

http://www.fire.blm.gov/

National Park Service Fire Management Program Center:

http:data2.itc.gov/fire/index.cfm

U.S. Fish and Wildlife Service Fire Management:

http://fire.r9.fws.gov/

Other Web Sites

Joint Fire Sciences web site:

http://jfsp.nifc.gov

Natural Resources Canada, Forest Fire in Canada

http://fire.cfs.nrcan.gc.ca/index_e.php

Mexico Incendios Forestales Info

http://www.incendiosforestales.info/

Federal Emergency Management Agency:

http://www.fema.gov/

Smokey Bear Web site:

http://smokeybear.com/

USDA-ARS-National Sedimentation Laboratory Web site:

http://ars.usda.gov/main/site_main.htm?modecode=64_08_05_00

Firewise Communities Web site:

http://www.firewise.org/

U.S. Geological Survey Fire Research Program

http://www.usgs.gov/themes/wildfire.html

Laboratory of Tree-Ring Research, University of Arizona:

http://www.ltrr.arizona.edu/

Wildfire News:

http://www.wildfirenews.com

Smokejumpers:

http://fs.fed.us/fire/people/smokejumpers/
http://fire.blm.gov/smokejumper/

A-10 Warthog Air Tankers:

http://FireHogs.com

Fire Weather (National Weather Service, Boise):

http://www.boi.noaa.gov/

Textbooks

The following is a list of textbooks that are cited in various chapters of this or the other volumes in this series that deal with fire effects on vegetation, soils, and water:

1. Biswell, Harold H. 1989. *Prescribed Burning in California Wildlands Vegetation Management.* University of California Press. 255 pp.
2. Bond, William J., and Brian W. vanWilgen. 1996. *Fire and Plants.* Chapman & Hall, London. 263 pp.
3. Bradstock, Ross A., Jann E. Williams, and A. Malcolm Gill, editors. 2002. *Flammable Australia, the Fire Regimes and Biodiversity of a Continent.* Cambridge University Press. 462 pp.
4. Collins, Scott L., and Linda L. Wallace, editors. 1990. *Fire in North American Tallgrass Prairies.* University of Oklahoma Press. 175 pp.
5. DeBano, Leonard F., Daniel G. Neary, and Peter F. Ffolliott. 1998. *Fire's Effects on Ecosystems.* John Wiley & Sons, New York. 333 pp.
6. Kozlowski, T. T., and C. E. Ahlgren, editors. 1974. *Fire and Ecosystems.* Academic Press, Inc., New York. 542 pp.
7. Pyne, Stephen J. 1982. *Fire in America—A Cultural History of Wildland and Rural Fire.* University of Washington Press. 654 pp.
8. Pyne, Stephen J. 2001. *Fire—A Brief History.* University of Washington Press. 204 pp.
9. Pyne, Stephen J., Patricia L. Andrews, and Richard D. Laven. 1996. *Introduction to Wildland Fire.* 2nd Edition. John Wiley & Sons, New York. 769 pp.
10. Tere, William C. 1994. *Firefighter's Handbook on Wildland Firefighting—Strategy, Tactics and Safety.* Deer Valley Press, Rescue, CA. 313 pp.
11. Whelan, Robert J. 1995. *The Ecology of Fire.* Cambridge University Press. 346 pp.
12. Wright, Henry A., and Arthur W. Bailey. 1982. *Fire Ecology, United States and Southern Canada.* John Wiley & Sons, New York. 501 pp.

Journals and Magazines

Journals and magazines constitute another traditional source of fire effects information. Some of the useful ones include:

Fire Management Today (formerly *Fire Management Notes*). Superintendent of Documents, P.O. Box 371954, Pittsburgh, PA 15250-7954. Also available at:

http://www.fs.fed.us/fire/fmt/index.htm

Wildfire. Official publication of the International Association of Wildland Fire, 4025 Fair Ridge Drive, Suite 300, Fairfax, VA 22033-2868. Also available at:

http://www.iawfonline.org/

Wildland Firefighter. Wildland Firefighter, P.O. Box 130, Brownsville, OR 97327. Also available at:

http://wildlandfire.com

International Journal of Wildland Fire. CSIRO Publishing, P.O. Box 1139, Collingwood, Victoria 3066, Australia. Also available at:

http://www.publish.csiro.au/nid/114.htm

Journal of Forestry, Forest Science and The Forestry Source. Society of American Foresters, 5400 Grosvenor Lane, Bethesda, MD. 20814-2198. Also includes its regional journals. Also available at:

http://safnet.org/periodicals/

Forest Ecology and Management. Elsevier Science, Regional Sales Office, Customer Support Department, P.O. Box 945, New York, NY 10159-0945. Also available at:

http://www.elsevier.com/locate/issn/foreco

Journal of Range Management. Society for Range Management, 445 Union Blvd, Suite 230, Lakewood, CO 80228. Also available at:

http://uvalde.tamu.edu/jrm/jrmhome.htm

Canadian Journal of Forest Research. NRC Research Press, National Research Council of Canada, Ottawa, ON K1A 0R6. Also available at:

http://www.cif-ifc.org/engusu/e-cjfr-spec-sub.shtml

Environmental Science and Technology. American Chemical Society, 1155 16th St., N.W., Washington, DC 20036. Also available at:

http://pubs.acs.org/journals/esthag/index.html

Journal of Environmental Quality. American Society of Agronomy, 677 South Segoe Road, Madison, WI 53711. Also available at:

http://jeq.scijournals.org/

Ecology and *Ecological Applications*. Ecological Society of America, Suite 400, 1707 H Street NW, Washington, DC 20006. Also available at the following Web site:

http://www.esapubs.org/publications/

Journal of the American Water Resources Association. 4 West Federal Street, P.O. Box 1626, Middleburg VA 20118-1626. Also available at:

http://www.awra.org/publicationindex.htm

Journal of Wildlife Management. The Wildlife Society, 5410 Grosvenor Lane, Bethesda, MD 20814-2197. Also available at:

http://www.wildlife.org/publications/

Other Sources

Other information sources for fire effects on soils and water include USDA Forest Service Research Station reports, bulletins, notes and other publications:

North Central Forest Experiment Station, 1992 Folwell Avenue, St. Paul, MN 55108

http://www.ncrs.fs.fed.us/

Northeastern Research Station, 11 Campus Boulevard, Newton Square, PA 19073

http://www.fs.fed.us/ne/

Pacific Northwest Research Station, P.O. Box 3890, Portland, OR 97208-3890

http://www.fs.fed.us/pnw/

Pacific Southwest Research Station, P.O. Box 245, Berkeley, CA 94701-0245

http://www.psw.fs.fed.us/psw/

Rocky Mountain Research Station, 2150 Centre Avenue, Fort Collins, CO. 80526

http://www.fs.fed.us/rm/

Southern Research Station, 200 Weaver Boulevard, P.O. Box 2680, Asheville, NC 28802

http://www.srs.fs.fed.us/

National Fire Plan is a cooperative effort of the USDA Forest Service, Department of the Interior, and the National Association of State Foresters for managing the impacts of wildfires on communities and the environment. It publishes annual attainment and information reports as well as maintaining a Web site:

http://www.fireplan.gov/

The National Incident Information Center Morning Fire Report, issued during periods of high fire activity, provides up-to-date fire activity and ecosystem impacts information at:

http://www.fs.fed.us/news/fire/

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Chapter 12: Summary and Research Needs

Volume Objective

The objective of this volume is to provide an overview of the state-of-the-art understanding of the effects of fire on soils and water in wildland ecosystems. Our challenge was to provide a meaningful summary for North American fire effects on these resources despite enormous variations produced by climate, topography, fuel loadings, and fire regimes. This volume is meant to be an information guide to assist land managers with fire management planning and public education, and a reference on fire effects processes, pertinent publications, and other information sources. Although it contains far more information and detailed site-specific effects of fire on soils and water than the original 1979 Rainbow volumes, it is not designed to be a comprehensive research-level treatise or compendium. That challenge is left to several textbooks (Chandler and others 1991, Agee 1993, Pyne and others 1996, DeBano and others 1998).

Soil Physical Properties Summary

The physical processes occurring during fires are complex and include both heat transfer and the associated change in soil physical characteristics. The most important soil physical characteristic affected by fire is soil structure because the organic matter component can be lost at relatively low temperatures. The loss of soil structure increases the bulk density of the soil and reduces its porosity, thereby reducing soil productivity and making the soil more vulnerable to postfire runoff and erosion. Although heat is transferred in the soil by several mechanisms, its movement by vaporization and condensation is the most important. The result of heat transfer in the soil is an increase in soil temperature that affects the physical, chemical, and biological properties of the soil. When organic substances are moved downward in the soil by vaporization and condensation, they can cause a water-repellent soil condition that further accentuates postfire runoff and extensive networks of surface rill erosion or erosion by raindrop splash. The magnitude of change in soil physical properties depends on the temperature threshold of the soil properties and the severity of the fire. The greatest change in soil physical properties occurs when smoldering fires burn for long periods.

Soil Chemistry Summary

The most basic soil chemical property affected by soil heating during fires is organic matter. Organic matter not only plays a key role in the chemistry of the soil, but it also affects the physical properties (see chapter 2) and the biological properties (see chapter 4) of soils as well. Soil organic matter plays a key role in nutrient cycling, cation exchange, and water retention in soils. When organic matter is combusted, the stored nutrients are either volatilized or are changed into highly available forms that can be taken up readily by microbial organisms and vegetation. Those available nutrients not immobilized are easily lost by leaching or surface runoff and erosion. Nitrogen is the most important nutrient affected by fire, and it is easily volatilized and lost from the site at relatively low temperatures. The amount of change in organic matter and nitrogen is directly related to the magnitude of soil heating and the severity of the fire. High- and moderate-severity fires cause the greatest losses. Nitrogen loss by volatilization during fires is of particular concern on low-fertility sites because N can only be replaced by N-fixing organisms. Cations are not easily volatilized and usually remain on the site in a highly available form. An abundance of cations can be found in the thick ash layers (or ash-bed) remaining on the soil surface following high-severity fires.

Soil Biology Summary

Soil microorganisms are complex. Community members range in activity from those merely trying to survive, to others responsible for biochemical reactions that are among the most elegant and intricate known. How they respond to fire will depend on numerous factors, including fire intensity and severity, site characteristics, and preburn community composition. Some generalities can be made, however. First, most studies have shown strong resilience by microbial communities to fire. Recolonization to preburn levels is common, with the amount of time required for recovery generally varying in proportion to fire severity. Second, the effect of fire is greatest in the forest floor (litter and duff). Prescriptions that consume major fuels but protect forest floor, humus layers, and soil humus are recommended.

Fire and Streamflow Regimes Summary

Fires affect water cycle processes to a greater or lesser extent depending on severity. Fires can produce some substantial effects on the streamflow regime of both small streams and rivers, affecting annual and seasonal water yield, peakflows and floods, baseflows, and timing of flows. Adequate baseflows are necessary to support the continued existence of many wildlife populations. Water yields are important because many forest, scrubland, and grassland watersheds function as municipal water supplies. Peakflows and floods are of great concern because of their potential impacts on human safety and property. Next to the physical destruction of a fire itself, postfire floods are the most damaging aspect of fire in the wildland environment. It is important that resource specialists and managers become aware of the potential of fires to increase peakflows.

Following wildfires, flood peakflows can increase dramatically, severely affecting stream physical conditions, aquatic habitat, aquatic biota, cultural resources, and human health and safety. Often, increased flood peakflows of up to 100 times those previously recorded, well beyond observed ranges of variability in managed watersheds, have been measured after wildfires. Potentials exist for peak flood flows to jump to 2,300 times prewildfire levels. Managers must be aware of these potential watershed responses in order to adequately and safely manage their lands and other resources in the postwildfire environment.

Water Quality Summary

When a wildland fire occurs, the principal concerns for change in water quality are: (1) the introduction of sediment; (2) the potential for increasing nitrates, especially if the foliage being burned is in an area of chronic atmospheric deposition; (3) the possible introduction of heavy metals from soils and geologic sources within the burned area; and (4) the introduction of fire retardant chemicals into streams that can reach levels toxic to aquatic organisms.

The magnitude of the effects of fire on water quality is primarily driven by fire severity, and not necessarily by fire intensity. Fire severity is a qualitative term describing the amount of fuel consumed, while fire intensity is a quantitative measure of the rate of heat release (see chapter 1). In other words, the more severe the fire the greater the amount of fuel consumed and nutrients released and the more susceptible the site is to erosion of soil and nutrients into the stream where it could potentially affect water quality. Wildfires usually are more severe than prescribed fires. As a result, they are more likely to produce significant effects on water quality. On the other hand, prescribed fires are designed to be less severe and would be expected to produce less effect on water quality. Use of prescribed fire allows the manager the opportunity to control the severity of the fire and to avoid creating large areas burned at high severity.

The degree of fire severity is also related to the vegetation type. For example, in grasslands the differences between prescribed fire and wildfire are probably small. In forested environments, the magnitude of the effects of fire on water quality will probably be much lower after a prescribed fire than after a wildfire because of the larger amount of fuel consumed in a wildfire. Canopy-consuming wildfires would be expected to be of the most concern to managers because of the loss of canopy coupled with the destruction of soil aggregates. These losses present the worst-case scenario in terms of water quality. The differences between wild and prescribed fire in shrublands are probably intermediate between those seen in grass and forest environments.

Another important determinant of the magnitude of the effects of fire on water quality is slope. Steepness of the slope has a significant influence on movement of soil and nutrients into stream channels where it can affect water quality. Wright and others (1976) found that as slope increased in a prescribed fire, erosion from slopes is accelerated. If at all possible, the vegetative canopy on steep, erodible slopes needs to be maintained, particularly if adequate streamside buffer strips do not exist to trap the large amounts of sediment and nutrients that can be transported quickly into the stream channel. It is important to maintain streamside buffer strips whenever possible, especially when developing prescribed fire plans. These buffer strips will capture much of the sediment and nutrients from burned upslope areas.

Nitrogen is of concern to water quality. If soils on a particular site are close to N saturation, it is possible to exceed maximum contamination levels of NO_3-N (10 ppm or 10 mg/L) after a severe fire. Such areas should not have N-containing fertilizer applied after the fire. Chapter 3 contains more discussion of N. Fire retardants typically contain large amounts of N, and they can cause water quality problems where drops are made close to streams.

The propensity for a site to develop water repellency after fire must be considered (see chapter 2). Water-repellent soils do not allow precipitation to penetrate down into the soil and therefore are conducive to erosion. Severe fires on such sites can put large amounts of sediment and nutrients into surface water.

Finally, heavy rain on recently burned land can seriously degrade water quality. Severe erosion and runoff are not limited to wildfire sites alone. But if postfire storms deliver large amounts of precipitation or short-duration, high-intensity rainfalls, accelerated erosion and runoff can occur even after a carefully planned prescribed fire. Conversely, if below-average precipitation occurs after a wildfire, there may not be a substantial increase in erosion and runoff and no effect on water quality.

Fire managers can influence the effects of fire on water quality by careful planning before prescribed burning. Limiting fire severity, avoiding burning on steep slopes, and limiting burning on potentially water-repellent soils will reduce the magnitude of the effects of fire on water quality.

Aquatic Biota Summary

The effects of wildland fire on fish are mostly indirect in nature. There are some documented instances of fires killing fish directly. The largest problems arise from the longer term impact on habitat. This includes changes in stream temperature due to plant understory and overstory removal, ash-laden slurry flows, increases in flood peakflows, and sedimentation due to increased landscape erosion. Most information on the effects of wildfire on fishes has been generated since about 1990. Limited observational information exists on the immediate, direct effect of fire on fishes. Most information is indirect and demonstrates the effects of ash flows, changes in hydrologic regimes, and increases in suspended sediment on fishes. These impacts are marked, ranging from 70 percent to total loss of fishes. The effects of fire retardants on fishes are observational and not well documented at present. Further, all information is from forested biomes as opposed to grasslands.

Anthropogenic influences, largely land use activities over the past century, cumulatively influence fire effects on fishes. Fire suppression alone has affected vegetation densities on the landscape and the severity and extent of wildfire and, in turn, its effects on aquatic ecosystems and fishes. Most all studies of fire effects on fishes are short term (less than 5 years) and local in nature. A landscape approach to analyses has not been made to date. Fish can recover rapidly from population reductions or loss but can be markedly limited or precluded by loss of stream connectivity imposed by human-induced barriers. Fisheries management postfire should be based on species and fisheries and their management status. Managers should be vigilant of opportunities to restore native fishes in event of removal of introduced, nonnative, translocated species.

Although the impact of fire on fishes appears to be marked, a larger impact may loom in the future. Recent advances in atmospheric, marine, and terrestrial ecosystem science have resulted in the correlation of ocean temperature oscillations, tree ring data, and drought. Based on current information on fire effects on fishes, combined with that of climate change, drought, and recent insect infestations, the greatest impacts on fish may lie ahead. The information emerging in new climate change analyses suggests that the Southeastern and parts of the Western United States have a high probability of continuing and future drought, with potential impact of wildfire on fishes in the Southwest. Based on some of these overarching global indicators, the rest of the story is starting to unfold.

The effects of fires on reptile and amphibian populations are also covered in volume 1 of this series (Smith 2000). A review by Russell and others (1999) concluded that there are few reports of fire-caused injury to herpetofauna in general much less aquatic and wetland species.

Similar to fishes, the recorded effects of fire on aquatic macroinvertebrates are indirect as opposed to direct. Response varies from minimal to no changes in abundance, diversity, or richness, to considerable and significant changes. Abundance of macroinvertebrates may actually increase in fire-affected streams, but diversity generally is reduced. These differences are undoubtedly related to landscape variability, burn size and severity, stream size, nature and timing of postfire flooding events, and postfire time. Temporally, changes in macroinvertebrate indices in the first 5 years postfire can be different from ensuing years, and long-term (10 to 30 years) effects have been suggested.

The effects of fires on bird populations are covered in volume 1 of this series (Smith 2000). Aquatic areas and wetlands often provide refugia during fires. However, wetlands such as cienegas, marshes, cypress swamps, spruce, and larch swamps do burn under the right conditions. The impacts of fires on individual birds and populations in wetlands would then depend upon the season, uniformity, and severity of burning (Smith 2000).

Aquatic and wetland dwelling mammals are usually not adversely impacted by fires due to animal mobility and the lower frequency of fires in these areas. Lyon and others (2000) discuss factors such as fire uniformity, size, duration, and severity that affect mammals. However, most of their discussion relates to terrestrial mammals, not aquatic and wetland ones. Aquatic and wetland habitats also provide safety zones for mammals during fires.

Wetlands Summary

While the connection between wetlands and fire appears incongruous, fire plays an integral role in the creation and persistence of wetland species and ecosystems. Wetland systems have come under various classification systems, reflecting both the current understanding of wetland processes and the missions of the individual land management agencies. We examined some of the complex water inflows and outflows that are balanced by the interrelationships between biotic and abiotic factors. The presence and movement of water are dominant factors controlling the interactions of fire with wetland dynamics, nutrient and energy flow, soil chemistry, organic matter decomposition, and plant and animal community composition.

In wetland soils, the effects produced by surface and ground fires are related to the intensity and duration of fire at either the soil surface or the interface between burning and nonburning organic soil materials. In general, surface fires can be characterized as short duration, variable severity ones, while ground fires can be characterized as having longer duration and lower severity. However, the latter type of fires can produce profound physical and chemical changes in wetlands.

Models Summary

A number of older modeling technologies are commonly used in estimating fire effects during and after fire (FOFEM, WATSED, WEPP, RUSLE, and others). Newer models include DELTA-Q and FOREST, and others are under construction. These process-based models provide managers with additional tools to estimate the magnitude of fire effects on soil and water produced by land disturbance. FOFEM was developed to meet the needs of resource managers, planners, and analysts in predicting and planning for fire effects. Quantitative predictions of fire effects are needed for planning prescribed fires that best accomplish resource needs, for impact assessment, and for long-range planning and policy development. FOFEM was developed to meet this information need. The WATSED technology was developed for watershed analysis. The RUSLE model was developed for agriculture and rangeland hillslopes and has been extended to forest lands. The WEPP model was designed as an improvement over RUSLE that can either be run as a stand-alone computer model by specialists, or accessed through a special Internet interface designed for forest applications, including wild fires.

All of these models have limitations that must be understood by fire managers or watershed specialists before they are applied. The models are only as good as the data used to create and validate them. Some processes such as extreme flow and erosion events are not simulated very well because of the lack of good data or the complexity of the processes. However, they do provide useful tools to estimate landscape changes to disturbances such as fire. Potential users should make use of the extensive documentation of these models and consult with the developers to ensure the most appropriate usage of the models.

Watershed Rehabilitation Summary

Spending on postfire emergency watershed rehabilitation has steadily increased since 1990. An evaluation of USDA Forest Service burned area emergency rehabilitation treatment effectiveness was completed jointly by the USDA Forest Service Research and Development and National Forest System staffs. Robichaud and others (2000) collected and analyzed information on past use of BAER treatments in order to determine attributes and conditions that led to treatment success or failure in achieving BAER goals after wildfires in the continental Western United States. The study found that spending on rehabilitation has risen sharply recently because the perceived threat of debris flows and floods has increased where fires are closer to the wildland-urban interface. Existing literature on treatment effectiveness is limited, thus making treatment comparisons difficult; however, the amount of protection provided by any treatment is limited—especially during short-duration, high-intensity rainfall events.

Relatively little monitoring of postfire rehabilitation treatments has been conducted in the last three decades. In the three decades prior to 2000, there were at least 321 fires that received BAER treatment, which cost the Forest Service around $110 million. Some level of monitoring occurred on about 33 percent of these project fires. In the early 2000s, the number, size, and severity of Western United States fires dramatically increased, with a concurrent increase in BAER spending of about $50 million annually. Monitoring efforts continue to be short term and inconsistent. Analysis of the literature, Burned Area Report forms, interview comments, monitoring reports, treatment effectiveness ratings, and the authors' continuing work in the area of postfire rehabilitation have led to the conclusions and recommendations.

Information Sources Summary

This volume updates the information available on fire's impacts on soils and water. It is just one source of information prepared to a given point in time. Barring more frequent updates of this volume (originally published in 1979; Wells and others 1979, Tiedemann and others 1979), some mechanisms for future information retrieval need to be continually dynamic and current. The World Wide Web has produced a quantum leap in the availability of information now at the fire manager's finger tips, with past and current information on the Web growing daily. The future problem will be synthesizing the mountains of information now available. It is hoped that this volume provides some of that synthesis.

Research Needs

This volume points out information gaps and research needs throughout its chapters. To complicate matters more, some regions of the country are experiencing larger and more severe fires that are producing ecosystem effects not studied before. Some of the key research needs are:

- Beyond the initial vegetation and watershed condition impacts of the physical process of fire, suppression activities can add further levels of disturbance to both soils and water. These disturbances need to be evaluated along with those produced by fire.
- The burning of concentrated fuels (such as slash, large woody debris) can cause substantial damage to the soil resource even though these long-term effects are limited to only a small proportion of the landscape where the fuels are piled. The physical, chemical, and biological effects of concentrated fuel burning need to be better understood.
- Improved understanding of the effects of fire on soil chemical properties is needed for managing fire on all ecosystems, and particularly in fire-dependent systems.
- Soil microorganisms are complex. Community members range in activity from those merely trying to survive to others responsible for biochemical reactions among the most elegant and intricate known. How they respond to fire will depend on numerous factors, including fire intensity and severity, site characteristics, and preburn community composition. Understanding of these responses is needed.
- Peakflows and floods are of great concern because of their potential impacts on human safety and property. Next to the physical destruction of a fire itself, postfire floods are the most damaging aspect of fire in the wildland environment. It is important that research focus on improving the ability of resource specialists and managers to understand the potential of fires to increase peakflows and to improve predictions of floods.
- Heavy rain on recently burned land can seriously degrade water quality. Severe erosion

and runoff are not limited to wildfire sites alone. But if postfire storms deliver large amounts of precipitation or short-duration, high-intensity rainfalls, accelerated erosion and runoff can occur even after a carefully planned prescribed fire. The effects of fire on municipal watersheds are not well documented, and the ability to predict magnitude and duration of water quality change is limited.

- Limited observational information exists on the immediate, direct effect of fire on fishes. Most information is indirect and demonstrates the effects of ash flows, changes in hydrologic regimes, and increases in suspended sediment on fishes. These impacts are marked, ranging from 70 percent to total loss of fishes. Also, the effects of fire retardants on fishes is observational and not well documented. Further, most information is from forested biomes, so research needs to be expanded to grasslands.
- Anthropogenic influences, largely land use activities over the past century, cumulatively influence fire effects on fishes. Fire suppression alone has affected vegetation densities on the landscape and the severity and extent of wildfire and, in turn, its effects on aquatic ecosystems and fishes. Most all studies of fire effects on fishes are short term (less than 5 years) and local in nature. A landscape approach to analyses and research needs to be made.
- Riparian areas are particularly important because they provide buffer strips that trap sediment and nutrients that are released when surrounding watersheds are burned. The width of these buffer strips is critical for minimizing sediment and nutrient movement into the streams. But little information exists on effective buffer strips and sizes.
- Although we have several models, all of them have limitations that must be understood by fire managers or watershed specialists before they are applied. The models are only as good as the data used to create and validate them. Some processes such as extreme flow and erosion events are not simulated well because of the lack of good data or the complexity of the processes.
- The need for improved effectiveness monitoring has gained momentum. In April 2003, the Government Accounting Office of Congress published a report (GAO 2003) that clearly stated that neither the U.S. Department of Agriculture's Forest Service nor the Department of the Interior's Bureau of Land Management could determine whether emergency stabilization and rehabilitation treatments were achieving their intended results. Although treatment monitoring is required, there is no agreed-on protocol for how and what to collect and analyze for determining effectiveness.
- Rainfall intensity as well as rainfall amount and duration affect the success of rehabilitation treatment. Rehabilitation treatments are least effective in short-duration, high-intensity rainstorms (that is, convective thunderstorms), particularly in the first 2 years after burning. Information is limited in this area.
- Quantitative data are needed to guide future responses to postfire rehabilitation and to build, test, and refine predictive models for different burned forest environments. Accurate climate, runoff, and erosion models for burned forest environments are dependent on (1) improved mapping of burn severity and better characterization of postfire soil water repellency; (2) improved prediction of runoff responses at different spatial scales, including short-duration, high-intensity thunderstorms; (3) quantitative data for the relative magnitudes and consequences of hillslope verses channel erosion; and (4) refined sediment deposition and routing models for various drainages.
- Consumption of organic soil horizons has only been quantified in a limited number of vegetation types. Although moisture vs. consumption patterns have emerged there is still a need to quantify consumptin and soil heating in a wider range of vegetation types. Likewise additional research is needed to further elucidate the physical mechanisms of organic soil consumption such that robust models based on combustion and heat transfer relationships can eventually replace the empirical studies leading to wider application.
- Salvage logging after wildfires is becoming a more common occurrence. Research in the past has examined the soil and water impacts of logging and wildfire separately, but rarely together. There is a large need for logging and its associated roading activities at various time intervals post-wildfire, and for different forest ecosystems and physiographic regions.

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Appendix A: Glossary

aerial fuels: Fuels more than 6.5 feet (2 m) above the mineral soil surface.

ammonification: Transformation of organic nitrogen (N) compounds, such as proteins and amino acids, into ammonia (NH_4). Ammonification is a process involved in the mineralization of N that is affected by fire.

ashbed effect: The accumulation of thick layers of ashy residue on the soil surface after fire resulting from combustion of concentrated fuels such as deep litter layers, piled slash, and windrows.

available fuel: Amount of fuel available for burning in a particular fire, a value varying widely in magnitude with the environmental conditions on a site.

average runoff efficiencies: The ratio of runoff to precipitation.

back fire: Fire set against an advancing fire to consume fuels and (as a consequence) prevent further fire. PAH formation increases since gaseous fuels have longer residence times in these types of combustion conditions.

backing fire: A fire that is burning against the slope or wind, i.e., backing down slope (this type of fire typically has the lowest fireline intensity but often longer flaming durations), or burning into the wind (the lee side of the fire, but again with the lowest fireline intensity on the perimeter).

baseflow: Streamflow sustained by subsurface flow and groundwater flow between precipitation events.

basidiomycetes: The phylum basidiomycota consists of fungi that produce spores that are formed outside a pedestal-like structure, the basidium. The members of this phylum, known as basidiomycetes, include all the fungi with gills or pores, including the familiar mushrooms and bracket fungi.

benefits: Favorable effects of fire-caused changes in the ecosystem.

biomass: All of the vegetative materials available for burning in natural ecosystems.

buffer strips: Vegetated bands along streams or around water.

burn area: The area over which a fire has spread.

burning: Refers to being set on fire.

Byram's intensity: The product of the available heat of combustion per unit area of ground surface and the rate of spread of the fire. It is also referred to as fireline intensity.

cellulose: Long-chain polysaccharides such as glucose, mannose, galactose, and xylose sugar derivatives with high oxidation levels that are found in both cell walls, but primarily in the secondary wall constituting the largest group of carbohydrates in wood (40 to 50 percent).

channel interception: Interception of precipitation that falls directly into a stream channel that contributes to streamflow.

char: A carbonaceous residue left on the surface of fuels by pyrolysis that is neither intact organic compound nor pure carbon.

char-height: The height of stem charring as a proportion of total tree height. Also is an indicator of postfire tree mortality in some species.

chemical energy: Solar energy fixed by plants in the synthesis of organic molecules and compounds.

chemical properties: Properties of fuels that affect the heat content and the types of pollutants emanating from a fire.

coarse woody debris: Made up of tree limbs, boles, and roots in various stages of decay, and that are greater than 3 inches (7.5) cm in diameter.

combustion: Is a rapid physical-chemical process, commonly called fire, that releases the solar energy stored in a chemical form in various fuels as heat and a variety of gaseous and particulate by-products.

combustion rate: The mass of fuel consumed by the combustion process (e.g. tons/minute, kg/sec,etc.) or the average speed (ha/min or acres/min) at which fuels are being burned.

condensation: Is the process whereby water changes from a gas to a liquid and releases heat.

controlled burning: Also called prescribed burning, is the controlled application of fire to fuel buildups, in either their natural or modified state, in specified environmental conditions that allow the fire to be confined to a predetermined area and, at the same time, to produce the fireline intensity and rate of spread required to attain the planned management objectives.

convection: Process whereby heat is transferred from one point to another by the mixing of one portion of a fluid with another fluid.

conversion burning: One vegetative community on a site is replaced by another because of fire.

crown fire: Fire that advances from top-to-top of trees or shrubs more or less independently of the surface fire.

crown fuels: Tree and shrub crowns.

cryptogamic crusts: Communities of lichens, blue-green bacteria (cyanobacteria), fungi, mosses, and algae that are found on the surface of rocks and soil in dryland regions throughout the world.

damages: Unfavorable effects of fire-caused changes in a resource system.

dead fuels: Grouped by size class as 1, 10, 100, or 1,000 hour timelag classes; size classes are often separated into sound or rotten.

decomposition: The breakdown of organic matter, results in catabolism of organic matter into smaller organic materials.

denitrification: Process of reducing nitrate (NO_3–N) to nitrogen gas (N_2) and nitrous oxide (N_2O) by biological means.

dependent crown fire: Fire in the crowns that moves only in spurts and is dependent on intense heating from the surface fire. This type of fire, which generally causes the most severe impact on a natural ecosystem, occurs mostly in coniferous forests and woodlands, and in shrublands comprised of waxy-leaved species.

downstream fire effects: Occur when hydrologic processes are altered for a long enough time that the changes can accumulate through time, when responses from a number of sites are transported to the same site, or when a transported response interacts with an on-site change at another site.

duff: The F and H layers of organic matter. Therefore, the top of the duff is where leaves, needles, and other castoff vegetation have begun to decompose, while the bottom of the duff is where decomposed organic matter is mixed with mineral soil.

ecto-mycorrhizae: One of the two types of mycorrhizae found in soil.

ecotones: Abrupt edges between adjacent vegetative types.

efficient burns: Flaming combustion with high intensity consuming most, if not all, of the available fuels. With low-intensity smoldering fires fuel consumprion is only about 50%.

endo-mycorrhizae: (arbuscular) One of the two types of mycorrhizae found in soil.

endothermic: Heat-absorbing reactions that pyrolysis and combustion start with.

evapotranspiration (ET): Evaporation from soils, plant surfaces, and water bodies, together with the water losses from transpiring plants.

excess water: Portion of total precipitation that flows off the land surface plus that which drains from the soil and, therefore, is neither consumed by ET nor leaked into deep groundwater aquifers.

exothermic: Heat-producing reaction that pyrolysis and combustion progress to.

extinction: The fifth and last phase of a fire where combustion ceases.

F layer: Fermentation layer, the accumulation of dead organic plant matter above mineral soil consisting of partly decomposed matter.

favorable effects: Effects that contribute to the attainment of fire management objectives.

field capacity: The maximum amount of water that a soil mantle retains against the force of gravity.

fine fuels: Fast-drying dead fuels, characterized by a high surface area-to-volume ratio, less than 1 cm in diameter; these fuels (grasses, leaves, needles, and so forth) ignite readily and are consumed rapidly.

fire: A manifestation of a series of chemical reactions that result in the rapid release of the heat energy stored in (living and dead) plants by photosynthesis.

fire behavior: Manner in which a fire reacts to its environment—to the fuels available for burning, climate, local weather conditions, and topography. Fire behavior changes in time, space, or both in relation to changes in these environmental components. Common terms used to describe fire behavior include smoldering, creeping, running, spotting, torching.

fire climax: A plant community maintained by periodic fire.

fire cycle: Also called the fire-return interval, is the length of time necessary for an area equal to the entire area of interest to burn—the size of the area should be specified.

fire-dependent ecosystems: Those ecosystems where fire plays a vital role in determining the composition, structure, and landscape patterns.

fire ecology: The study of relationships among fire, the environment, and living organisms.

fire effects: The physical, chemical, and biological impacts of fire on the environment and ecosystem resources.

fire frequency: Also referred to as fire occurrence, is the number of fires in a specified time and area.

fire intensity: Describes the rate at which a fire produces thermal energy. When it is based on a line (of implied depth, D) it is Byram's fireline intensity, and when it is defined as a heat per unit area it is Rothermel's intensity.

fire interval: Also referred to as fire-free interval, is the time between two successive fires in a designated area—the size of the area should be clearly specified.

fireline intensity: The product of the available heat of combustion per unit area of ground surface and the rate of spread of the fire. It is also referred to as Byram's intensity.

fire occurrence: Also referred to as fire frequency, this is the number of fires in a specified time and area.

fire regime: Largely determined by the combinations of three factors: how often fire occurs (frequency), when it occurs (season), and how fiercely it burns (intensity).

fire resistance: The ability of vegetation to survive the passage of fire.

fire-return interval: Also called the fire cycle, is the length of time necessary for an area equal to the entire area of interest to burn—the size of the area should be specified.

fire severity: Describes ecosystems responses to fire and can be used to describe the effects of fire on the soil and water system, ecosystem flora and fauna, the atmosphere, and society. It reflects the amount of energy (heat) that is released by a fire which affects resource responses. Fire severity, loosely, is a product of fire intensity and residence time and is generally considered to be light, moderate, or high.

fire triangle: Fuels available for burning, along with heat and oxygen, represent the components needed for fire to occur.

fire type: Type of vegetation that commonly follows a fire, or otherwise is dependent upon the occurrence of fire.

flame: A gas-phase phenomenon of fire.

flaming: The second phase of a fire involving combustion in which pyrolysis continues.

flammability: The relative ease with which a substance ignites and sustains combustion.

forage: Grass and grasslike plants, forbs, and half-shrubs available to, and eaten by, livestock or other herbivores.

forced convection: Occurs when external mechanical forces alter the fluid flow from its natural "free" direction and velocity.

free convection: Occurs when the fluid motion of the gases is dependent upon the differences in densities resulting from temperature differences.

free radicals: Molecules that do not have a balanced charge due to excess electrons.

fuel: Term used interchangeably with *fuel available for burning* when referring to the biomass that decomposes as a result of ignition and combustion.

fuel available for burning: Term used interchangeably with *fuel* when referring to the biomass that decomposes as a result of ignition and combustion.

fuel loading: Total dry weight of fuel per unit of surface area, a measure of the potential energy that might be released by a fire.

fuel reduction burning: Removes fuel buildups to reduce the likelihood of ignition or lessen potential damage and the resistance to control of fire when it occurs on a site.

fuel state: The moisture condition of the fuel that largely determines the amount of fuel available for burning at any given time and on which fuel classifications can also be based.

glowing: The fourth phase of a fire also involving combustion in which pyrolysis virtually ceases.

good condition watershed: Precipitation infiltrates into the soil and does not contribute excessively to erosion, since the resultant overland flow (when a pathway of flow on the watershed) does not dislodge and move soil particles. Streamflow response to precipitation is relatively slow and baseflow (when a pathway of flow) is sustained between storms.

greenhouse gas: Gas that has potential to impact the global climate by warming Earth's atmosphere.

ground fire: Fire that burns the organic material in the upper soil layer and, at times, the surface litter and low-growing plants.

ground fuels: Fuels generally defined as lying below the litter (L or O_i) layer (i.e.,fermentation (For O_e) and humus (H or O_a) layers, logs with their center axis below the surface of the F-layer, peat and muck soils.

H layer: Humus layer (O_a layer), the accumulation of dead organic plant matter above the minerals soil consisting of well-decomposed organic matter.

healthy riparian ecosystem: Maintains a dynamic equilibrium between the streamflow forces acting to produce change and the resistance of vegetative, geomorphic, and structural features.

heat tolerance: The ability of plant tissue to withstand high temperatures.

heavy fuels: Snags, logs, large branches, peat of larger diameter (>3.1 inches or 8 cm) that ignite and burn more slowly than fine fuels.

heterotrophs: Organisms that are able to derive C and energy for growth and cell synthesis by utilizing organic compounds.

high fire severity: High soil heating, or deep ground char occurs, where the duff is completely consumed and the top of the mineral soil is visibly reddish or orange on severely burned sites. Less than 20 percent of the trees exhibit no visible damage, with the remainder fire-damaged, largely by root-kill; less than 40 percent of the fire-damaged trees survive.

high severity burn: All of the organic material is removed from the soil surface and organic material below the surface is consumed or charred. More than 10 percent of the area has spots that are burned at high severity, more than 80 percent moderately severe or severely burned, and the remainder is burned at a low severity.

human-caused fire: Fire caused directly or indirectly by a person or people.

hydrograph: A graphical relationship of streamflow discharge (ft^3/sec or m^3/sec) (m^3/sec) to time.

hydrologic function: Relates to the ability of a watershed to receive and process precipitation into streamflow without ecosystem deterioration.

ignition: The initiation of self-sustaining pyrolysis and flaming combustion, marks the transition point between the mainly endothermic preignition and exothermic flaming phases.

infiltration: The process of water entering the soil.

infiltration capacity: Maximum rate at which water can enter the soil.

instream flow: The streamflow regime required to satisfy the conjunctive demands being placed on water while the water remains in the stream channel.

interception: The process in which vegetative canopies, litter accumulations, and other decomposed organic matter on the soil surface interrupt the fall of precipitation (rain or snow) to the soil surface. It plays a hydrologic role of protecting the soil surface from the energy of falling raindrops.

interflow: Also called subsurface flow, it is that part of the precipitation input that infiltrates into the soil and then flows to a stream channel in a time short enough to be part of the stormflow.

intermittent stream: A stream that flows periodically, fed by channel interception, overland flow or surface runoff, and subsurface flow.

L layer: Litter layer (O_i layer), the accumulation of dead organic plant matter above the mineral soil consisting of unaltered leaves, needles, branches, and bark.

ladder fuels: Fuels continuous between ground fuels and crown fuels, forming a ladder by which a fire can spread into tree or shrub crowns.

latent heat of vaporization: The amount of heat and energy involved in the change in physical state of water. The latent heat vaporization of water is 560 cal/g, and this same amount is released during condensation.

light severity burn: A fire that leaves the soil covered with partially charred organic material.

live fuels: Living plants grouped by category as woody or herbaceous fuels.

low fire severity: Low soil heating, or light ground char, occurs where litter is scorched, charred, or consumed. The duff is left largely intact, although it can be charred on the surface. Woody debris accumulations are partially consumed or charred. Mineral soil is not changed. At least 50 percent of the trees exhibit no visible damage, with the remainder fire-damaged by scorched crowns, shoot-kill (top kill but sprouting), or root-kill (top kill and no sprouting); over 80 percent of the fire-damaged trees survive.

low severity burn: Less than 2 percent of the area is severely burned, less than 15 percent moderately burned, and the remainder of the area burned at a low severity or unburned.

macro-nutrients: Nutrients that are needed in the largest concentrations required for plant growth such as phosphorus, nitrogen, su;fur, iron, calcium, potassium, and magnesium.

mass transport of heat: Occurs during fires by air-borne spotting and downslope rolling.

methane: The third most abundant greenhouse gas contributing to global warming.

micro-nutrients: Nutrients needed in trace amounts for plant growth such as zinc, manganese, cobalt, molybdenum, and nickel.

mineralization: This is conversion of an element from an organic form to an inorganic state as the result of microbial activity. Mineralization includes the transformation of organic N compounds (such as proteins and amino acids) into ammonia (ammonification) and, subsequently, into nitrite and nitrate (nitrification); and the conversion of organic C into carbon dioxide.

moderate fire severity: Moderate soil heating, or moderate ground char, occurs where the litter on forest sites is consumed and the duff is deeply charred or consumed, but the underlying mineral soil surface is not visibly altered. Some 20 to 50 percent of the trees exhibit no visible damage, with the remainder fire-damaged; 40 to 80 percent of the fire-damaged trees survive.

moderate severity burn: Less than 10 percent of the area is severely burned, but over 15 percent is burned moderately severe, and the remainder is burned at low severity or unburned.

mycorrhizae: The plant root zone contains these fungi that enhance nutrient uptake by plants and contribute directly to the productivity of the terrestrial ecosystem.

natural fire: Fire of natural origin—lightning, spontaneous combustion, or volcanic activity.

natural fuels: Result from natural processes and, therefore, are not generated by management practices.

net precipitation: Precipitation that reaches the soil surface, moves into the soil, forms puddles of water on the soil surface, or flows over the surface of the soil.

nitrification: Transformation of organic N compounds into nitrite (NO_2) and nitrate (NO_3). Nitrification is a process involved in the mineralization of N that is affected by fire.

Nitrobacter: Responsible for the oxidation of nitrite to nitrate is almost exclusively in natural systems.

Nitrosolobus: Genera of autotrophic bacteria are able to oxidize ammonium nitrogen (NH_4-N) to nitrite.

Nitrosomonas: Genera of autotrophic bacteria are able to oxidize NH_4-N to nitrite.

Nitrospira: Genera of autotrophic bacteria are able to oxidize NH_4-N to nitrite.

O horizon: Organic matter overlying mineral soil made up of fresh litter (O_i horizon), partially decomposed litter (O_e horizon), and completely decomposed litter (O_a horizon).

old-field successions: Successional sequences on abandoned agricultural fields.

on-site fire effects: Include impacts on vegetation, soil, and nutrient cycling.

overland flow: Also called surface runoff, this is waterflow that has not infiltrated into the mineral soil and flows off the surface to a stream channel.

packing ratio: The proportion of a fuel bed volume actually occupied by fuel. The tighter fuels are packed together, the higher the packing ratio, and the lower the combustion efficiency because air supply to fire is restricted by the fuel density.

perennial stream: Stream that flows continuously throughout the year. It is fed by groundwater or baseflow, that sustains flow between precipitation events.

physical properties: Properties of fuels that affect the manner in which a fire burns and, ultimately, the generation of energy and production of air pollutants by the fire. Physical properties of interest to a fire manager generally include the quantity (fuel loading), size and shape, compactness, and arrangement.

phytobiomass: Aboveground vegetative material available for burning in natural ecosystems—often considered to be the total fuel available to burning.

polymer: Organic compound formed of repeating structural units such as simple sugars.

poor condition watershed: Precipitation flows on the soil surface and excessive erosion occurs during precipitation events. Streamflow response to precipitation is rapid and there is little or no baseflow between storms.

potential fuel: Material that might burn during an intense fire and is generally less than the total fuel.

preignition: The first phase of a fire involving fuel heating that results in dehydration and pyrolysis.

prescribed burning: Also called controlled burning, is the controlled application of fire to fuel buildups, in either their natural or modified state, in specified environmental conditions that allow the fire to be confined to a predetermined area and, at the same time, to produce the fireline intensity and rate of spread required to attain the planned management objectives.

prescribed fire: Fire burning with prescription, resulting from planned ignition that meets management objectives.

prescribed natural fire: Fire of natural origin that is allowed to burn as long as it is accomplishing one or more management objectives.

prescription: A statement specifying the management objectives to be attained, and the air temperature, humidity, wind direction and wind speed, fuel moisture conditions, and soil moisture conditions in which a fire will be allowed to burn.

primary succession: Progression to a climax plant community that is initiated on lava flow, sand dunes, alluvial deposits, and other newly exposed sites.

protective cover: Cover important to wildlife when escape from predators becomes necessary.

pyrolysis: A chemical decomposition process brought about by heating by which the fuel is converted to gases. An endothermic reaction set off by thermal radiation or convection from an advancing fire front that drives water from the surface of a fuel, elevates fuel temperatures, and then decomposes long-chain organic molecules in plant cells into shorter ones.

radiant intensity: Rate of thermal radiation emission that is intercepted at (or near) the ground surface, or at some specified distance ahead of the flame front.

radiation: Transfer of heat from one body to another not in contact with it by electromagnetic wave motion. All bodies at temperatures above $0^{o}K$ produce radiant energy.

reaction intensity: Total heat release per unit area of fuelbed divided by the burning time. It is the time-averaged rate of heat release of the active fire front that is calculated in the field by estimating the amounts of fuels burned per second and assuming heat yields for the fuels.

rhizosphere: The root environment, provides a favorable environment for soil microorganisms and is an important site of microbial activity.

riparian ecosystems: Areas that are situated in the interfaces between terrestrial and aquatic ecosystems that can be found along open bodies of water, such as the banks of rivers and ephemeral, intermittent, and perennial streams, and around lakes, ponds, springs, bogs, and meadows.

running crown fire: Fire in the crowns that races ahead of the fire on the surface in what is called a running crown fire.

secondary succession: Progression to a climax plant community that follows disturbance such as fire.

sediment: Eroded soil that is transported from watershed surfaces to stream channels by overland flow, and then through stream systems in streamflow and therefore, is the product of erosion.

sedimentation: The process of deposition of sediment in stream channels or downstream reservoirs.

sediment yield: The amount of sediment outflow from a watershed in a stream.

slash: Concentrations of downed fuels resulting from either natural events (wind, fire, snow breakage, and so forth) or management activities (logging, road construction, and so forth).

slash disposal: Treatment of slash (by burning or otherwise) to reduce the fire hazard or meet other purposes.

smoldering: The third phase of a fire also involving combustion in which pyrolysis beings to diminish.

snags: Cavities resulting from wood decay, or holes created by other species in deteriorating or dead trees used by numerous species of mammals, reptiles, amphibians, and invertebrates.

soil erosion: The dislodgement and transport of soil particles and small aggregates of soil by the actions of water and wind.

soil mass movement: The process where cohesive masses of soil are displaced by downslope movement driven by the force of gravity of soil, rock, and debris masses. This movement might be rapid (landslides) or relatively slow (creep).

soil productivity: Reflects the capabilities of a watershed for supporting sustained plant growth and plant communities, or the natural sequences of plant communities.

soil wood: Consists of buried or partially buried woody debris.

spotting: Involves the physical removal of burning material by thermal updrafts from flaming fuels and their subsequent deposition in unignited fuels some meters or kilometers away. This is a predominate mechanism of fire spread in fast moving uncontrollable fires.

stand-replacing fire: Fire that kills all or most of the overstory trees in a forest and, in doing so, initiates secondary succession or regrowth.

stormflow: The sum of channel interception, surface flow, and subsurface flow during a precipitation or snowmelt event.

subsurface flow: Also called interflow, is that part of the precipitation input that infiltrates into the soil and then flows to a stream channel in a time short enough to be part of the stormflow.

surface erosion: Caused by the actions of falling raindrops, thin films of water flowing on the soil surface, concentrated overland flow, or the erosive power of wind.

surface fire: Fire that consumes only surface fuels such as litter, low-growing plants, and dead herbaceous plants accumulated on the surface. Surface fire can ignite snags (dead standing trees), can consume shrubs and tree seedlings, and can “torch out” an occasional densely crowned mature tree. It remains a surface fire so long as its rate of spread depends on surface fuels.

surface fuels: Fuels below the aerial fuels (< 6.5 feet or 2m) and above the ground fuels.

surface runoff: Also called overland flow, this is waterflow that has not infiltrated into the mineral soil and flows off the surface to a stream channel.

thermal conductivity: Expresses the quantity of heat transferred per unit length per unit time per degree of temperature gradient and is expressed in SI units as $W/m/^{o}K$.

thermal cover: Cover critical to wildlife for the maintenance of body heat.

thermal energy: Energy that results from changes in the molecular activity or structure of a substance.

timelag: Measure of the rate at which a fuel approaches its equilibrium moisture content after experiencing environmental changes.

tolerance: Applies to light, soil nutrients, or other physiological requirements a species can tolerate.

total fire intensity: Rate of heat output of the fire as a whole, which is a function of the rate of area burned, fuel loading, and estimated heat yield.

total fuel: The amount of biomass that could potentially burn.

turbidity: A measure of suspended fine mineral or organic matter that reduces sunlight penetration of water, and influences photosynthesis rates and water quality.

unfavorable effects: Effects that make attainment of fire management more difficult.

urban-rural interface: The line, area, or zone where structures and other human developments meet, or intermingle with, undeveloped wildland areas.

vaporization: Is the process of adding heat to water until it changes phase from a liquid to a gas.

vegetation-replacing fire: Kills all or most of the living plants (including trees, shrubs, and herbaceous plants) on a site and, as a result, initiates secondary succession or regrowth.

vegetative resources: Plant communities of value to people and when demanded, are available through the implementation of prescribed management practices.

watershed condition: A subjective term to indicate the health (status) of a watershed in terms of its hydrologic function and soil productivity.

water quality: Refers to the physical, chemical, and biological characteristics of water in reference to a particular use.

wetlands: Areas that are saturated by surface water, groundwater, or combinations of both at a frequency and duration sufficient to support a prevalence of vegetation adapted to saturated soil conditions.

wettability: Property that can influence infiltration of the soil.

wildfire: Fire that is not meeting management objectives and, therefore, requires a suppression response.

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**Rocky Mountain
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Wildland Fire in Ecosystems

Effects of Fire on Air

Abstract

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This state-of-knowledge review about the effects of fire on air quality can assist land, fire, and air resource managers with fire and smoke planning, and their efforts to explain to others the science behind fire-related program policies and practices to improve air quality. Chapter topics include air quality regulations and fire; characterization of emissions from fire; the transport, dispersion, and modeling of fire emissions; atmospheric and plume chemistry; air quality impacts of fire; social consequences of air quality impacts; and recommendations for future research.

Keywords: smoke, air quality, fire effects, smoke management, prescribed fire, wildland fire, wildfire, biomass emissions, smoke dispersion

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Wildland Fire in Ecosystems

Effects of Fire on Air

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Cover photo—Photo by Roger Ottmar. Smoke blots out the sun during the 1994 Anne Wildfire in western Montana.

Preface

In 1978, a national workshop on fire effects in Denver, Colorado, provided the impetus for the "Effects of Wildland Fire on Ecosystems" series. Recognizing that knowledge of fire was needed for land management planning, state-of-the-knowledge reviews were produced that became known as the "Rainbow Series." The series consisted of six publications, each with a different colored cover, describing the effects of fire on soil, water, air, flora, fauna, and fuels.

The Rainbow Series proved popular in providing fire effects information for professionals, students, and others. Printed supplies eventually ran out, but knowledge of fire effects continued to grow. To meet the continuing demand for summaries of fire effects knowledge, the interagency National Wildfire Coordinating Group asked Forest Service research leaders to update and revise the series. To fulfill this request, a meeting for organizing the revision was held January 4-6, 1993, in Scottsdale, Arizona. The series name was then changed to "The Rainbow Series." The five-volume series covers air, soil and water, fauna, flora and fuels, and cultural resources.

The Rainbow Series emphasizes principles and processes rather than serving as a summary of all that is known. The five volumes, taken together, provide a wealth of information and examples to advance understanding of basic concepts regarding fire effects in the United States and Canada. As conceptual background, they provide technical support to fire and resource managers for carrying out interdisciplinary planning, which is essential to managing wildlands in an ecosystem context. Planners and managers will find the series helpful in many aspects of ecosystem-based management, but they will also need to seek out and synthesize more detailed information to resolve specific management questions.

— The Authors
December 2002

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Several scientists provided significant input without requesting authorship in this volume. We acknowledge valuable contributions by Sue A. Ferguson, Timothy E. Reinhardt, Robert Yokelson, Dale Wade, and Gary Achtemeier. We also thank the following individuals for their suggestions, information, and assistance that led to substantial technical and editorial improvements in the manuscripts: Scott Goodrick, Allen R. Riebau, Sue A. Ferguson, and Patti Hirami. Finally, we appreciate Marcia Patton-Mallory and Louise Kingsbury for persistence and support.

Contents

Summary

Wildland fire is an integral part of ecosystem management and is essential in maintaining functional ecosystems, but air pollutants emitted from those fires can be harmful to human health and welfare. Because of the public and governmental concerns about the possible risk of wildland fire smoke to public health and safety, as well as nuisance, visibility, ozone generation, and regional haze impacts, increasingly effective smoke management programs and air quality policies are being implemented with support from research and land management agency programs.

This state-of-knowledge review of what is known about the effects of fire on air quality has been prepared to assist those in the fire and air quality management communities for future discussion of management, policy, and science options for managing fire and air quality. The introduction sets up a framework in which to discuss the interaction between pollutants emitted from fire, and air quality at the national, State, and local levels applied to air resource management, fire management, and geographical scale components. It also provides an overview of science reviews conducted since 1979 and discusses recent changes in fire policy, strategies, and funding. The Clean Air Act and its amendments are discussed in chapter 2, in the context of how and why fire impacts each issue, what information is needed, and who needs it to fulfill legal requirements under the act. National ambient air quality standards, regional haze and visibility, hazardous air pollutants, and best available control methods are some of the topics covered. Chapter 3 covers the magnitude of the impacts of prescribed and wildland fire on air quality, and contains an overview of smoke management plans intended to manage those impacts.

Chapters 4 through 7 present scientific and technical discussions. Chapter 4 discusses the characterization and production rate of emissions from fire in terms of fuels, fire behavior, stages of combustion, fuel consumption, and emission factors of various pollutants. The basic elements and modeling of transport and dispersion are covered in chapter 5, including, plume, puff, particle, and grid models. Chapter 6 considers plume and atmospheric chemistry, the chemical reactions that occur in plumes, with a focus on ozone formation and particle formation. Use of emission inventories, air quality monitoring, and source apportionment methods, and mechanistic models to estimate the impacts of fire on air quality are covered in chapter 7. Chapter 8 reviews the health, welfare, economic, and safety consequences of these impacts. The final chapter recommends priorities for future research to better understand and quantify fire and its effect on air quality.

Metric Equivalents

When you know:	Divide by:	To find:
Feet (ft)	3.28	Meters
Pounds (lb)	2.21	Kilograms
Acres	2.47	Hectares
Pounds per acre	0.89	Kilograms per hectare
Fahrenheit (°F)	1.8 and subtract 32	Celsius

Chapter 1: Introduction

A state-of-knowledge review, *Effects of Fire on Air*, was written in 1979 to inform environmental agencies, fire managers, and land management planners, and to guide research strategies in the intervening years (Sandberg and others 1979). That review is still technically sound for the most part, but substantial new knowledge is now available. In this volume, we update that review of knowledge important for managing the effects of fire on air and for adjusting the course of new research. In addition, we expand the scope of our review to place the information in the context of new policies regarding fire management and air quality management

Acquisition of scientific knowledge regarding air pollution from fires is motivated by active policy development both to restore the role of fire in ecosystems and to improve air quality. Land managers require quantitative analysis and goal-seeking solutions to minimize the negative consequences of fire management. Managing fire and air quality to the standards set by Congress requires an increasingly detailed base of scientific knowledge and information systems.

The *Federal Wildland Fire Policy* (U.S. Department of the Interior and U.S. Department of Agriculture 1995) and the *Clean Air Act as Amended 1990* (PL 101-549) resulted in the need to significantly raise the level of knowledge about fire's effects on air in order to meet regulatory and management requirements. For example, new information is needed to assess, monitor, predict, and manage:

- Emissions and air quality impacts from wildfires
- Acute health effects of human exposure to smoke
- Natural and anthropogenic sources of visibility reduction
- Cumulative air quality impacts from expanded fuel management programs
- Tradeoffs between air quality impacts from wildland fire and prescribed fire

Likewise, management of fire and air quality is also undergoing substantial policy development that has led to the need for new and different information to satisfy regulatory and management requirements. As both legal and management issues mature, there is less a sense that environmental regulation is a limitation on fire management, and more of a sense that ecosystem management goals, fire safety, and air quality are goals to be met collectively. For example, new air quality rules recognize the importance of the role of fire in sustaining ecosystems and the inherent tradeoffs between prescribed fire and wildland fire occurrence. At the same time, land management plans and real-time fire management decisions increasingly factor in the expected consequences to air quality.

Since 1995, researchers and land managers have concentrated a great deal of energy to extend what is known about fire and its effect on air quality; to expand information systems that make knowledge readily available to policy, management, and public clients; to merge what is known about sustainable ecosystems and disturbance ecology with what is known about the chemistry, physics, biology, and social impacts of air pollution; and to redefine the research agenda.

Objective

This review summarizes the current state of knowledge of the effects of fire on air, and defines research questions of high priority for the management of smoke from fires. We also intend this as a reference document for future discussion of management, policy, and science options for managing fires and air quality. This review is limited to readily available published and unpublished knowledge and to original contributions by the authors. No new analysis of data or policy, nor assessment of impacts and options, is included herein.

Related Publications

This document does not stand alone. There are several excellent sources for information on the effects of fire on air. We advise the reader to include at least the following publications, each of which will be abstracted in this document, in your reference library:

- *Smoke Management Guide for Prescribed and Wildland Fire: 2001 Edition* (Hardy and others 2001)
- *National Strategic Plan: Modeling and Data Systems for Wildland Fire and Air Quality* (Sandberg and others 1999)
- *Introduction to Visibility* (Malm 2000)
- *Fire Effects on Air* (Sandberg and others 1979)
- *Southern Forestry Smoke Management Guidebook* (Southern Forest Fire Laboratory Personnel 1976)
- *Development of Emissions Inventory Methods for Wildland Fire* (Battye and Battye 2002)

Why, then, is another state-of-knowledge review necessary on the subject of fire effects on air? First, because policy and regulatory development in air quality management and in fire management is advancing rapidly, and there is a continuing need to reassess current knowledge about what is required to meet new expectations. Second, this document addresses the advancement of science at a much higher level than the above-mentioned references. Third, because the Joint Fire Science Program has sponsored a series of reviews, nicknamed the Rainbow Series (see "Preface"), to compile a broad reference of fire effects to serve practitioners and policymakers charged with using and managing fire, and this is the third volume in that series. Finally, we hope you will find this volume a useful attempt to abstract and fill in the gaps left by the previous publications.

Scope

This review includes all health and welfare effects of air pollution from fires, but does not include the effects of air resource management on ecosystem health or any other value. Unless otherwise specifically stated, the term "fires" in this manuscript includes all prescribed and wildland fires on wildlands. Prescribed fires are ignited intentionally to achieve ecosystem management or fire protection objectives, whereas wildland fires result from unplanned ignitions on wildlands. Wildlands include all the nonagricultural and nonresidential rural lands of the United States, including the wildland-urban interface, regardless of ownership, sovereignty, or management objective. Management response to wildland fires differs greatly according to economic efficiency, the values at risk (including air quality), and the expected ecological consequences. Wildfires are at one end of the spectrum of wildland fires in that they are unwanted and unplanned, and are managed to minimize cost plus loss. At the other end of the spectrum are wildland fires that benefit ecosystem values, and are managed to maximize their benefit. Ideally, each wildland fire is evaluated with respect to expected costs, losses, risks, and benefits in order to provide an appropriate and preplanned response. Because fires are a significant emitter of air pollutants, many other fire management activities such as fire prevention or fuel treatment may have an indirect effect on air quality.

Framework

The issues, responsibilities, and tools that address fire and air quality are varied and complex, sometimes resulting in confusion about the physical scale and temporal stage of three characteristics: the application to fire management, the application to air resource management, and the physical process of air pollution. *National Strategic Plan: Modeling and Data Systems for Wildland Fire and Air Quality* (Sandberg and others 1999) provides a conceptual framework for visualizing fire's effects on air by representing the scope of the problem as a three dimensional array of air resource management, fire management, and scale components (fig. 1-1). The air resource component is ordered in time from emissions source strength, to ambient air quality, and to effects. The fire management component includes planning, operations, and

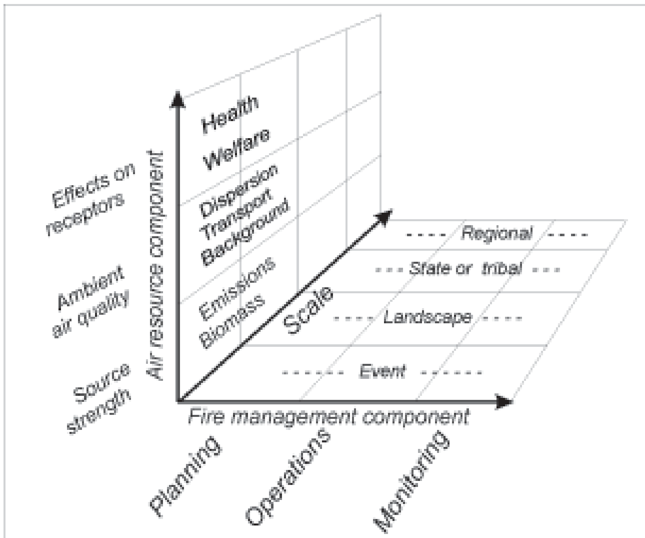

Figure 1-1—Three primary components of the issues, responsibilities, and tools related to wildland fire and air quality: air resource management, fire management, and scale (Sandberg and others 1999).

monitoring. The scale component includes the event, landscape, state or tribal, and regional scales.

We have organized this volume around the air resource component and expanded it to include a regulatory perspective (fig. 1-2). Fire in the context of the regulatory environment is the subject of chapters 2 and 3. Biomass consumption and emissions are the subject of chapter 4; transport and dispersion of pollutants in the atmosphere the subjects of chapters 5 and 6; air quality impacts the subject of chapter 7; and the effect on human values from exposure to air pollutants the subject of chapter 8. We conclude with a review of recommendations for future research in chapter 9.

Prior Work

Since the publication of *Effects of Fire on Air* (Sandberg and others 1979), significant changes have come to pass in both the technical and policy issues that surround the fire and air quality dilemma. The conferences, stakeholder group discussions, and technical publications discussed here have helped to shape the current fire management programs and will influence future programs.

Smoke Management Guide For Prescribed and Wildland Fire: 2001 Edition

Smoke Management Guide for Prescribed and Wildland Fire: 2001 Edition (Hardy and others 2001) has been developed by the Fire Use Working Team of the National Wildfire Coordinating Group (NWCG) and involves most of the same authors as this current publication. The guide provides fire management and smoke management practitioners with a fundamental understanding of fire emissions processes and impacts, regulatory objectives, and tools for the management of smoke from fires. It is a comprehen-

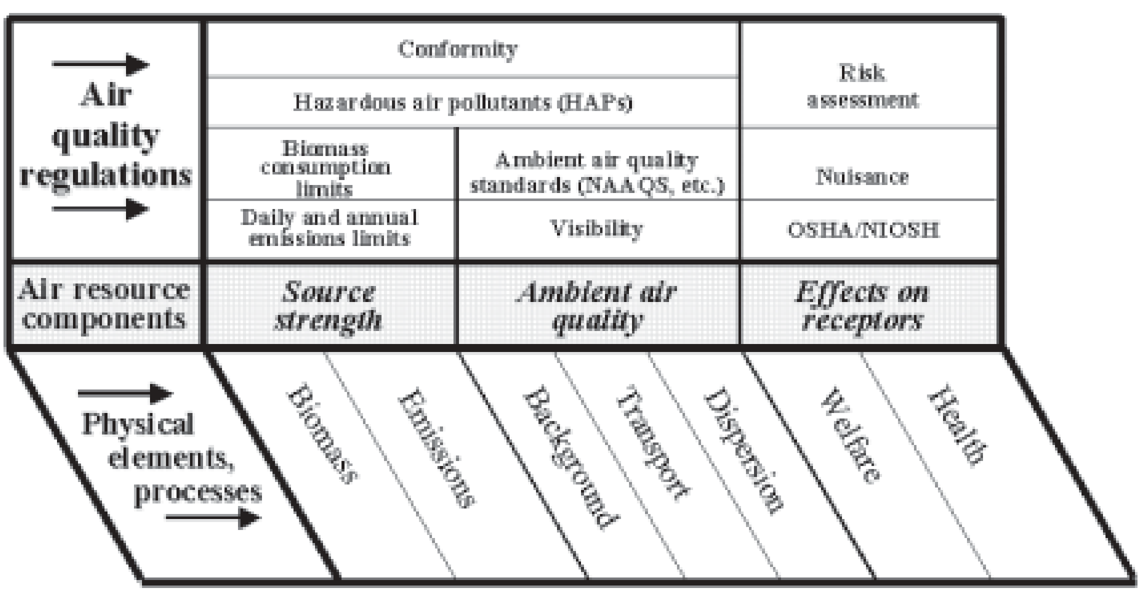

Figure 1-2—The relations of air regulations and physical processes to the three categories within the air resource component. OSHA/NIOSH = Occupational Safety and Health Administration/National Institute for Occupational Safety and Health (Sandberg and others 1999).

sive treatment of the state of knowledge regarding fire and air quality and provides guidance to practitioners. We will not attempt to duplicate its level of detail in this volume. Rather, we add some technical background and analysis of research needs relative to new requirements for management.

First published in 1985, the guide is intended to provide national guidance for the planning and managing of smoke from prescribed fires to achieve air quality requirements through better smoke management practices (NWCG 1985). This guide has been widely distributed within the fire community and air quality regulatory agencies, and to private and Tribal land managers, providing a single comprehensive source of information on fire and air quality issues.

Much has changed since 1985 in prescribed burning practices, smoke management programs, and air quality regulatory requirements. These changes are reflected in the 2001 edition of the guide, which includes expanded sections on fire and emissions processes, smoke impacts on health, welfare, safety, and nuisance; regulations for smoke management; and the fundamentals of responsible smoke management (Hardy and others 2001). These fundamentals include fire planning, use of smoke management meteorology, techniques to reduce emissions, smoke dispersion prediction systems, air quality monitoring methods, and program assessment.

The most significant change in the guide is the expanded and updated section on techniques to reduce emissions and impacts. While the 1985 guide focused primarily on minimizing smoke impacts by meteorological scheduling and dispersion, the 2001 guide provides detailed information on emissions reduction techniques, used in different regions of the country, that have been useful, practicable, and effective in the field. This emphasis on actual reduction of emissions rather than dispersion was provided in response to air quality regulations that now target regional emissions reductions.

Readers will also find that the 2001 guide has a great deal more information on the latest developments in national air quality regulations that affect fire programs including the regional haze and visibility protection programs, Clean Air Act's conformity requirements, EPA's *Interim Air Quality Policy on Wildland and Prescribed Fires* (EPA 1998), and NEPA planning guidance. The guide was drafted by 16 authors and five editor/compilers working under the sponsorship of the NWCG Fire Use Working Team with support from the EPA.

Wildland Fire and Air Quality: National Strategic Plan

Another recent publication also provides a systematic review of the state of knowledge and information systems. This strategic plan was also sponsored by the NWCG and the Environmental Protection Agency (EPA).

In 1997, the NWCG Fire Use Working Team sanctioned a small group of fire research scientists and air quality managers to develop a *National Strategic Plan: Modeling and Data Systems for Wildland Fire and Air Quality* (Sandberg and others 1999) to foster development and implementation of models and data systems that could be used to manage air quality impacts of fires. The resulting report provides a conceptual design and strategic direction toward meeting the increasing need for information required to manage emissions from fire (Sandberg and others 1999). In November 1997, after 2 years of drafting and extensive review of a draft plan, 86 experts attended a national workshop, and using the discussion framework presented in this chapter, they defined the current state of knowledge, desired future condition, and recommendations for research and development for each cell in the discussion framework.

The strategic plan targets a more technical, scientific, and policy-oriented audience than the smoke management guide, and recommends a research and development strategy to reach a desired future state for smoke management information systems. It also provides a comprehensive treatment of policy and technical issues that we will not duplicate in this volume.

Introduction to Visibility

Air pollution impacts on visibility are discussed in detail in *Introduction to Visibility* (Malm 2000). The discussion is not specific to the impacts of fire but is relevant because of the regulatory attention given to fire in the EPA Regional Haze Rule (40 CFR Part 51 1999) and because Federal land managers have the responsibility of managing fires and the impacts of fires and all other pollution sources on visibility in many National Parks and wilderness areas. We make no attempt in this volume to duplicate this discussion of the atmospheric physics, meteorology, historic visibility trends, monitoring and apportionment methodologies, or human perceptions that are so admirably covered in *Introduction to Visibility*.

The Federal Advisory Committee Act White Papers

During the 1997 to 1998 development of proposed national ambient air quality standards (NAAQS) for PM2.5 (particulate matter with an aerodynamic diameter less than or equal to 2.5 microns) and regional haze regulations, EPA used provisions of the Federal Advisory Committee Act (FACA) to convene a large group of stakeholders who were interested in providing input to the regulatory process. A FACA committee for the development of ozone, particulate matter, and regional haze implementation programs was formed to address both policy and technical issues. The committee's Science and Technology Wildland Fire Issues Group, one of several working groups and subcommittees, researched and drafted five reports that are briefly summarized below (EPA 2000c).

Air Monitoring for Wildland Fire Operations provides recommendations for conducting air-monitoring programs designed to support fire activities that also monitor for compliance with NAAQS. It also describes how monitoring can support burning programs and how land managers can collaborate with air agencies, and it provides guidance for selecting monitoring equipment.

Elements of a Smoke Management Program discusses recommendations for a basic level smoke management program. The document summarized information from an EPA-sponsored workshop held to respond to specific questions posed by EPA. The document describes the six basic components of a smoke management program:

- Authorization to burn
- Minimizing emissions
- Burn plan components
- Public education
- Surveillance and enforcement
- Program evaluation

It also provides examples of monitoring methods, public awareness programs, and program enforcement.

Emission Inventories for State Implementation Plan [SIP] Development describes several levels of inventory complexity: a default level based on currently available information; a basic level program that is considered the minimal program needed to support SIP development; and a detailed inventory level when a greater level of analysis or accountability in inventory precision is needed. Elements of each level of inventory are described, data sources are identified and data management issues are discussed.

What Wildland Fire Conditions Minimize Emissions and Hazardous Air Pollutants and Can Land Management Goals Still be Met? This paper is a discussion of fire conditions and techniques that minimize pollutant emissions. Both wildland emissions and prescribed fire emissions are discussed. The discussion of emissions reduction techniques for prescribed burning is also found in *Smoke Management Guide for Prescribed and Wildland Fire: 2001 edition* (Hardy and others 2001).

Estimating Natural Emissions from Wildland and Prescribed Fire addresses how best to define "natural emissions" from fire. This is critical to implementing regional haze goals of reducing visibility degradation caused by human-made sources of air pollution. The paper discusses a matrix of choices: (1) emissions from fire necessary to restore and sustain desired ecosystem characteristics, (2) fire needed to manage fuels to a condition where they can be dealt with most effectively from a wildfire control standpoint, (3) no net increase in fire emissions, and (4) no change from current emissions.

Stakeholders reviewed, discussed, and drafted, additional work on these five reports. The reports and other technical references were considered by EPA during the formulation of the regional haze regulations and revisions to the particulate matter NAAQS.

Environmental Regulation and Prescribed Fire Conference

In March 1995 a conference on new developments in environmental regulations related to prescribed fire was held in Tampa, FL (*Conference Proceedings: Environmental Regulation & Prescribed Fire: Legal and Social Challenges*, Bryan 1997). This 3-day meeting included sessions on challenges and strategies regarding the use of fire, air quality regulation, and liability, as well as social and economic issues. Sponsored by numerous State and Federal environmental and forestry agencies, the conference provided a forum for discussion of the Clean Air Act, Endangered Species Act, and other Federal statutes that guide national, State, and local regulations pertaining to prescribed fire.

Significantly, a joint declaration drafted by the conference steering committee and presented to conference attendees was later signed by representatives of the EPA, State of Florida, National Biological Survey, The Wilderness Society, Forest Service, and Mariposa County, Florida. In summary, the declaration upheld the following principles:

- Practitioner liability is a major obstacle to the increased use of fire. Legislation should be considered on the Federal level to protect properly certified fire practitioners except in cases where negligence is proven.
- Partnerships among all of the stakeholders are vital to the future use of fire. Efforts to enhance such partnerships must be encouraged especially in the exchange of information, development of best management practices, public education campaigns, and funding initiatives.
- Agencies should work together to evaluate tradeoffs between public health risks from fire and ecological damage caused by fire exclusion.

- Public education regarding the use of prescribed fire, ecosystem health, and risks of wildfire versus those from prescribed burning is encouraged.
- The role of fire in ecosystem management needs to be understood by all stakeholders. The ramifications of not using prescribed fire are serious and must also be appreciated as limits on fire use may conflict with other public mandates.
- Actions pertaining to the use of fire must be based on sound science. There are several crucial knowledge gaps that must be filled. Consequences to public safety caused by delaying the increases of prescribed fire are great.
- Public and private property owners need to retain the right to use prescribed fire to protect and enhance the productivity of their lands while also protecting nearby property owners from adverse impacts of burning.
- Administrators responsible for allocating funds should do so on the basis of regional priorities with greater emphasis on prevention than in the past.
- An increased emphasis on training for prescribed fire practitioners is needed to enhance public acceptance.

Southern Forestry Smoke Management Guidebook

The *Southern Forestry Smoke Management Guidebook* (Southern Forest Fire Laboratory Personnel 1976) was one of the first smoke management guidebooks developed in the United States for use by land, fire, and air resources managers. The guide provides an improved understanding of: (1) smoke management and air quality regulations; (2) contents of smoke and variables affecting production; (3) smoke transport and dispersion; (4) potential effects on human health, human welfare, and visibility; and (5) what can be done to mitigate its impacts. A system for predicting and modifying smoke concentrations from prescribed fires was introduced for Southern fuels.

Changes in Fire Policy

The *Federal Wildland Fire Policy* (USDI and USDA 1995; USDI and others 2001) requires that "… fire, as a critical natural process, must be reintroduced into the ecosystem to restore and maintain sustainable ecosystems. This will be accomplished across agency boundaries and will be based on the best available science." The policy requires "the use of fire to sustain ecosystem health based on sound scientific principles and balanced with other social goals including public health and safety, air quality, and other specific environmental concerns." Early in the planning process, action is required to "involve public health and environmental regulators in developing the most workable application of policies and regulations." Agencies are called on to "create a system for coordination and cooperation among land managers and regulators that explores options within existing laws to allow for the use of fire to achieve goals of ecosystem health while protecting individual components of the environment, human health, and safety." The policy also requires that air quality values be considered during preparedness and fire protection. When setting protection priorities, land managers must "define values to be protected working in cooperation with state, local, and tribal governments, permittees, and public users. Criteria will include environmental, commodity, social, economic, political, public health, and other values."

Several strategies and funding programs were developed to improve the ability of managers to fully implement this policy.

Joint Fire Science Program

The Joint Fire Science Program (JFSP) was created by Congress in the 1998 Appropriations to Interior and Related Agencies bill to augment the delivery of science and information systems necessary to manage the increased use of fire and other fuel treatments. The legislation provides a mandate to protect air quality in conjunction with economic efficiency and ecological consequences. The program (National Interagency Fire Center 2002 unpaginated) recognizes that:

> Land managers are rapidly expanding the use of fire for managing ecosystems while air resource managers are accelerating efforts to reduce the local and regional impacts of smoke. Smoke management (meeting air quality standards) is a legal requirement of the Clean Air Act, as well as a health and safety issue for the general populace and fireline personnel. The JFSP will attempt to define these social relationships and develop analytical tools and communication practices to help mangers include social considerations in decision making.

One of the goals of the JFSP is "to evaluate various treatment techniques for cost effectiveness, ecological consequences, and air quality impacts." The program plan states:

> Methods have not been developed to assess the opportunities, costs, and effectiveness of employing smoke reduction techniques throughout the country. Current models to assess regional scale cumulative effects on air quality and water quality will need to be expanded. The program will develop a nationally consistent system of models for fuel consumption, emissions production, and smoke dispersal that can assess cumulative

> effect. This research would also contribute to understanding the potential national and global impacts of changes in biomass use, prescribed fire, and wildland fire on wood supply, atmospheric chemistry, and carbon sequestration.

Cohesive Strategy

Protecting People and Sustaining Resources in Fire-Adapted Ecosystems: A Cohesive Strategy (Laverty and Williams 2000) is the Federal framework established to restore and maintain ecosystem health to reduce the threat and consequences of wildfires. It is presumed that fire suppression over the past 100 years has excluded fire from many ecosystems, fueling conditions for unnaturally intense fires that, among other effects, threaten air quality. Citing serious air quality impacts from long duration wildfire episodes in recent years, the report expresses concern that:

> The extent to which management for ecosystem resilience can improve air quality over the long term is not completely known. Present regulatory policies measure prescribed fire emissions, but not wildland fire emissions. The emissions policy tends to constrain treatments and – in short interval fire systems — may act to inadvertently compound wildland fire risks. (p 34)

The cohesive strategy directs land management agencies to collaborate with the EPA in addressing long-term impacts, tradeoffs, and issues regarding air quality and other impacts. The report acknowledges that programmatic analysis of air quality impacts will be a necessary step in implementing the planned increases in prescribed burning necessary to restore the health of fire-prone ecosystems. The strategy estimates that the USDA Forest Service Regions would increase fuel treatments by five-fold in the West and two-fold in the East and South to achieve restoration goals within 10 years; or employ a slightly smaller increase to obtain results in 20 years. Most, but not all, of the treatments would involve burning.

The relative risk to air quality was projected to decrease by about 25 percent as a result of improving the resilience of ecosystems, according to current models.

The cohesive strategy is responsive to regulatory responsibilities. The planned increase in burning is constrained in part by the consideration to regulatory obligations, with an acknowledgment that a more rigorous assessment of impacts could substantially change the planned extent and schedule of treatments. Concerns for public health issues and firefighter safety in relation to smoke are also expressed. The strategy acknowledges that air quality issues must be analyzed more thoroughly at smaller scales as it is stepped down to landscape and project level planning.

National Fire Plan

The National Fire Plan was established in *A Report to the President In Response to the Wildfires of 2000* (USDA and USDI 2000), and implemented using *Collaborative Approach for Reducing Wildfire Risks to Communities and the Environment: 10-Year Comprehensive Strategy* (Western Governors' Association 2001). Stakeholder groups under the sponsorship of the USDA Forest Service, USDI, and the Western Governors' Association prepared the implementation strategy. This strategy recognizes that key decisions in setting priorities for restoration, fire, and fuel management should be made at local levels. As such, the plan requires an ongoing process whereby the local, Tribal, State and Federal land management, scientific, and regulatory agencies exchange the required technical information, including the assessment of air quality tradeoffs, to inform this decisionmaking process. The strategy has a goal of maintaining and enhancing community health and economic and social well-being; and requires that public health risks from smoke are reduced, airshed visibility is improved, and smoke management plans are developed in conjunction with prescribed fire planning and implementation.

Notes

Chapter 2: Air Quality Regulations and Fire

This chapter introduces the regulatory environment for smoke from prescribed and wildland fire, providing updated discussion of the laws, regulations, standards, and regulatory strategies that have changed since about 1980. We explain roles and responsibilities of the regulatory agencies and land managers, and we frame the technical discussion in the context of who needs what information to fulfill legal requirements.

Air pollution is the presence in the atmosphere of one or more contaminants of a nature, concentration, and duration to be hazardous to human health or welfare (Sandberg and others 1999). Welfare includes potential to harm animal or ecosystem health, economic activity, or the comfortable enjoyment of life and property. Air pollution is created from both human (that is, anthropogenic) and natural sources. Anthropogenic air pollution is the presence in the atmosphere of a substance or substances added directly or indirectly by a human act, in such amounts as to adversely affect humans, animals, vegetation, or materials (Williamson 1973). Air pollutants are classified into two major categories: primary and secondary. Air pollutant emissions, or simply "emissions," are the production and release of air contaminants emitted from fires that have a potential to cause air pollution. This definition includes particulates, hydrocarbons, carbon monoxide (CO), metals, and all other trace gases that may be hazardous or that are chemical precursors to secondary air pollution. Primary pollutants are those directly emitted into the air. Under certain conditions, primary pollutants undergo chemical reactions within the atmosphere and produce new substances known as secondary pollutants. Hazardous air pollutants are a special class of air pollutants identified in the Clean Air Act Amendments of 1990 as constituting a hazard to human health.

Air quality is a measure of the presence of air pollution. Ambient air quality is defined by the Clean Air Act of 1963 as the air quality anywhere people have access, outside of industrial site boundaries. Ambient air quality standards are standards of air quality designed to protect human health or welfare. Air resource management includes any activity to anticipate, regulate, or monitor air pollution, air pollutant emissions, ambient air quality, or the effects of air pollution resulting from fires or fire management.

In the past, emissions from prescribed fire were considered human-caused, and wildland fires were considered natural sources of emissions. But recent policy debate has focused on what should be considered natural; that is, to be reasonably unaffected by human influence. This debate resulted from the paradox that not all wildland fires are vigorously suppressed and that some prescribed burning is done to maintain healthy natural ecosystems where fire has previously been excluded.

Air resource management includes any activity to anticipate, regulate, or monitor air pollution, air pollutant emissions, ambient air quality, or the effects of air pollution resulting from fires or fire management.

Emissions and impacts on air quality from fires are managed and regulated through a complex web of interrelated laws and regulations. The primary legal basis for air quality regulation across the nation is the Federal Clean Air Act (CAA), which is actually a series of acts, amendments, and regulations that include:

- Federal Air Pollution Control Act of 1955 (PL 84-159). Provides for research and technical assistance and authorizes the Secretary of Health, Education, and Welfare to work toward a better understanding of the causes and effects of air pollution.
- Federal Clean Air Act of 1963 (PL 88-206). Empowers the Secretary of Health, Education, and Welfare to define air quality criteria based on scientific studies. Provides grants to state and local air pollution control agencies.
- Federal Air Quality Act of 1967 (PL 90-148). Establishes a framework for defining "air quality control regions" based on meteorological and topographical factors of air pollution.
- Federal Clean Air Act Amendments of 1970 (PL 91-604). Principal source of statutory authority for controlling air pollution. Establishes basic U.S. program for controlling air pollution.
- Environmental Protection Agency (EPA) promulgates national ambient air quality standards (NAAQS) for particulates, photochemical oxidants (including ozone), hydrocarbons, carbon monoxide, nitrogen dioxide, and sulfur dioxide (1971).
- Clean Air Act Amendments of 1977 (PL 95-95). Sets the goal for visibility protection and improvement in Class I areas and assigns Federal land managers the affirmative responsibility to protect air quality related values.
- Clean Air Act Amendments of 1990 (PL 101-549). Establishes authority for regulating regional haze and acknowledges the complexity of the relation between prescribed and wildland fires.
- Regional Haze Regulations, Final Rule (40 CFR Part 51) (1999). EPA promulgates the Regional Haze Rule supported in part by the 1998 Interim Air Quality Policy on Wildland and Prescribed Fires.

Roles and Responsibilities Under the Clean Air Act

States have the lead in carrying out provisions of the Clean Air Act because appropriate and effective design of pollution control programs requires an understanding of local industries, geography, transportation, meteorology, urban and industrial development patterns, and priorities. The EPA has the task of setting air quality standards (national ambient air quality standards, or NAAQS). In addition, EPA develops policy and technical guidance describing how various Clean Air Act programs should function and what they should accomplish. States develop State implementation plans (SIPs) that define and describe customized programs they will implement to meet requirements of the Clean Air Act. Tribal lands are legally equivalent to State lands, and Tribes prepare Tribal implementation plans (TIPs) to describe how they will implement the Clean Air Act. Individual States and Tribes can require more stringent air quality standards but cannot weaken clean air goals set by EPA.

Federal land managers have the complex role of managing a fire as a source of air pollutants, while fulfilling monitoring and regulatory responsibilities tied to visibility and regional haze. Federal land managers are given the responsibility by the Clean Air Act for reviewing prevention of significant deterioration (PSD) permits (discussed later in this chapter) of major new and modified stationary pollution sources and commenting to the State on whether there is concern for visibility impacts (or other resource values) in Class I areas downwind of the proposed pollution source. Some States require modeling of source impacts on Class I areas, and Federal land managers customarily comment on the model results.

The 1990 Clean Air Act Amendments require planned Federal actions to conform to SIPs. This "general conformity rule" prohibits Federal agencies from taking any action within a nonattainment or maintenance area that (1) causes or contributes to a new violation of air quality standards, (2) increases the frequency or severity of an existing violation, or (3) delays the timely attainment of a standard as defined in the applicable SIP or area plan. The general conformity rule covers direct and indirect emissions of criteria pollutants, or their precursors, which are caused by a Federal action, are reasonably foreseeable, and can practicably be controlled by the Federal agency through its continuing program responsibility.

National Ambient Air Quality Standards

The purpose of the Clean Air Act is to protect humans against negative health or welfare effects from air pollution. National ambient air quality standards (NAAQS) are defined in the Clean Air Act as amounts of pollutant above which detrimental effects to public health or welfare may result. NAAQS have been established for the following criteria pollutants: particulate matter (PM10 and PM2.5; NAAQS for particulate matter are established for two aerodynamic diameter classes: PM10 is particulate matter less than 10 microns in diameter, and PM2.5 is less than 2.5 microns in diameter; total suspended particulate matter is called PM or sometimes TSP), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), ozone, carbon monoxide (CO) and lead (Pb) (table 2-1). Primary NAAQS are set at levels to protect human health; secondary NAAQS are to protect human welfare effects including visibility as well as plant and materials damage.

An area that is found to be in violation of a primary NAAQS is labeled a nonattainment area (fig. 2-1); an area once in nonattainment but recently meeting NAAQS, and with appropriate planning documents approved by EPA, is a maintenance area; all other areas are attainment or unclassified (due to lack of monitoring). State air quality agencies can provide up-to-date locations of local nonattainment areas (PM2.5 is a newly regulated pollutant, so attainment/nonattainment status had not been determined at the time of publication of this document; monitoring must take place for at least 3 years before designation can be made, which means PM2.5 status will likely not be known until at least 2003). States are required through their SIPs to define programs for implementation, maintenance, and enforcement of the NAAQS within their boundaries. Wildland fire in and near nonattainment areas will be scrutinized to a greater degree than in attainment areas and may be subject to general conformity rules. Extra planning, documentation, and careful scheduling of prescribed fires will likely be required to minimize smoke effects in the nonattainment area to the greatest extent possible. In some cases, the use of fire may not be possible if significant impacts to a nonattainment area are likely.

The major pollutant of concern in smoke from fire is fine particulate matter, both PM10 and PM2.5. Studies indicate that 90 percent of all smoke particles emitted during wildland burning are PM10, and 90 percent of PM10 is PM2.5 (Ward and Hardy 1991). The most recent human health studies on the effects of particulate matter indicate that fine particles, especially PM2.5, are largely responsible for health effects including mortality, exacerbation of chronic disease, and increased hospital admissions (Dockery and others 1993; Schwartz and others 1996).

Prevention of Significant Deterioration

Another provision of the Clean Air Act with some applicability to wildland burning activities is the prevention of significant deterioration (PSD) provisions.

Table 2-1—National ambient air quality standards (NAAQS) (U.S. Environmental Protection Agency 2000b). Primary NAAQS are set at levels to protect human health; secondary NAAQS are to protect human welfare.

Pollutant	Averaging time	Primary	Secondary
PM10	Annual arithmetic mean	50 $\mu g/m^3$ [a]	50 $\mu g/m^3$
	24-hour average	150 $\mu g/m^3$	150 $\mu g/m^3$
PM2.5	Annual arithmetic mean	15 $\mu g/m^3$	15 $\mu g/m^3$
	24-hour average	65 $\mu g/m^3$	65 $\mu g/m^3$
Sulfur dioxide (SO_2)	Annual average	0.03 ppm[b]	—
	24-hour average	0.14 ppm	—
	3-hour average	—	0.50 ppm
Carbon monoxide (CO)	8-hour average	9 ppm	—
	1-hour average	35 ppm	—
Ozone (O_3)	8-hour average	0.12 ppm	0.12 ppm
	1-hour average	0.08 ppm	0.08 ppm
Nitrogen dioxide (NO_2)	Annual average	0.053 ppm	0.053 ppm
Lead (Pb)	Quarterly average	1.5 $\mu g/m^3$	1.5 $\mu g/m^3$

[a] $\mu g/m^3$ = micrograms per cubic meter.
[b] ppm = parts per million.

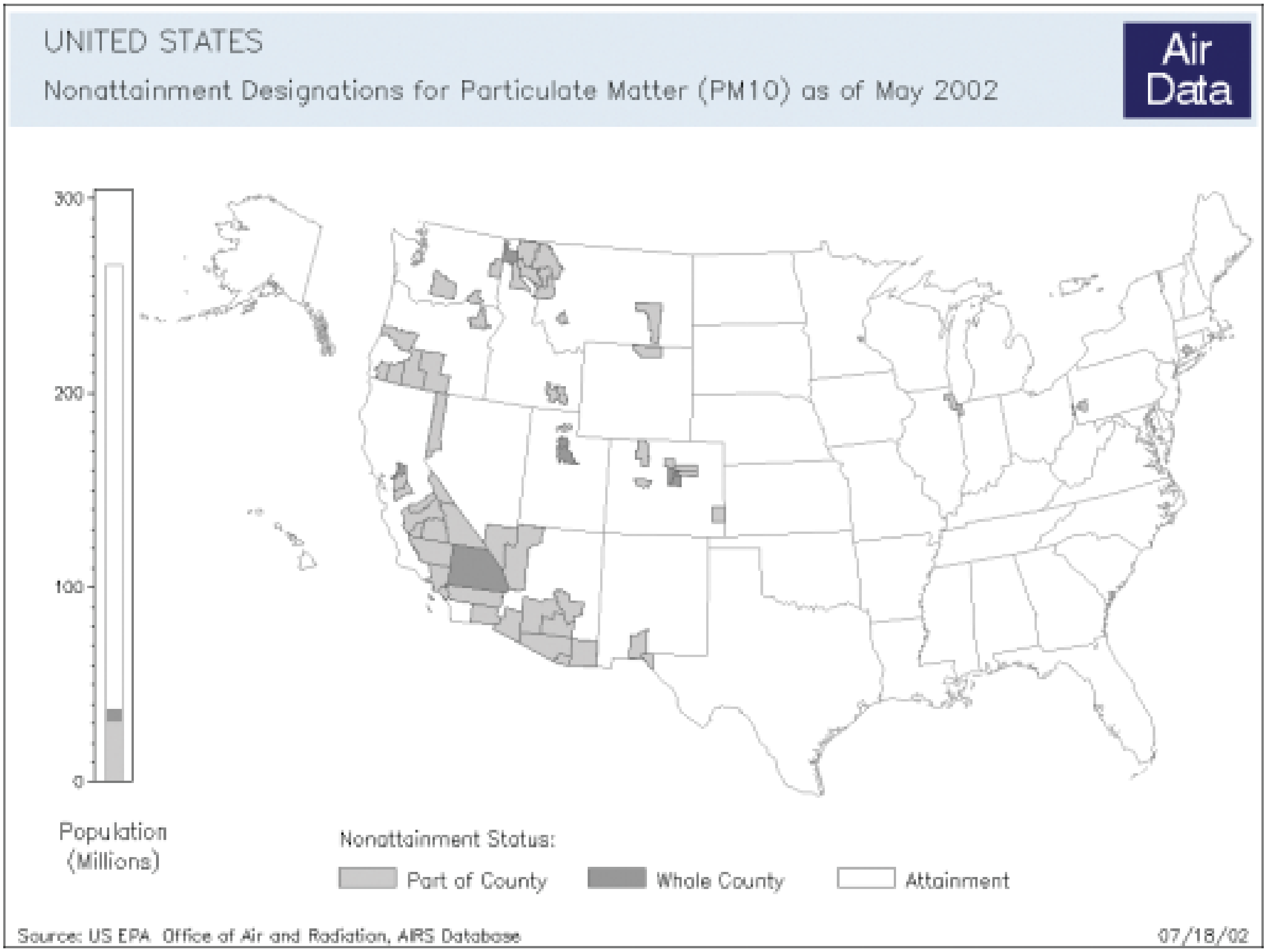

Figure 2-1—PM10 nonattainment areas as of May 2002. Current nonattainment status for PM10 and all other criteria pollutants are available from the Environmental Protection Agency (EPA) aerometric information retrieval system (AIRS) Web page at http://www.epa.gov/air/data/index.html (EPA 2002).

The goal of PSD is to prevent areas that are currently cleaner than is allowed by the NAAQS from being polluted up to the maximum ceiling established by the NAAQS. Three air quality classes were established by the Clean Air Act PSD provisions including Class I (which allows very little additional pollution), Class II (which allows some incremental increase in pollution), and Class III (which allows pollution to increase up to the NAAQS). Class I areas include wildernesses and national memorial parks over 5,000 acres, National Parks exceeding 6,000 acres, and all international parks that were in existence on August 7, 1977, as well as later expansions to these areas (fig. 2-2).

Historically, EPA has regarded smoke from wildland fires as temporary and therefore not subject to issuance of a PSD permit; whether or not wildland fire smoke should be considered when calculating PSD increment consumption or PSD baseline was not defined. EPA recently reaffirmed that States could exclude prescribed fire emissions from increment analyses provided the exclusion does not result in permanent or long-term air quality deterioration (EPA 1998). States are also expected to consider the extent to which a particular type of burning activity is truly temporary, as opposed to an activity that could be expected to occur in a particular area with some regularity over a long period. Oregon is the only State that has chosen to include prescribed fire emissions in PSD increment and baseline calculations.

Visibility

The 1977 amendments to the Clean Air Act include a national goal of "the prevention of any future, and the remedying of any existing, impairment of visibility

Figure 2-2—Mandatory class 1 areas (Hardy and others 2001).

in mandatory Class I Federal areas which impairment results from manmade air pollution" (42 U.S.C § 7491). States are required to develop implementation plans that make "reasonable progress" toward the national visibility goal.

Atmospheric visibility is affected by scattering and absorption of light by particles and gases. Particles and gases in the air can obscure the clarity, color, texture, and form of what we see. Fine particles most responsible for visibility impairment are sulfates, nitrates, organic compounds, elemental carbon (or soot), and soil dust. Sulfates, nitrates, organic carbon, and soil tend to scatter light, whereas elemental carbon tends to absorb light. Fine particles (PM2.5) are more efficient per unit mass than coarse particles (PM10 and larger) at causing visibility impairment. Naturally occurring visual range in the Eastern United States is estimated to be between 60 and 80 miles, while natural visual range in the Western United States is between 110 and 115 miles (these estimates do not consider the effect of natural fire on visibility) (Trijonis and others 1991). Currently, visual range in the Eastern United States is about 15 to 30 miles and about 60 to 90 miles in the Western United States. (40 CFR Part 51). The theoretical maximum visual range about 240 miles.

Regional Haze

Regional haze is visibility impairment produced by a multitude of sources and activities that emit fine particles and their precursors and are located across a broad geographic area. This contrasts with visibility impairment that can be traced largely to a single, large pollution source. Until recently, the only regulations for visibility protection addressed impairment that is reasonably attributable to a permanent, large emissions source or small group of large sources. In 1999, EPA issued regional haze regulations to manage and mitigate visibility impairment from the multitude of diverse regional haze sources (40 CFR Part 51). The regional haze regulations call for States to establish goals for improving visibility in Class I National Parks and wildernesses, and to develop long-term strategies for reducing emissions of air pollutants that cause visibility impairment.

Regional Haze Planning Process—Because regional haze is a multi-State issue, regional haze regulations encourage States, land managers, and other stakeholders to work together to develop control programs through regional planning organizations that can coordinate development of strategies across a multi-State region. In the Western United States, the Western Regional Air Partnership (WRAP), sponsored through the Western Governors' Association and the National Tribal Environmental Council, is coordinating regional planning and technical assessments. The WRAP was the first of five regional planning organizations to be established and has been active in many technical and policy developments. Other regional planning organizations have begun assessments of fire and air quality in their regions. In the Eastern United States, four formal groups are addressing planning issues: CENRAP (Central States Regional Air Partnership), OTC (Ozone Transport Commission), VISTAS (Visibility Improvement State and Tribal Association of the Southeast); and the Midwest Regional Planning Organization (fig. 2-3).

As inter-State smoke transport becomes a larger issue, agencies are expanding coordination of their burns. Multi-State, interagency partnerships are developing to help coordinate burning and mitigate cumulative impacts of smoke. For example, the Montana/Idaho airshed group includes private, State, Tribal, and Federal partners in supporting an integrated smoke management program that includes emissions monitoring and smoke forecasting (Levinson 2001).

Regional Haze and Fire Emissions—The adoption of regional haze regulations marks a turning point in how fire emissions are treated under the nation's Federal and State air quality regulations,

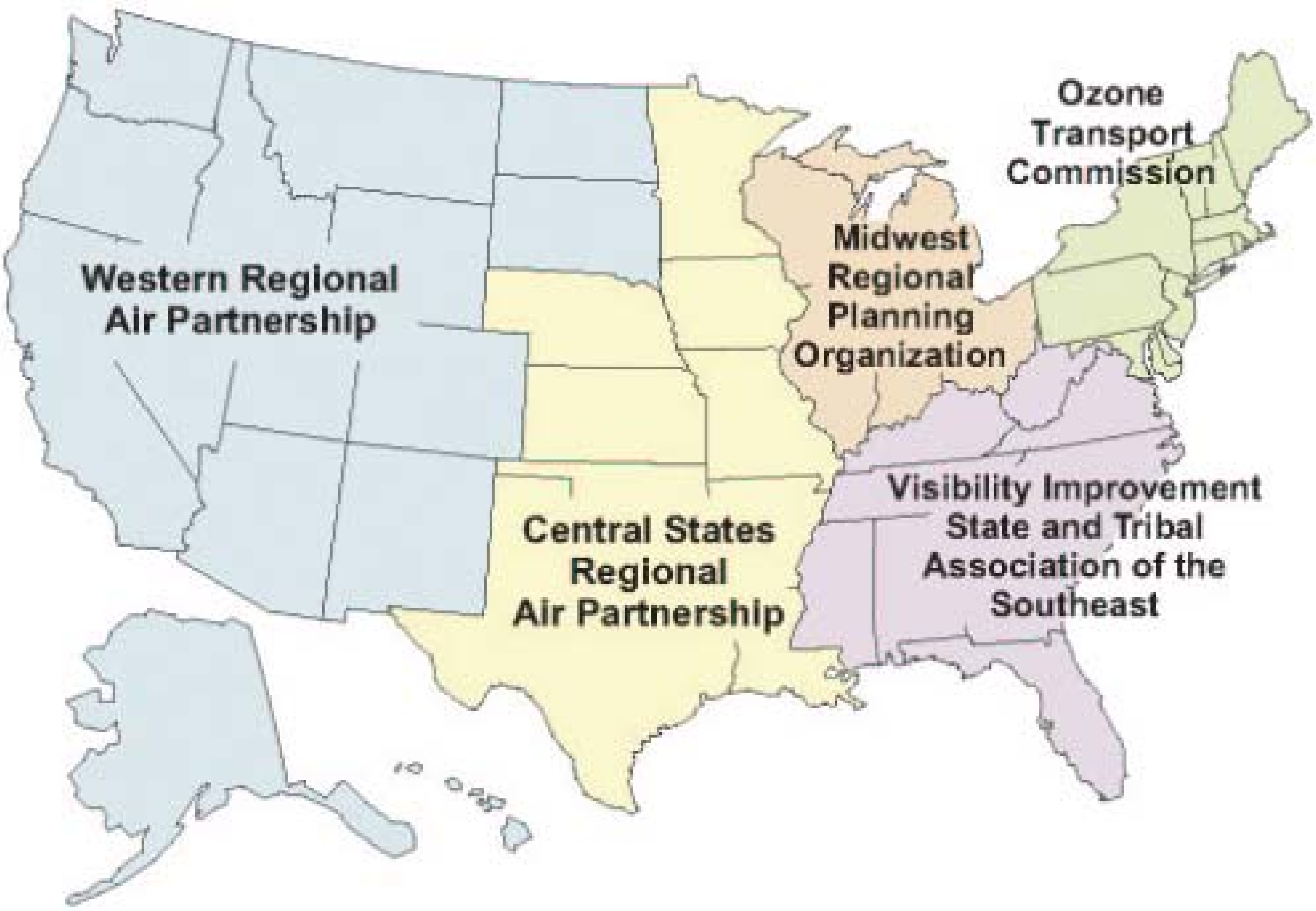

Figure 2-3—Regional air quality planning groups (Hardy and others 2001).

although the regulations leave several definitions open to subsequent policy interpretation:

- The role of fire in forest ecosystems is formally recognized for the first time.
- Emissions from “natural” sources are distinguished from “anthropogenic” sources and treated differently under the rule.
- The rule is the first to require development of emissions inventories for fire, including wildland fires.
- Emissions from fire are now subject to regional air quality planning processes as well as requirements to achieve “reasonable progress” in emissions reductions

The policy discussion to determine what types of fire emissions are considered natural is still in progress, but the WRAP has recommended a national policy that would (1) define “natural background” as fire emissions that would occur in the future without fire management; that is, without reference to historic fire occurrence or historic vegetation types; and (2) include prescribed burning as natural sources of visibility impacts when fire is used to maintain healthy and sustainable ecosystems.

Current data from a national visibility-monitoring network (Sisler and others 1996) do not show fire to be the predominant long-term source of visibility impairment in any Class I area (40 CFR Part 51), although emissions from fire are an important episodic contributor to visibility-impairing aerosols. Certainly the contribution to visibility impairment from fires can be significant over short periods, but fires in general occur relatively infrequently and thus have a lesser contribution to long-term averages. Specific goals for visibility improvement focus efforts on improving air quality on the most impaired days, so fires may prove to be an important target for control efforts in some areas

Fire Consortia for Advanced Modeling of Meteorology and Smoke (FCAMMS)—Multiagency consortia are building in the Pacific Northwest, Rocky Mountain region, and Northeastern and Southeastern United States as part of the U.S. Department of Agriculture, Forest Service, Fire Consortia for Advanced Modeling of Meteorology and Smoke. The Pacific Northwest consortium is developing a real-time smoke prediction and emission tracking system that addresses needs of several smoke management plans from collaborating States, Tribes, and local air agencies (Ferguson and others 2001). California and Nevada are working together through the California and Nevada Smoke and Air Committee (CANSAC) with similar objectives of tracking and predicting cumulative smoke impacts (Chris Fontana, personal communication).

Each group or regional consortium must respond to local, State, and Tribal smoke management programs. In addition, each region of the country has its own particular atmospheric processes that impact fire behavior and smoke dispersion in different ways. For example, while in the Southeast, timing of frontal passages and onshore flow regimes become critical, in the Western United States, complex flow through mountainous terrain is an important consideration in managing smoke. These regionally specific demands are forcing research to focus on subtle aspects of smoke emissions and dispersion instead of traditional development of worst-case air pollution scenarios.

Reasonable Progress

Visibility rules require States to make “reasonable progress” toward the Clean Air Act goal of “prevention of any future, and the remedying of any existing, impairment of visibility.” The regional haze regulations did not define visibility targets but instead gave States flexibility in determining reasonable progress goals for Class I areas. States are required to conduct analyses to ensure that they consider the possibility of setting an ambitious reasonable progress goal, one that is aimed at reaching natural background conditions in 60 years. The rule requires States to establish goals for each affected Class I area to (1) improve visibility on the haziest 20 percent of days, and (2) ensure no degradation occurs on the clearest 20 percent of days over the period of each implementation plan.

States are to analyze and determine the rate of progress needed for the implementation period extending to 2018 such that, if maintained, this rate would attain natural visibility conditions by the year 2064. To calculate this rate of progress, each State must compare baseline visibility conditions to estimate natural visibility conditions in Class I areas and to determine the uniform rate of visibility improvement that would need to be maintained during each implementation period to attain natural visibility conditions by 2064. Baseline visibility conditions will be determined from data collected from a national network of visibility monitors representing all Class I areas in the country for the years 2000 to 2004. Each State must determine whether this rate and associated emissions reduction strategies are reasonable based on several statutory factors. If the State finds that this rate is not reasonable, it must provide a demonstration supporting an alternative rate.

Hazardous Air Pollutants

Hazardous air pollutants (HAPs) are identified in Title III of the Clean Air Act Amendments of 1990

(PL 101-549) as 188 different pollutants "which present, or may present, through inhalation or other routes of exposure, a threat of adverse human health or environmental effects whether through ambient concentrations, bioaccumulation, deposition, or other routes." The list of HAPs identified in the Clean Air Act are substances that are known or suspected to be carcinogenic, mutagenic, teratogenic, neurotoxic, or which cause reproductive dysfunction.

EPA Interim Air Quality Policy on Wildland and Prescribed Fires

In 1998, the EPA issued a national policy to address how best to achieve national clean air goals while improving the quality of wildland ecosystems through the increased use of fire. The *Interim Air Quality Policy on Wildland and Prescribed Fires* (U.S. Environmental Protection Agency 1998) was developed through a partnership effort involving EPA, the U.S. Departments of Agriculture, Defense, and the Interior, State foresters, State and Tribal air regulators, and others. The group that developed the policy relied on the assumption that properly managed prescribed fires can improve the health of wildland ecosystems and reduce the health and safety risks associated with wildfire, while meeting clean air and public health goals through careful planning and cooperation among land managers, air quality regulators, and local communities.

Natural Events Policy

PM10 NAAQS exceedances caused by natural events are not counted toward nonattainment designation if a State can document that the exceedance was truly caused by a natural event and prepares a natural events action plan (NEAP) to address human health concerns during future events (Nichols 1996). Natural events are defined by this policy as wildfire, volcanic, seismic, and high wind events.

A wildfire NEAP should include commitments by the State and stakeholders to:

1. Establish public notification and education programs.
2. Minimize public exposure to high concentrations of PM10 due to future natural events such as by:
   a. Identifying the people most at risk.
   b. Notifying the at-risk public that an event is active or imminent.
   c. Recommending actions to be taken by the public to minimize their pollutant exposure.
   d. Suggesting precautions to take if exposure cannot be avoided.
3. Abate or minimize controllable sources of PM10 including the following:
   a. Prohibition of other burning during pollution episodes caused by wildfire.
   b. Proactive efforts to minimize fuel loadings in areas vulnerable to fire.
   c. Planning for prevention of NAAQS exceedances in fire management plans.
4. Identify, study, and implement practical mitigating measures as necessary.
5. Periodic reevaluation of the NEAP.

Collaboration Among Stakeholders

Because smoke from fire can negatively affect public health and welfare, air quality protection regulations must be understood and followed by responsible fire managers. Likewise, air quality regulators need an understanding of how and when fire use decisions are made and should become involved in fire and smoke management planning processes, including the assessment of when and how alternatives to fire will be used. Cooperation and collaboration between fire managers and air quality regulators is of great importance. Table 2-2 contains recommendations for various types of cooperation by these two groups depending on the applicable air quality protection instrument.

Best Available Control Measures

The application of best available control measures (BACM) for prescribed fire is a required element of State implementation plans for PM10 nonattainment areas that are significantly impacted by prescribed fire smoke (EPA 1992a). The application of BACM is also a requirement of EPA's *Air Quality Policy on Wildland and Prescribed Fires* (EPA 1998) (see "Prior Work" section in chapter 1). EPA's BACM guidance includes basic smoke management program elements and emissions reduction techniques that can be used by land managers to minimize air quality impacts from fire. These program elements and emissions reduction techniques are fully documented in the *Smoke Management Guide for Prescribed and Wildland Fire: 2001 Edition* (Hardy and others 2001).

Briefly, the BACM guidance notes that there are two basic approaches to minimizing the impact of prescribed fire on air quality: reducing the amount of pollutants emitted, or reducing the impact of the pollutants emitted on sensitive locations or regional haze through smoke dilution or transport (redistributing emissions). Although each method can be discussed

Table 2-2—Recommended cooperation between wildland fire managers and air quality regulators, depending on air quality protection instrument (Hardy and others 2001).

Air quality protection instrument	Wildland fire managers	Air quality regulators
National ambient air quality standards (NAAQS)	Aware[a]	Lead [b]
Attainment status	Aware	Lead
State implementation plan (SIP) planning and development	Involved[c]	Lead
Conformity	Involved	Lead
Smoke management programs	Partner[d]	Lead
Visibility protection	Involved	Lead
Regional planning groups	Partner	Lead
Natural emissions	Partner	Lead
Natural events action plan	Partner	Lead
Land use planning	Lead	Involved
Project NEPA documents	Lead	Involved
Other fire planning efforts	Lead	Involved

[a]Aware: Responsibility to have a complete working knowledge of the air quality protection instrument but likely little or no involvement in its development or daily implementation.

[b]Lead: Responsibility to initiate, bring together participants, complete, and implement the particular air quality protection instrument.

[c]Involved: Responsibility to participate in certain components of development and implementation of the air quality protection instrument although not at full partner status.

[d]Partner: Responsibility to fully participate with lead organization toward development and implementation of the air quality protection instrument in a nearly equal relationship.

independently, fire practitioners often choose fire and fuels manipulation techniques that complement or are at least consistent with meteorological scheduling for maximum smoke dispersion and favorable plume transport. The following emissions reduction and redistributing emissions techniques are a compilation of our knowledge base, and depending on specific fire use objectives, the project locations, time, and cost constraints may or may not be applicable.

Reducing Emissions

At least 24 methods within six major classifications have been used to reduce emissions from prescribed burning (Hardy and others 2001). These techniques include methods designed to minimize emissions by reducing the area burned; reducing the fuel load by reducing the fuel production, or fuel consumption, or both; scheduling burns before new fuels appear; and increasing combustion efficiency. Each of these methods has specific practices associated with it.

Redistributing Emissions

These measures are commonly practiced in smoke management programs and include burning when dispersion is good, cooperating with other burners in a single airshed to schedule burns, avoiding sensitive areas, burning smaller units, and burning more frequently.

Ozone and Fire

Ozone is a criteria air pollutant, but there is little monitoring or research data that directly link fire emissions with ground-level ozone concentrations. Regulating efforts to reduce ozone have therefore focused on more obvious industrial and urban sources of the pollutants that form ozone (NO_X and VOCs). Fires are known to emit VOCs and a minor amount of NO_X, but much is uncertain about the magnitude of ozone formation in the plume, the degree of mixing with urban sources of ozone precursors, and transport of ozone to ground level. EPA plans to begin including fire emissions in future regional ozone strategy modeling. Field observations of ozone formation in smoke plumes from fires date back nearly 25 years when measurements from aircraft detected ozone at the edge of forest fire smoke plumes aloft. A recent study (Wotawa and Trainer 2000) did link high ground-level ozone concentrations to forest fire plumes that had been transported great distances. Chapter 6 explores these issues more fully.

Notes

Chapter 3: Overview of Air Pollution from Fire

This chapter provides a brief overview of and an appreciation for the national, regional, and local importance of smoke to ambient air quality. We discuss the significance of fire emissions and air quality impacts on a national and regional scale. Chapter 7 of this document adds additional depth to this discussion.

Magnitude of Fire Contributions

Air quality impacts associated with wildland fires are distinguished from those resulting from prescribed burning because emissions from these two sources have in the past been treated differently under the Clean Air Act and by State and local air quality regulations. In addition, it is important to have a historical perspective of these issues given the increased use of fire in the recent past.

A comparison by Leenhouts (1998) of estimated levels of biomass burning suggests that 10 times more area burned annually in the pre industrial era than in the contemporary era. After accounting for land use changes such as urbanization and agriculture, Leenhouts concluded that about 50 percent of historical levels would burn today if historical fire regimes were restored to all wildlands to maintain ecosystem health (figs. 3-1 and 3-2). This suggests a four- to sixfold increase from the current magnitude of wildland fire emissions.

This section discusses: (1) smoke from wildland fires; (2) smoke from prescribed fires; (3) impacts on national ambient air quality standards (NAAQS); (4) and magnitude with respect to regional and subregional scale visibility degradation. The second section discusses smoke management programs.

Smoke from Wildland Fires

Although wildland fires occur throughout the nation, the largest fires and greatest number of fires occur in Alaska, the Southeastern States, and the West. Figure 3-3 shows the location of major fires during the 2000 fire season when 90,674 fires burned 7,259,159 acres (2,938,931 ha) at a fire suppression cost of $1.6 billion. The 10-year average acreage burned between 1990 and 1999 was 3.78 million acres (1.53 million ha), testifying to the severity of the 2000 wildfire season. Figure 3-4 shows those States that had more than 100,000 acres (40,486 ha) burned per year, on average, over the 1987 through 1997 period, illustrating that Alaska wildfires burn far more acres than fires in any other State. Area burned in California, the States in the Intermountain West, Florida, and the Southwest follow (Peterson 2000).

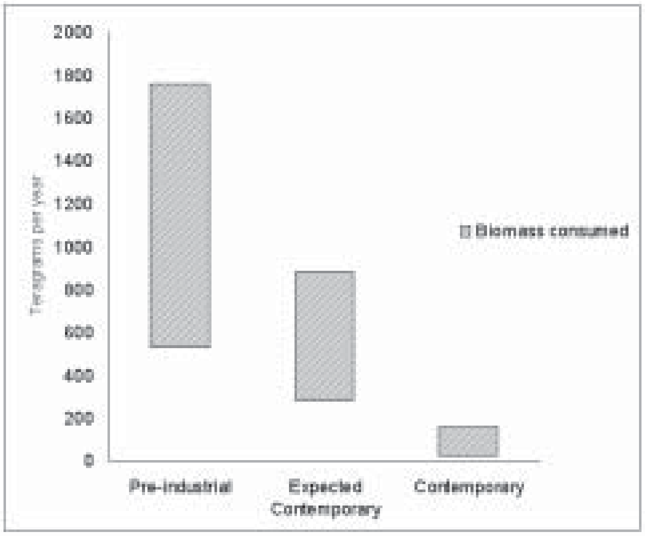

Figure 3-1—Estimated annual preindustrial, expected contemporary, and contemporary area (Mha) for the conterminous United States (from Leenhouts 1998).

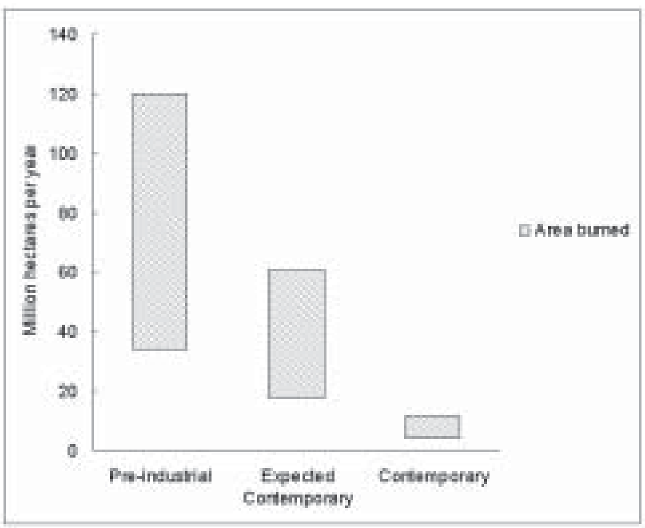

Figure 3-2—Estimated annual preindustrial, expected contemporary, and contemporary biomass consumed (Tg x10^2) for the conterminous United States (from Leenhouts 1998).

Wildfires occur throughout the year. The 2000 wildfire season began with a Florida fire on January 1, continued with two 40,000-acre fires in New Mexico, an early May, 47,000-acre fire near Los Alamos and peaked on August 29, 2000, when fires that eventually burned 1,642,579 acres were burning in 16 States (NIFC 2001a). Generally, the occurrence of wildfires moves northward from the Southeastern and Southwestern States as summer approaches, fuels dry and fire danger increases.

Wildfires, both in number and total acreage burned, vary widely from year to year and from region to region. Figures 3-5 and 3-6 show no consistent relation between the number of fires and acres burned. It is known, however, that smoke from these fires impacts air quality on both an episodic and long-term average basis over wide regions.

Wildfires occur as episodic events. For example, in 1999, smoke from fires reduced visibility to less than 100 feet (30 m) in Florida, prompting officials to advise people with respiratory problems to stay indoors (New York Daily News 1999). In the West, fires in six States (California, Nevada, Oregon, Montana, Washington, and Idaho) put thick smoke in many communities. In Reno and cities in California's Central Valley, smoke from nearby wildfires prompted authorities to warn residents with asthma to avoid unnecessary activity (USA Today 1999). Wildfire smoke is also transported across international boundaries. Fires in Canada were found to cause high concentrations of carbon monoxide and ozone over a period of 2 weeks in the Southeastern United States and across the Eastern seaboard during the summer of 1995 (Wotawa and Trainer 2000).

Smoke impacts during these episodic events can threaten public health, cause smoke damage to buildings and materials, and disrupt community activities. Although particulate concentrations in ambient air rarely reach health-threatening levels within major cities, several communities in the United States have experienced particulate matter concentrations from wildfire smoke that exceeded the Environmental Protection Agency (EPA) significant harm emergency action level of 600 $\mu g/m^3$ defined as an "imminent and substantial endangerment of public health" (EPA 1992b).

For example, the Yellowstone National Park wildfires of 1988 impacted communities in three States. Concentrations of suspended particulate matter — both total suspended particulate (TSP) and PM10 — measured in communities near the fires exceeded NAAQS, triggering public health alerts and advisories (Core 1996). An estimated 200,000 people were exposed to high concentrations of smoke. In 1987, the Klamath fires of northern California burned for more than 60 days, resulting in widespread smoke intrusions into numerous communities in northern California and southern Oregon. More recently, wildfire impacts during the 2000 season were also severe in several communities. Twenty-four average PM10 concentration measured in Salmon, ID, reached 225 $\mu g/m^3$ on August 15, 2002, and 281 $\mu g/m^3$ on August 18, 2000, during wildfire smoke intrusions (Idaho Department of Environmental Quality n.d.).

Wildfire smoke can also be the dominant cause of visibility reduction during episodic events in the Rocky Mountain States, on the Pacific Coast, and in the Southeast (National Research Council [NRC]

Figure 3-3—Location of major wildfires in 2000 available at http://www.nifc.gov/fireinfo/2000/Top10fires.html.

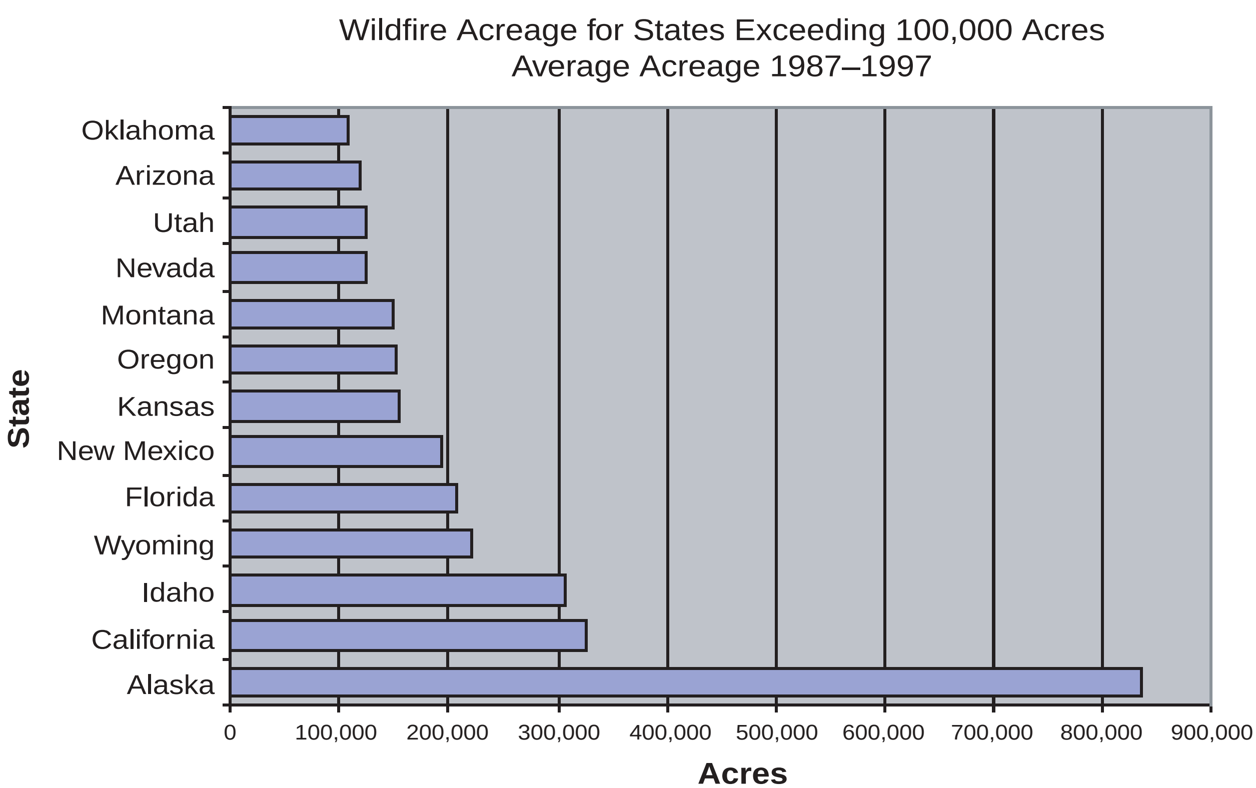

Figure 3-4—States with more than 100,000 acres per year burned by wildfires.

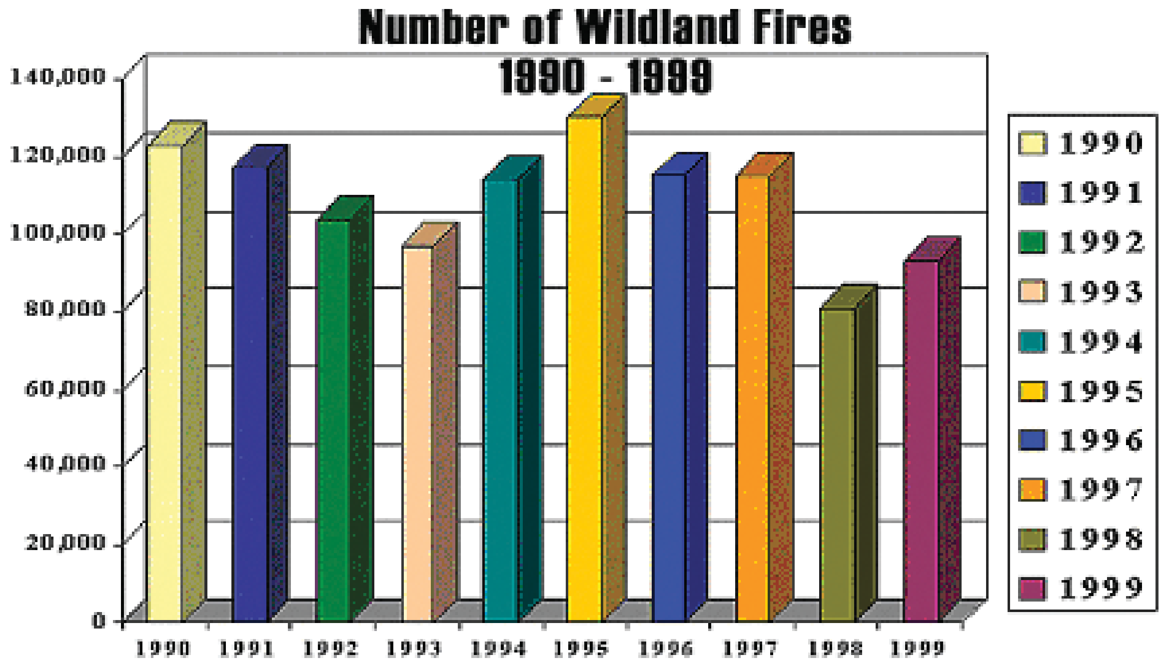

Figure 3-5—Number of wildfires per year 1990 through 1999 (National Interagency Fire Center 2002).

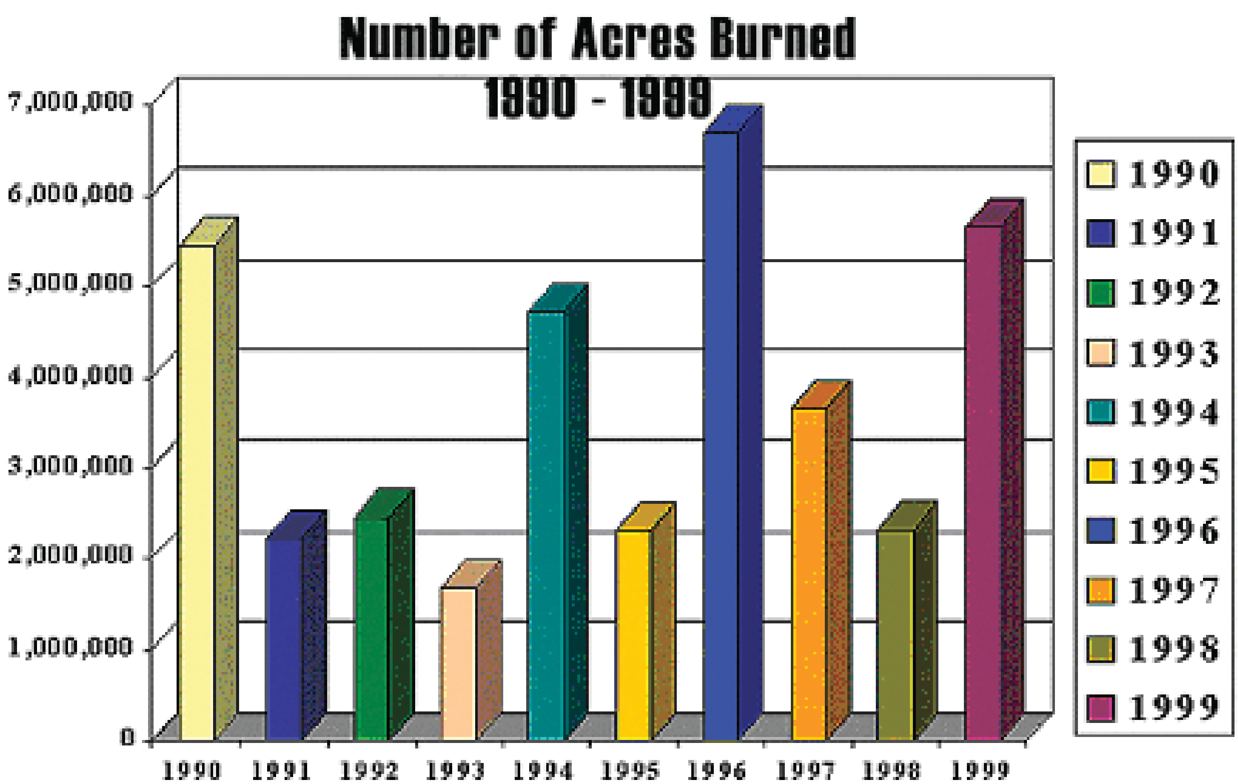

Figure 3-6—Number of acres burned by wildfires per year 1990 through 1999.

1993). Figures 3-7 and 3-8 are examples of the dense plumes of smoke that can be transported over hundreds of kilometers across State and international boundaries, degrading air quality, scenic values, and highway safety. Between 1979 and 1988, 28 fatalities and more than 60 serious injuries were attributed to smoke that drifted across roadways in the Southern United States (Mobley 1989).

Figure 3-7—Big Bar Fire, Shasta-Trinity National Forest, California, August 1999 (National Interagency Fire Center 2000).

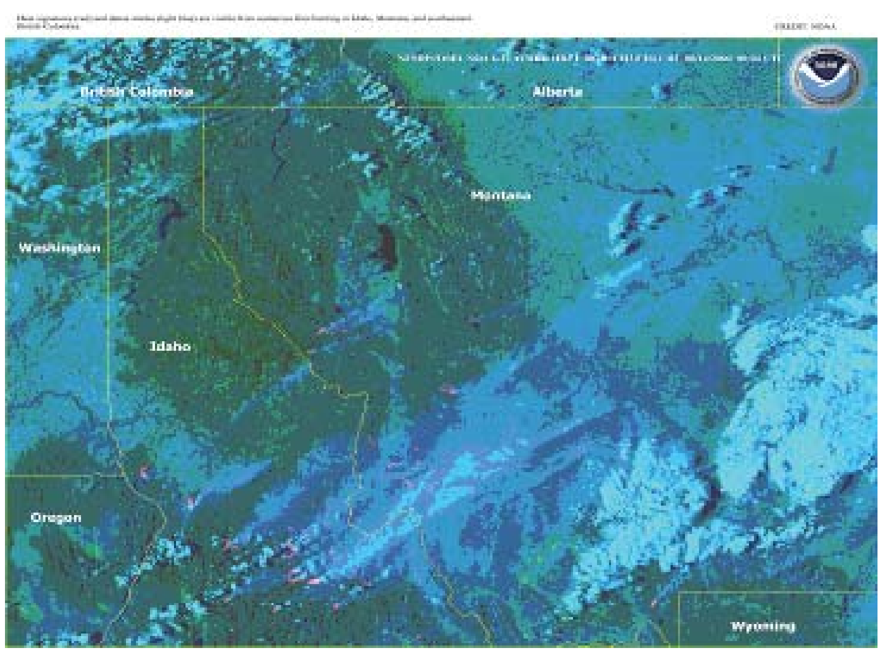

Figure 3-8—Wildfire smoke transported across State lines, August 14, 2000 (NASA).

Smoke from Prescribed Fires

On a national annual basis, PM10 emissions from prescribed burns in 1989 were estimated to be over 600,000 tons, half of which (380,000 tons) occurred in the Southeastern States. Of the remaining 42 States, seven (Arizona, California, Idaho, Montana, Oregon, Texas, and Washington) were estimated to have annual emissions over 10,000 tons of PM10 from prescribed forest and rangeland burning (EPA 1992a; Peterson and Ward 1990). More recent estimates of prescribed fire PM2.5 emissions in the West (EPA regions 8, 9 and 10) totaled 193,293 tons (Dickson and others 1994). These national, annual estimates are less significant in terms of air quality impact than those prepared at the State level. For example, the 211,000 tons of prescribed fire PM10 emissions in Georgia in 1989 is about 30 percent of the total estimated particulate inventory for all sources (EPA 1992a). On a seasonal basis, emissions from prescribed burning are likely to be an even more significant percentage of total emissions in some States.

Acreage treated by prescribed burning on Federal lands increased from 918,300 acres in 1995 to 2,240,105 acres in 1999, demonstrating renewed interest in the use of fire as an important tool in the management of wildlands (NIFC 2001b).

Impacts on National Ambient Air Quality Standards

Characterization of the true extent of effects of prescribed and wildland fires on ambient air quality is incomplete due to the deficiency of air quality monitoring sites in rural areas. Also, particulate standards are based on 24-hour and annual averages, whereas smoke plumes may significantly degrade air quality in a community for just a few hours before moving or dispersing. These short-term, acute impacts likely cause discomfort at the least, and possibly even affect health, but may not result in a violation of the NAAQS.

Numerous exceedances of 24-hour PM10 and PM2.5 standards have been attributed to wildfires but, as mentioned previously, violations of NAAQS caused by wildfire do not result in nonattainment if a State can document that the cause of the violation was truly wildfire and then prepares a natural events action plan for future events.

At present, prescribed fires are not considered to be a significant cause of nonattainment, but with increased burning to reduce fuels, this situation may change as land managers move forward with implementing a several-fold increase in the use of fire to sustain ecosystems (USDI and USDA 1995; USDA 1997). In general, little information is available on a national level to identify the contribution of prescribed burning to PM10 or PM2.5 within nonattainment areas (EPA 1992a). It appears, however, that there is no clear relation between total acres burned (or particulate emissions) and the nonattainment status of nearby airsheds, possibly because of successful smoke management programs.

In areas where air quality standards are being or may be violated, however, land managers are being directed to reduce air quality impacts through smoke management programs. This is because any source that contributes even a few micrograms per cubic meter of particulate matter toward violation of the NAAQS may be required to reduce emissions to assure that air quality standards are attained.

Significance of Visibility Degradation

As noted above, wildland fires can significantly degrade visibility during episodic events. With the new emphasis on the reduction of regional haze in the Class I National Parks and wilderness areas of the nation, smoke from fire is of special concern, especially in the West. In their report to the EPA, the Grand Canyon Visibility Transport Commission (GCVTC) noted that emissions from fire, both wildland fire and prescribed fire, are likely to have the single greatest impact on visibility at Class I areas through 2040. During periods of intense fire activity, smoke from wildland fires is likely to make the worst 20 percent of days at the Grand Canyon even worse rather than impair visibility on clear days (GCVTC 1996b). The Commission recommended several actions to reduce impacts on regional haze including enhanced smoke management programs and establishment of annual emissions goals for all fire programs.

Greenhouse Gas Emissions from Fires

Globally, fires are a significant contributor of carbon dioxide and other greenhouse gases in the atmosphere. Fires account for approximately one-fifth of the total global emissions of carbon dioxide (Levine and Cofer 2000; Schimel 1995). Andreae and Merlet (2001) calculate that 5,130 Tg per year of biomass is consumed in fires, emitting 8,200 Tg per year of carbon dioxide, 413 Tg per year of carbon monoxide, and 19.4 Tg per year of methane. The accuracy of these global estimates is thought to be within plus or minus 50 percent, with the bulk of the error resulting from inaccuracies in the estimates of the area burned and the mass of fuel consumed.

Fires in temperate ecosystems are minor contributors compared to the world's savannas, boreal forests, and tropical forests. More than 60 percent of the totals listed in the previous paragraph are released from savannas and grasslands, and another 25 percent from tropical forests. Burning in tropical Africa is dominated by savanna fires; in tropical Asia, by forest fires; and in tropical South America, about equally represented by savannas and tropical forests (Hao and Liu 1994). Lavoué and others (2000) detail contributions from temperate and boreal fires, demonstrating that about 90 percent of the global boreal fire area is in Russia and Canada. Alaska accounts for only about 4.5 percent of the global boreal forest, but it accounts for at least 10 percent of the emissions from that source, because of the heavier fuel loads in Alaska. Alaska accounts for an average of 41 percent of total U.S. fire emissions, with a huge year-to-year variability. In 1990, 89 percent of U.S. fire emissions were from Alaska fires.

Smoke Management Programs

Smoke management programs establish a basic framework of procedures and requirements when managers are considering resource benefits. These programs are typically developed by States and Tribes with cooperation and participation by wildland owners and managers. The purposes of smoke management programs are to mitigate the nuisance (such as impacts on air quality below the level of ambient standards) and public safety hazards (such as visibility on roads and airports) posed by smoke intrusions into populated areas; to prevent significant deterioration of air quality and NAAQS violations; and to address visibility impacts in Class I areas.

The *Interim Air Quality Policy on Wildland and Prescribed Fires* (EPA 1998) provides clear guidelines for establishing the need for and content of smoke management programs and assigns accountability to State and Tribal air quality managers for developing and adopting regulations for a program. Measured PM10 NAAQS exceedances attributable to fires, including some prescribed fires and wildland fires managed for resource benefits, can be excluded from air quality data sets used to determine attainment status for a State. Special consideration will be given if the State or Tribal air quality manager certifies in a letter to the administrator of EPA that at least a basic smoke management program has been adopted and implemented.

States with smoke management programs that have authorized a central agency or office to make burn/no-burn decisions include Arizona, Colorado, Oregon, Idaho/Montana, Washington, California, Nevada, New Mexico, Florida, South Carolina, Utah, North Carolina, and Wyoming (Battye and others 1999). In many other States, the decision to burn rests in the hands of the persons conducting the burn, local fire departments, or local authorities. These States include Alaska, Alabama, Arkansas, Georgia, Louisiana, Mississippi, Tennessee, Texas, and Virginia. In yet other States (New York, Illinois, Massachusetts, and others), burn permits are required and may be subject to State air agency oversight if burning is conducted near nonattainment areas or areas sensitive to smoke (Core 1998; Hardy and others 2001). In addition, many private landowners, nonprofit conservation organizations and government agencies voluntarily practice responsible smoke management to maintain goodwill in their communities.

Smoke management programs have been established and are operated on an on-going basis because of local, regional, and national concerns about the impact of prescribed burning on air quality. The number, complexity, and cost of operating these programs underscore the potential significance of prescribed fire's impact on air quality on a national scale.

Smoke management programs across the nation have changed significantly since the mid-1980s. In the Pacific Northwest, there have been reductions in prescribed fire smoke management programs because of the decline in large-scale clearcut burning of forest harvesting residues. Current smoke management programs across the West have to place a much greater focus than in the past on understory burning to restore declining forest health, on burns to reduce fire hazards, or on burns to meet wildlife habitat objectives. All across the nation, an increasing number of people living within the wildland-urban interface have placed new emphasis on the need to minimize smoke impacts on residents living near fires. Increasing air quality regulatory pressures, fire manager liability issues, and the increased likelihood of fire escapement in overstocked forestlands have all placed ever-greater demands on fire practitioners.

As these demands have increased, so have the number and complexity of smoke management programs nationwide (Hardy and others 2001). Although the complexity of these programs varies widely from State to State, the key to a successful program always lies in its ability to balance the use of prescribed fire with air quality, environmental, legal, and social requirements. Increasingly, this has meant adoption of formalized burn authorization procedures issued by program managers who are responsible for overseeing burning on both public and private lands on a daily basis. Coordinated burn operations are based on meteorological forecasts, the location of smoke-sensitive receptors, fuel conditions, and a myriad of other considerations. Increasingly, public notification of planned burning activity and monitoring of smoke transport, as well as fire practitioner training and program enforcement, are becoming more common (Battye and others 1999).

As inter-State smoke transport becomes a larger issue, agencies are expanding coordination. For example, land management agencies in California's

San Joaquin Valley are using a new centralized, electronic database, Prescribed Fire Incident Reporting System (PFIRS), to schedule fires and to share information on expected emissions and smoke transport with California and Nevada air and land management agencies (Little n.d.). This trend is likely to continue as States begin to work on regional haze control programs.

The Western Regional Air Partnership (WRAP) Fire Emissions Joint Forum (FEJF) has issued a draft policy to set the criteria for enhanced smoke management plans for visibility protection in the West (Fire Emissions Joint Forum 2002). The policy document concludes that the regional haze rule can be satisfied only by the States and Tribes establishing an emission tracking system for all prescribed fires and wildland fires; by managing smoke from all fires; and by implementing smoke management systems that include nine elements:

1. Actions to minimize emissions from fire
2. Evaluation of smoke dispersion
3. Alternatives to fire
4. Public notification of burning
5. Air quality monitoring
6. Surveillance and enforcement
7. Program evaluation
8. Burn authorization
9. Regional coordination

The enhanced smoke management plan (ESMP) policy would enable Western States and Tribes to minimize increases in emissions and show reasonable progress toward the natural visibility goal. The Fire Emissions Joint Forum is developing additional policy and technical tools that will support ESMP policy and its implementation, such as recommendations for creation of an annual emissions goal, availability and feasibility of alternatives to burning, recommendations for managing fire emissions sources, guidance for feasibility determinations, and a method for tracking fire emissions.

Chapter 4: Characterization of Emissions from Fires

All fires emit air pollutants in addition to nonpolluting combustion products; but fires vary widely in what pollutants are emitted in what proportion. Characterizing and managing air pollution from fires first requires knowledge of the amount and timing of what pollutants are emitted. Fires are a complex combustion source that involve several stages of combustion, several categories of fuels, and fire behavior that changes over time and with fuel and weather conditions; so the amount, rate, and nature of pollutants also vary widely. Characterizing emissions from fires requires explicit knowledge of fuelbed character and condition, combustion environment, and fire behavior.

This chapter reviews the state of knowledge and predictive models necessary to characterize air pollutant emissions from prescribed and wildland fires.

All components of smoke from fires, with the exception of carbon dioxide and water, are generated from the inefficient combustion of biomass fuels. The amount of smoke produced is derived by determining the fuel consumed (tons per acre) in each combustion stage and knowing the size of the area burned, fuel characteristics, fire behavior, and combustion conditions (fuel moisture, weather parameters, and so forth). The fuel consumption is then multiplied by an emissions factor for each pollutant, which is an expression of the efficiency of combustion. An emission factor is the ratio of the mass of pollutant per unit mass of fuel consumed, and is a statistical average of measurements made in the plumes of fires containing differing fuel types and combustion stages. Errors and uncertainties arise in the estimates made during each step in the process of estimating emissions.

Area Burned

At first glance, amount of area burned seems relatively easy to calculate. However, individual estimates of fire size tend to be systematically exaggerated, and fires are frequently double-counted in inventories. For example, geographic features, nonuniform fuelbeds, or a change in the weather will often cause a fire to create a mosaic of burned, partially burned, and unburned areas, although the entire landscape within the fire perimeter is often reported as burned. In addition, large-scale (such as continental) inventories of area burned are often derived from remote sensing data that have resolutions from 250 m to 1 km (SAI 2002), limiting their precision. Remote sensing accuracy is currently inadequate in landscapes that change slope and fuel characteristics over a few tens of meters.

Preburn Fuel Characteristics

Large variations in fuel characteristics can contribute up to 80 percent of the error associated with predicting emissions (Peterson 1987; Peterson and Sandberg 1988). Fuel characteristics can vary widely across the landscape (figs. 4-1 and 4-2). For instance, fuel loads can range from less than 3 tons per acre for perennial grasses with no rotten woody material or duff, 6 tons per acre in a sagebrush shrubland, 60 tons per acre in a ponderosa pine and Douglas-fir forest with rotten woody material, stumps, snags, and deep duff, to 160 tons per acre in a black spruce forest with deep moss and duff layer. The greatest errors occur when the fuel load is inferred from vegetation type as is usual when deriving biomass emissions from remotely sensed data (Crutzen and Andrae 1990; Levine 1994). Preburn fuel characteristics, such as relative abundance for particular fuelbed components (grasses, shrubs, woody fuels, litter, duff, and live vegetation) and the condition of the fuel (live, dead, sound, rotten) are needed to calculate fuel consumption, and the resulting smoke.

The ongoing development of several techniques, including the natural fuels photo series (Ottmar and Vihnanek 2000a) and the fuel characteristic classification (FCC) system (Sandberg and others 2001), will provide managers new tools to better estimate fuel loadings and reduce the uncertainty that currently exists when assigning fuel characteristics across a landscape. The photo series is a sequence of single and stereo photographs with accompanying fuel characteristics. The FCC is a national system designed for classifying wildland fuelbeds according to a set of

Figure 4-1—Fuelbed types and fuel loads (a) grassland (3 tons per acre), (b) sagebrush (6 tons per acre), (c) ponderosa pine with mortality in mixed fir (60 tons per acre), and (d) black spruce with deep duff and moss (160 tons per acre). (Photos by Roger Ottmar)

Figure 4-2—Various fuelbeds across a single landscape. (Photo by Roger Ottmar)

inherent physical properties, thereby providing the best possible fuels estimates and probable fire parameters based on available site-specific and remotely sensed information.

Fire Behavior

Fire behavior is the manner in which fire reacts to the fuels available for burning (DeBano and others 1998) and is dependent upon the type, condition, and arrangement of fuels, local weather conditions, topography, and in the case of prescribed fire, ignition pattern and rate (fig. 4-3). Important aspects of fire behavior include:

- Fire intensity (rate of energy release per unit area or unit length of fire perimeter, generally during the flaming combustion period).
- Rate of spread (rate of advancement of flaming front, length per unit time), crowning potential (involvement of tree and shrub foliage and spread within the canopy), smoldering potential (smoldering combustion of fuels that have been preheated or dried during the flaming stage).
- Residual smoldering potential (propagation of a smoldering combustion front within porous fuels such as rotten logs or duff, independent of preheating or drying).
- Residence time in the flaming, smoldering, and residual stages of combustion.

These aspects influence combustion efficiency of consuming biomass, as well as the resulting pollutant chemistry and emission factor (fig. 4-4).

The Emissions Production Model (EPM) (Sandberg 2000; Sandberg and Peterson 1984) and FARSITE (Finney 1998) take into account fire behavior and ignition pattern to estimate emission production rates. Fire behavior during the flaming stage of combustion in surface woody fuels and some shrub vegetation is effectively predicted within models such as BEHAVE (Andrews and Bevins 1999) and its spatial application, FARSITE (Finney 1998). However, EPM and other applications do not consider fire intensity or other fire behavior attributes when estimating emissions from flames, and that may result in a reasonable approximation for criteria pollutants but also be a limitation to the estimate of hazardous air pollutants or trace gases. BURNUP (Albini and Reinhardt 1997), FARSITE (Finney 1998), and EPM v2.0 (Sandberg 2002) attempt to model the extent and duration of flaming and smoldering combustion in downed woody fuels and duff. Current capability to model residual combustion, combustion in rotten logs and duff, and fire behavior in the foliage canopies of trees and some shrubs remains inadequate to predict emission rates with any reasonable degree of accuracy.

The Los Alamos and Lawrence Livermore national laboratories offer an approach to predicting fire behavior, plume trajectory, and dispersion, by combining a fire physics model, FIRETEC, with a dynamic atmosphere model, HIGRAD, to produce a highly detailed numerical simulation of fire spread and atmospheric turbulence (Bradley and others 2000). The approach builds on prior experience in predicting the dispersion of hazardous air pollutants from fires such as burning oil fields or "nuclear winter" scenarios. This modeling approach is limited to the propagating front but is unique in its coupling of atmospheric and fire physics.

Figure 4-3—Fire behavior in the leaf layer of a longleaf pine forest. (Photo by Roger Ottmar)

Figure 4-4—Fuel consumption in (a) large rotten log during a fall prescribed burn, (b) pile burning during a prescribed burn, (c) litter and duff during a prescribed burn, (d) grass during a wildfire, and (e) sagebrush during a prescribed fire. (Photos by Roger Ottmar)

Combustion Stages

At least three important stages of combustion exist when fuel particles are consumed (Mobley 1976; NWCG 1985): flaming, smoldering, and residual (also known as "glowing," "residual smoldering," or "residual combustion") (fig. 4-5). The efficiency of combustion is distinct for each stage, resulting in a different set of chemical compounds and thermal energy being released at different rates into the atmosphere. In the flaming phase, combustion efficiency is relatively high and usually tends to emit the least amount of pollutant emissions compared with the mass of fuel consumed.

The predominant products of flaming combustion are CO_2 and water vapor. During the smoldering phase, combustion efficiency is lower, resulting in more particulate emissions generated than during the flaming stage.

Smoldering combustion is more prevalent in certain fuel types such as duff, organic soils, and rotten logs, and often less prevalent in fuels with high surface to volume ratios such as grasses, shrubs, and small diameter woody fuels (Sandberg and Dost 1990).

Figure 4-5—Flaming, smoldering, and residual combustion stages during a fire. (Photo by Roger Ottmar)

The residual stage differs from the smoldering stage in that the smoldering stage is a secondary process that occurs in fuels preheated or dried by flaming combustion, while residual is an independent process of propagation in a fuelbed unaffected by the flaming stage. This phase is characterized by little smoke and is composed mostly of CO_2 and carbon monoxide. All combustion stages occur sequentially at a point, but simultaneously on a landscape.

Fuel Consumption

Fuel consumption is the amount of biomass consumed during a fire and is another critical component required to estimate emissions production from fire. Biomass consumption varies widely among individual fires depending on the fuelbed type, arrangement, and condition, weather parameters, and the way the fire is applied in the case of prescribed fire. As with fuel characteristics, extreme variations can be associated with fuel consumption resulting in an error contribution of 30 percent or more when emissions are estimated (fig. 4-6) (Peterson 1987; Peterson and Sandberg 1988).

Biomass consumption of woody fuels, piled slash, and duff in forested areas has become better understood in recent years (Albini and Reinhardt 1997; Brown and others 1991; Ottmar and others 1993; Ottmar and others [N.d.]); Reinhardt and others 1997; Sandberg 1980; Sandberg and Dost 1990). Consumption of forested crowns and shrublands are the least understood components of biomass consumption, and research is currently under way (Ottmar and Sandberg 2000) to develop or modify existing consumption equations for these fuel components. Equations for predicting biomass consumption in the flaming and smoldering combustion stages are widely available in two major software packages, Consume 2.1 (Ottmar and others [N.d.]) and the First Order Fire Effects Model (FOFEM 5.0) (Reinhardt and Keane 2000).

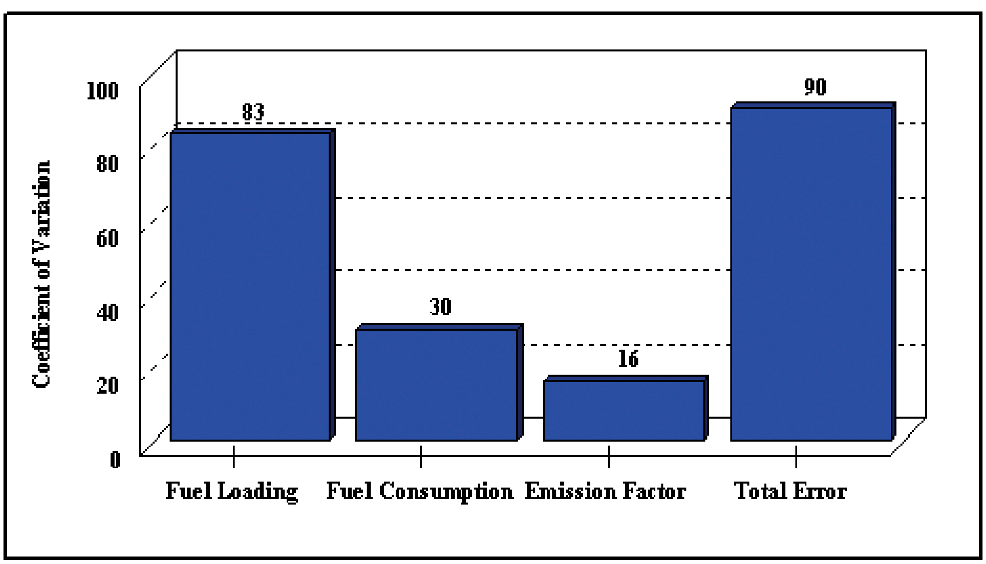

Figure 4-6—The largest errors are associated with fuel loading and fuel consumption estimates when determining emission production and impacts from wildland fire (Peterson and Sandberg 1988).

Emission Factors

Emissions from fires or from points over fires have been observed extensively by researchers since about 1970. The result is a complete set of emission factors (pounds of pollutant per ton of fuel consumed) for criteria pollutants and many hazardous air pollutants for most important fuel types. These are available in several publications (for example, Battye and Battye 2002, EPA 1972, Hardy and others 2001, Ward and others 1989) and are not reproduced here.

Less complete compilations of emission factors are for particulate matter components such as size class distribution, elemental and organic carbon fractions, and particulate hazardous air pollutants; and for methane, ammonia, aldehydes, compounds of nitrogen, volatile organic hydrocarbons, and volatile hazardous air pollutants (for example, Battye and Battye 2002, Goode and others 1999, Goode and others 2000, Lobert and others 1991, McKenzie and others 1994, and Yokelson and others 1996).

Source Strength

Source strength is the rate of air pollutant emissions in mass per unit of time, or in mass per unit of time per unit of area. Source strength is the product of the rate of biomass consumption (that is, fuel consumption) and an emission factor for the pollutant(s) of interest and is representative of the physical and chemical fuel characteristics (fig. 4-7). Source strength or emission rate is required as an input to dispersion models (Breyfogle and Ferguson 1996), or to break down emission inventories into time periods shorter than the duration of a fire event. Source strength is also required in photochemical models such as the community multiscale air quality model (CMAQ) (Byun and Ching 1999) to account for timing of chemical reactions with diurnal patterns and interaction with other sources.

Total emissions from a fire or class of fires are the source strength integrated over the time of burning. Total emissions from a single class of fires (that is, a set of fires similar enough to be characterized by a single emission factor) can be estimated by multiplying that emission factor by the level of activity, which is the total biomass consumed by the class of fires. An emission inventory is the aggregate of total emissions from all fires or classes of fire in a given period for a specific geographic area.

Managing the source strength (or level of activity) of fires is the most direct way to control air pollution from wildland and prescribed fires. Prediction of source strength is sometimes used to manage the rate of emissions from fires, and it also is needed as an input to dispersion models. Standards or regulations are commonly set to limit the total emissions of pollutants, emission of specific hazardous air pollutants, or the level of activity, so that estimates of biomass consumption can be essential for environmental assessment, permitting of prescribed fires, or measuring compliance. Emission inventories are a critical part of impact analyses and strategy development so the level of activity must be estimated whenever there is a regulatory application.

The Emissions Production Model (Sandberg 2000; Sandberg and Peterson 1984) is currently the most widely used model for predicting source strength for prescribed fires. EPM v.1 predicts flaming and residual emissions rates for each criteria pollutant based on a simple formula that assumes a constant rate of ignition of a prescribed fire in uniform fuels. The software package pulls fuel consumption predictions from Consume 2.1 or FOFEM 5.0 and uses ignition pattern, ignition periods, and burn area components to calculate source strength for the flaming and residual combustion phases. EPM v.1 does not consider smoldering emissions (for example, long-duration, self-propagating glowing combustion), multiple fires or multiple burn periods, wildland fire or piled burning emissions, or diurnal and spatial changes in the fire environment. EPM v.2, now under development (Sandberg 2000), corrects all of these shortcomings in a dynamic simulation model. EPM v.2 will satisfy the requirement to provide hourly estimates of emission rates for most fires and fuelbeds needed for input into Models-3/CMAQ (see the "Grid Models" section in chapter 5) and into currently envisioned smoke management screening systems.

Figure 4-7—A high-intensity Alaska wildfire with heavy fuel loads, causes a high rate of emissions. (Photo by Roger Ottmar)

FARSITE (Finney 2000) has been modified to predict emissions source strength as well as fire behavior in a detailed spatial simulation. FARSITE incorporates BURNUP (Albini and Reinhardt 1997), which estimates consumption and rates of individual fuel elements.

Accurate characterization of emissions from fires is critical to predicting the impact emissions will have upon communities and across broader landscapes and airsheds. Managers will increasingly be required to provide this type of information prior to prescribed burns, as well as during the course of wildland fires, and the information provided here summarizes the strengths and weaknesses of the various means of prediction.

Notes

Chapter 5: Transport, Dispersion, and Modeling of Fire Emissions

To anticipate the impacts of smoke, the timing and location of smoke concentrations become important. Data on the site-specific surface concentrations of respirable particles and gases often are needed for estimating impacts on public health and welfare, requiring atmospheric dispersion and transport models that can approximate the atmospheric physics and chemical reactions that occur during transport near the ground. Data on the cumulative concentrations of elements that scatter and absorb light also are needed to estimate impacts on visibility and haze, requiring models that can approximate aqueous reactions as well as physical and chemical reactions at all levels of the atmosphere.

Although progress is being made, none of the currently available models fully meet the needs of fire planners and air resource managers. Much of the deficiency in current modeling approaches is caused by inherent uncertainties associated with turbulent motions between the fire, smoke, and the atmosphere that are compounded by the highly variable distribution of fuel elements, composition, and condition.

Another source of deficiency is that most available models were originally designed for well-behaved sources such as industrial stacks or automobile emissions, while emissions from fire can be extremely variable in both time and space. Also, outputs from currently available models do not always match the temporal or spatial scale needed for land management application.

To help readers understand the strengths and weaknesses of available models, we describe basic elements of the trajectory and dispersion of smoke. This chapter concludes with a summary of currently available models and a brief guide to applications.

Basic Elements of Trajectory and Dispersion

Ambient air quality can be measured at a point or as distribution of air quality over any space and time of interest. Ambient air quality is affected by the pollutants emitted to the atmosphere from fires, the background air quality that has already been degraded by other sources, the transport of the polluted parcels of the atmosphere, dispersion due to atmospheric movement and turbulence, secondary reactions, and removal processes. Plume rise is an important component of transport, because it determines where in the vertical structure of the atmosphere dispersion will begin.

Overall, dispersion has proven extremely difficult to model accurately, especially in complex terrain. For example, detailed, gridded, three-dimensional meteorological data are required to model transport and dispersion, but expert judgment is often required to supplement or substitute for such modeled predictions.

Despite the difficulties of modeling, since about 1990 modeling systems used to assess the air quality impact of fires have grown increasingly important to both the fire planning and air quality communities. There is a broad range of acceptable tools from relatively simple methods used by local fire managers for estimating likely impacts on air quality standards (for example, SASEM: Riebau and others 1988; and VSMOKE: Lavdas 1996), to complex terrain and regional-scale models that incorporate atmospheric chemistry to assess impacts on regional haze (for example, Calpuff: Scire and others 2000a, and Models-3: Byun and Ching 1999).

The tremendous growth in model application places increasingly greater demands on the user, requiring access to detailed fuel characteristics, fuel consumption, ignition pattern, fire behavior, and meteorological inputs. Also needed is the ability to interpret the complex smoke dispersion model outputs.

In this section we describe such processes of heat release, plume rise and buoyancy, advection and diffusion, scavenging, and chemical transformations.

Heat Release

The consumption of biomass produces thermal energy, and this energy creates buoyancy to lift smoke particles and other pollutants above the fire. Heat release rate is the amount of thermal energy generated per unit of time. Total heat release from a fire or class of fires is a function of the heat content of the biomass, fuel consumed, ignition method and pattern, and area burned.

The early work of Anderson (1969) and Rothermel (1972) created fundamental equations for combustion energy in a variety of fuelbeds. Sandberg and Peterson (1984) adapted the combustion equations to model the temporal change in energy during flaming and smoldering combustion (Emission Production Model, EPMv.1.02). Currently, EPM provides heat release rates for most biomass smoke dispersion models (Harms and others 1997; Harrison 1995; Lavdas 1996; Sestak and Riebau 1988; Scire and others 2000a) and has been used to estimate the change in global biomass emissions patterns due to changes in land use (Ferguson and others 2000). The model, however, requires a constant rate of ignition with constant slope and wind. Such homogeneous conditions may be approximated during prescribed fires that are ignited with a deliberate pattern of drip torches or airborne incendiaries, or during portions of wildfires that experience relatively constant spread rates, both over fuelbed strata that retain a relatively consistent spatial and compositional pattern. To use EPM effectively for modeling source strength, the fire area and ignition duration are broken into space and time segments that meet the steady-state criteria.

Albini and others (1995), Albini and Reinhardt (1995), and Albini and Reinhardt (1997) do not explicitly derive temporal changes in combustion energy in their model, BurnUp, but they do assign source heat in steps of flaming and smoldering that are estimated from total fuel consumption. They have linked their model with the fire spread model, FARSITE (Finney 1998), which allows ignition rates and subsequent heat-release rates to vary over the landscape. The coupled system is computationally expensive and not yet associated with a plume rise component but may offer a reasonable approximation of the temporal and spatial varying emission rates of fires.

Plume Rise and Buoyancy

Heat, particle, and gas emissions from fires vary in time and space, causing unique patterns of convection and resulting plume rise. This plume rise is a function of free convection in the atmosphere, which is caused by density differences within the fluid. As a fire heats and expands air near the ground, large density differences between the heated volume and the surrounding air mass are created, causing the heated parcel to rise. The potential height of the resulting plume depends on the heat energy of the source and rise velocity, which is affected by the exchange and conservation of mass, radiant heat loss, the buoyancy force, and turbulent mixing with the ambient air.

Hot, flaming fires can develop central convective columns with counter-rotating vortices that involve massive entrainment of the surrounding air mass (Clark and others 1996; Haines and Smith 1987; Haines and Updike 1971). This stage of fire can produce fast-rising plumes and turbulent downdrafts, carrying sparks that ignite new fires. Cumulonimbus clouds often develop with accompanying lightning and rain. Dynamic plume rise brings gas and particles high into the atmosphere where strong winds can disperse the smoke hundreds to thousands of kilometers. As high intensity fires cool, however, the central column often collapses, creating numerous small convective cells that are less dynamic but equally active in carrying smoke into the atmosphere. Smoldering fires often create plumes that are neutrally buoyant, limiting widespread dispersion but allowing surface winds to dominate smoke trajectories. This can lead to accumulations of smoke in valleys and basins at night.

Because plume rise can eventually result in widespread dispersion, plume rise calculations are essential for determining the height above ground from which plume dispersion is initiated. Uncertainties in such calculations can result in inaccurate predictions of plume transport and downwind smoke impacts. Given the pressing need to predict the impact of plumes from fires, the need for improved plume rise calculations is apparent.

The basic mechanisms and algorithms used to describe plume rise and buoyancy were developed in the mid-1960s by Briggs (1969) for industrial, ducted emissions. These methods are still used today to estimate the plume rise and buoyancy of fires in spite of the significant differences in characteristics between ducted emissions and prescribed and wildland fires:

- Heat released from ducted sources is precisely known and usually emitted at relatively constant rates during a single phase of combustion. Heat released from fires is a function of fuel loading, fuel conditions, and ignition method through several phases of combustion (pre-ignition, flaming, smoldering, and residual), which create highly variable magnitudes and rates of heat release.
- Nearly all of the energy generated at the source of a ducted plume is transmitted to convection energy. In open burning, however, significant amounts of energy are lost by conduction and radiation, reducing the amount of available energy for convection.
- Plumes from ducted sources create single convective columns, but low intensity understory burning that occurs over broad areas does not develop a cohesive plume.

To improve plume rise predictions, emission production models need to do a better job of characterizing the spatial and temporal pattern of heat release from fires, and plume rise models need to be improved to account for the energy lost from the convective system through radiation and turbulent mixing. While models such as EPM and Burnup described in the previous section simulate variable rates of heat release from fires, both models use general estimates of spatial distributions of fuel, including structure, composition, and moisture content. Also, significant elements of fires that influence convective energy — such as the distribution of naturally piled fuel (“jackpots”), amount and density of rotten fuel and duff, and release of water vapor — are not adequately captured.

Rough approximations on the proportion of energy available for convection were made more than 40 years ago (Brown and Davis 1959). Despite efforts to improve plume rise calculations by removing the density difference assumption (Scire and others 2000a), they still are in use today.

Low intensity fires that typically do not have a cohesive convective column must be treated, from a modeling perspective, as an area source in Eulerian grid models. In Lagrangian dispersion models, there is currently no valid means of calculating plume rise from unconsolidated convection. Eulerian coordinates (used by box and grid models) are coordinate systems that are fixed in space and time, and there is no attempt to identify individual particles or parcels from one time to the next. Lagrangian models (bell-shape or Gaussian distribution pattern, often applied to plume and puff models) are used to show concentrations crosswind of the plume.

Another complication for modeling is that once plumes from fires enter the atmosphere, their fluctuating convection dynamics make them more susceptible to erratic behavior than well-mannered industrial stacks. For example, different parts of a plume can be carried to different heights in the atmosphere at the same time. This causes unusual splitting patterns if there is a notable wind shear between lofted elevations, causing different portions of the plume to be transported in different directions. Therefore, predictions of the plume’s impact on visibility and air quality under these conditions become highly uncertain (Walcek 2002). Even when the behavior of plumes from fires resembles that of stack plumes, the varying and widely distributed locations of wildland sources prevent consistent study. For example, down-wash of plumes has been observed from ducted (stack) emissions after an inversion breaks up — conditions that are common at the end of an onshore breeze if the plume is above the inversion at its source (de Nevers 2000; Venkatram 1988) or if horizontal stratification in the lower atmosphere is disrupted by mountains (de Nevers 2000).

These characteristics of plumes from fire are strikingly different than those of ducted industrial emissions yet little research has been done on this topic in the past several decades.

Advection and Diffusion

In most existing models, the horizontal advection of smoke and its diffusion (lateral and vertical spread) are assumed to be controlled mainly by wind, and the formation and dissipation of atmospheric eddies. These elements are greatly simplified by assuming constant wind (at least for an hourly time step) in some cases (such as VSMOKE and SASEM), and a Gaussian dispersion is nearly always imposed. Perhaps the most critical issues are the constantly changing nature of the plume due to scavenging, chemical transformation, and changing convection dynamics that affect plume transport.

Many photochemical and dispersion models depend on gridded meteorological inputs. Unfortunately, numerical formulations of dynamic meteorological models (for example, MM5: Grell and others 1995; RAMS: Pielke and others 1992) do not adequately conserve several important scalar quantities (Byun 1999a,1999b). Therefore, modelers often introduce mass-conserving interpolations. For example, Models-3/CMAQ (Byun and Ching 1999) uses the MCIP scheme (Byun and others 1999), Calpuff (Scire and others 2000a) employs CALMET (Scire and others 2000b), and TSARS+ (Hummel and Rafsnider 1995) is linked with NUATMOS (Ross and others 1988). Driving a photochemical or dispersion model without these mass-conserving schemes will produce inaccurate results, especially near the ground surface.

Scavenging

Smoke particles by nature of their small size provide efficient cloud condensation nuclei. This allows cloud droplets to condense around fine particles, called nucleation scavenging. Scavenging within a cloud also can occur as particles impinge on cloud droplets through Brownian diffusion, inertial impaction, or collision by electrical, thermal, or pressure-gradient forces (Jennings 1998). Cloud droplets eventually coalesce into sizes large enough to precipitate out, thus removing smoke aerosols from the atmosphere. While interstitial cloud scavenging, especially nucleation scavenging, is thought to dominate the pollution removal process, particles also may be removed by impacting raindrops below a cloud. Jennings (1998) reviews several theories on pollution scavenging but contends that there is little experimental evidence to support such theories.

The size and chemical structure of particles determine their efficiency in nucleation or other scavenging mechanisms. While the chemical composition of smoke is reasonably well known (see chapter 6), distributions of particle size from fire are not. The few airborne measurements (Hobbs and others 1996; Martins and others 1996; Radke and others 1990) do not distinguish fire characteristics or combustion dynamics, which play important roles in the range of particle sizes emitted from a fire. Therefore, the efficiency of scavenging biomass smoke particles out of the atmosphere by cloud droplets, rain, or other mechanism has not been quantified.

Chemical Transformations

Chemical transformations provide another mechanism for changing particle and gas concentrations within a plume. Chemical transformation in the plume can be important in regional-scale modeling programs where sulfate chemistry and ozone formation are of interest (see chapter 6). Oxidation within the smoke plume causes a loss of electrons during chemical transformation processes, which increases polarity of a molecule and improves its water solubility (Schroeder and Lane 1988). This improves scavenging mechanisms by cloud and rain droplets. Chemical transformation rates depend on complex interactions between catalysts and environmental conditions such as turbulent mixing rates.

Transport and Dispersion Models

Trajectories show the path of air parcels along a streamline in the atmosphere. Their simplicity allows trajectory methods to be used as a diagnostic tool for identifying the origin of air parcels from a potential receptor. This commonly is called a backward trajectory or back trajectory analysis. Because these models integrate over time the position of a parcel of air that is transported by wind, their accuracy is limited by the grid resolution of the model. Also, the flow path of a single parcel may have little relation to an actual plume dispersion pattern.

Current models to predict trajectory or air quality impacts from fires are inadequate in coverage and are incomplete in scope (Sandberg and others 1999). But because of new interest in modeling emissions on a regional scale, land managers need transport and dispersion models that include all fire and fuel types as well as multiple sources. Such models need to be linked to other systems that track fire activity and behavior as well as provide for variable scaling to fit the area of interest. At the operational level, models that support real-time decisionmaking during fire operations in both wildland fire situation analysis and go/no-go decision making are also needed (Breyfogle and Ferguson 1996). Transport and dispersion models fall into four major categories. These categories include plume, puff, particle, and grid.

Plume Models

One of the simplest ways of estimating smoke concentrations is to assume that plumes diffuse in a Gaussian pattern along the centerline of a steady wind trajectory. Plume models usually assume steady-state conditions during the life of the plume, which means relatively constant emission rates, wind speed, and wind direction. For this reason, they can be used only to estimate concentrations relatively near the source or for a short duration. Their steady-state approximation also restricts plume models to conditions that do not include the influence of topography or significant changes in land use, such as flow from a forest to grassland or across a land-water boundary.

Gaussian plume models have a great benefit in places and circumstances that restrict the amount of available input data. They can be run fast and have simple but realistic output that can be easily interpreted. Many regulatory guidelines from the EPA are based on Gaussian plume models.

Plume models typically are in Lagrangian coordinates that follow particles or parcels as they move, assigning the positions in space of a particle or parcel at some arbitrarily selected moment. (Lagrangian coordinates are used by plume, puff, and particle models.) Examples adapted for wildland biomass smoke include VSMOKE (Harms and others 1997; Lavdas 1996) and SASEM (Riebau and others 1988; Sestak and Riebau 1988). Both models follow regulatory guidelines in their development and offer a simple screening tool for examining potential concentrations at receptor locations from straight-line trajectories relatively near the source. However, SASEM directly compares downwind concentrations with ambient standards and calculates visibility impairment in a simple manner. It is also used as a State regulatory model in Wyoming, Colorado, New Mexico, and Arizona, and has been recommended for use by the EPA.

Plume rise models developed for other applications might be useful if adapted to fire environments. For example, ALOFT-FT (A Large Outdoor Fire Plume Trajectory Model - Flat Terrain), developed for oil-spill fires (Walton and others 1996), is a computer-based model to predict the downwind distribution of smoke particulate and combustion products from large outdoor fires. It solves the fundamental fluid dynamic equations for the smoke plume and its surroundings with flat terrain. The program contains a graphical user interface for input and output, and a database of fuel and smoke emission parameters that can be modified by the user. The output can be displayed as downwind, crosswind, and vertical smoke concentration contours.

Puff Models

Instead of describing smoke concentrations as a steadily growing plume, puff models characterize the source as individual puffs being released over time. Each puff expands in space in response to the turbulent atmosphere, which usually is approximated as a Gaussian dispersion pattern. Puffs move through the atmosphere according to the trajectory of their center position. Because puffs grow and move independently of each other, tortuous plume patterns in response to changing winds, varying topography, or alternating source strengths can be simulated with some accuracy.

Some models allow puffs to expand, split, compact, and coalesce (Hysplit: Draxler and Hess 1998; Calpuff: Scire and others 2000a) while others retain coherent puffs with constantly expanding volumes (NFSpuff: Harrison 1995). In either case, the variability of puff generation, movement, and dispersion does not restrict the time or distance with which a plume can be modeled. Most puff models are computed in Lagrangian coordinates that allow accurate location of specific concentrations at any time.

Particle Models

In a particle model, the source is simulated by the release of many particles over the duration of the burn. The trajectory of each particle is determined as well as a random component that mimics the effect of atmospheric turbulence. This allows a cluster of particles to expand in space according to the patterns of atmospheric turbulence rather than following a parameterized spatial distribution pattern, such as common Gaussian approximations. Therefore, particle models tend to be the most accurate way of simulating concentrations at any point in time. Because of their numerical complexity, however, particle models usually are restricted to modeling individual point sources with simple chemistry or sources that have critical components such as toxins that must be tracked precisely. Particle models use Lagrangian coordinates for accurate depiction of place of each time of particle movement (for example, Hysplit: Draxler and Hess 1998; PB-Piedmont: Achtemeier 1994, 2000).

Grid Models

Grid models use Eulerian coordinates, disperse pollutants uniformly within a cell, and transport them to adjacent cells. The simplicity of advection and diffusion in a grid model allows these models to more accurately simulate other characteristics of the pollution, such as complex chemical or thermal interactions, and to be used over large domains with multiple sources. This is why grid models commonly are used for estimating regional haze and ozone and are often called Eulerian photochemical models. Much of the future work on fire impact assessment and planning at regional to national scales will be done by using grid models.

Because of their nature, grid models are not used to define accurate timing or locations of pollutant concentrations from individual plumes, only concentrations that fill each cell. This means that sources small relative to the grid size, which create individual plumes, will introduce unrealistic concentrations in places that are outside of the actual plume. Ways of approximating plume position and its related chemical stage include nesting grids to finer and finer spatial resolutions around sources of interest (Chang and others 1993; Odman and Russell 1991), establishing nonuniform grids (Mathur and others 1992), and creating "plume-in-grid" approximations (Byun and Ching 1999; Kumar and Russell 1996; Morris and others 1992; Myer and others 1996; Seigneur and others 1983).

Many regional haze assessments use the Regulatory Modeling System for Aerosols and Acid Deposition (REMSAD) (Systems Applications International 2002). This model was adapted from the urban airshed model–variable grid (UAMV) by removing its plume-in-grid feature and parameterizing explicit chemistry to improve computational efficiency. REMSAD incorporates both atmospheric chemistry and deposition processes to simulate sulfate, nitrate, and organic carbon particle formation and scavenging. As such, it is quite useful for simulations over large regions.

The Models-3/ CMAQ modeling system is designed to integrate the best available modules for simulating the evolution and dispersion of multiple pollutants at a variety of scales (Byun and Ching 1999). It includes chemical transformations of ozone and ozone precursors, transport and concentrations of fine particles and toxics, acid deposition, and visibility degradation.

At the other end of the grid modeling spectra are simple box models that describe pollution characteristics of a small area of interest. Box models instantaneously mix pollutants within a confined area, such as a valley. This type of model usually is restricted to weather conditions that include low wind speeds and a strong temperature inversion that confines the mixing height to within valley walls (Lavdas 1982; Sestak and others 1988). The valley walls, valley bottom, and top of the inversion layer define the box edges. The end segments of each box typically coincide with terrain features of the valley, such as a turn or sudden elevation change. Flow is assumed to be down-valley, and smoke is assumed to instantaneously fill each box segment. Few box models include the complex chemical or particle interactions that are inherent in larger grid models.

Model Application

Modeling of the transport and dispersion of industrial stack plumes has occurred for decades, prompting a variety of techniques. But application to fires is much more limited (Breyfogle and Ferguson 1996). Part of the reason for this is that source strength from undulating and meandering fires is so difficult to simulate accurately. Therefore, applications have been appropriate mainly for relatively homogeneous fuelbeds and steady state burn conditions. This has restricted most transport and dispersion modeling to fires on a local scale and to those started in harvest residue from land clearing operations where fuels are scattered uniformly over the landscape or collected into piles (Hardy and others 1993; Hummel and Rafsnider 1995; Lavdas 1996; Sestak and Riebau 1988). Global-scale modeling also has taken place where fuelbed and ignition patterns are assumed to be approximately steady state in relation to the grid size (Kasischke and Stocks 2000; Levin 1996).

Gaussian plume models (Harms and others 1997; Lavdas 1996; Sestak and Riebau 1988; Southern Forest Fire Laboratory Personnel 1976) are useful for places with relatively flat terrain, for circumstances when input data are scarce, and for evaluating surface concentrations relatively near the source. These models typically require only an estimate of atmospheric stability, trajectory wind speed and direction, and emission rates. Fires are modeled independently. Therefore, accumulations of smoke from multiple fires are ignored. Some Western States require SASEM modeling of prescribed burns before they can be permitted (Battye and Battye 2002).

Puff models (Draxler and Hess 1998; Harrison 1995; Hummel and Rafsnider 1995; Scire and others 2000a) are needed when simulating long-range transport, or transport that occurs during changeable environmental conditions such as influences from complex terrain or variable weather. NFSpuff has an easy user interface, but because of its internal terrain data files it is restricted to applications in the Western States, excluding Alaska (Harrison 1995). Hysplit (Draxler and Hess 1998) currently is programmed to accept only 16 individual sources and assumes a constant rate of emissions with no plume rise. Hysplit (Draxler and Hess 1998) and Calpuff (Scire and others 2000a) both include simple chemistry. NFSpuff is the most commonly used puff model for prescribed fire planning (Dull and others 1998). All three models are linked to the MM5 meteorological model (Grell and others 1995). NFSpuff can function with a simple trajectory wind, and Hysplit and Calpuff can accept other gridded weather input data.

Particle models are used in coupled fire-atmosphere modeling (Reisner and others 2000) and for tracking critical signature elements (Achtemeier 1994, 2000; Draxler and Hess 1998). The sophistication of these types of models and their computational requirements, however, has thus far limited their application to research development or individual case studies.

Eulerian photochemical grid models are highly useful in estimating smoke concentrations from many sources over large domains. In addition, their ability to model secondary chemical reactions and transformations is needed for determining ozone concentrations and regional haze conditions. Regional planning organizations such as the Western Regional Air Partnership (WRAP), are evaluating the photochemical models Models-3/CMAQ (Byun and Ching 1999) and REMSAD (Systems Applications International 2002) for use in guiding State implementation plans (SIPs) and Tribal implementation plans (TIPs).

Additional work is needed to fill critical gaps in the modeling systems identified above. As the need for

Chapter 6: Atmospheric and Plume Chemistry

Traditionally, ozone and secondary aerosol precursors have been discussed within the context of urban smog caused by auto exhaust and reactive organic compounds emitted from industrial facilities. But the same pollutant and tropospheric chemical reactions occur in both urban settings and in rural areas where wildfire smoke may be an important if not dominant source of ozone precursor emissions. In these situations, emissions from fire may play an important role in ozone formation as well as nitrate and, indirectly, sulfate aerosol formation, which results in visibility impairment and increased PM2.5 concentrations.

At present, there is an urgent need to understand the impact of fire emissions on emerging visibility and ambient air standards as they relate to fire planning at the strategic, programmatic, and operational scales (Fox and Riebau 2000; Sandberg and others 1999). Chemical processes that occur in plumes from fires, directly or indirectly, touch on a number of these issues and are critical to the development of a regional model that will be used to assess the impact of fire on air quality.

Because of the Environmental Protection Agency's (EPA) pressing regulatory need to assess inter-State ozone transport and sources of precursor emissions, a new regional-scale mechanistic model called Models-3/CMAQ (Byun and Ching 1999) is being used by the Ozone Transport Commission (OTC) region of North-eastern and Mid-Western States, and the Western Regional Air Partnership (WRAP). Future applications will likely involve regional haze modeling in other areas of the country. Oxides of nitrogen (NO_X) and volatile organic compounds (VOCs) emissions from fire in the OTC have not previously been considered significant, but the new model photochemistry module requires that precursor emissions be included for all sources. As Models-3/CMAQ develops, NO_X and VOC emissions from fire will be included in ozone and secondary modeling.

Ozone Formation in Plumes

Field observations of ozone formation in smoke plumes from fires date back nearly 25 years when aircraft measurements detected elevated ozone at the edge of forest fire smoke plumes far downwind (Stith and others 1981). More recent observations (Wotawa and Trainer 2000) suggest that high concentrations of ozone are found in forest fire plumes that are transported great distances and across international boundaries. Measurements made during EPA's 1995 Southern Oxidant Study indicate that Canadian forest fires changed the photochemical properties of air masses over Tennessee on days with strong fire influence. Regional background ozone levels were elevated by 10 to 20 ppb on fire impact days as compared with nonimpact days during the study. Aircraft measurements found that, although forest fire plumes were always well defined with respect to carbon monoxide, they gradually lost their definition with respect to ozone after being mixed into the boundary layer. The amount of ozone transported to the surface measurement sites was found to depend upon where and when the plumes reached the ground. Elevated plumes were always marked by enhanced ozone concentrations, at times reaching values of 80 to 100 parts per billion (ppb) above tropospheric background.

Stith and others (1981) mapped ozone mixing ratios in an isolated, fresh, biomass-burning plume. At the source, or near the bottom, of the horizontally drifting plume they measured low or negative changes in ozone values, which they attributed to titration by NO and low ultraviolet (UV) intensity. Near the top of the plume, 10 km downwind, and in smoke less than 1 hour old, they measured change in ozone values as high as 44 parts per billion by volume (ppbv). Greater changes in ozone were positively correlated with high UV. Thus the initial destruction of ozone by reactive species in the plume followed by its gradual formation was documented.

A new and potentially useful tool for assessing impacts of long-range plume transport is based on the concept of using $\Delta O_3/\Delta CO$ (excess O_3 over excess CO) as a "photochemical clock" to denote the degree of photochemical processing in a polluted air mass by using carbon monoxide as a stable plume signature. As the plume disperses, its volume expands and absolute values of ozone can drop even though net production of ozone is still occurring. The $\Delta O_3/\Delta CO$ normalizes for plume expansion and is a useful measure of net ozone production. In the course of atmospheric chemistry research, numerous observations of $\Delta O_3/\Delta CO$ ratios have been made in biomass burning haze layers. Unfortunately, the observations represent haze of various ages and uncertain origin. In haze layers 1 to 2 days old, changes in the $\Delta O_3/\Delta CO$ ratios of 0.04 to 0.18 were measured over Alaska (Wofsy and others 1992) and ratios of 0.1 to 0.2 were measured over Eastern Canada (Mauzerall and others 1996). High ratios, up to 0.88, were measured at the top of haze layers that had aged about 10 days in the tropics (Andreae and others 1994).

In 1997, airborne Fourier transform infrared spectroscopy (FTIR) measurements in large isolated biomass burning plumes in Alaska revealed new details of downwind chemistry. Downwind smoke samples that had aged in the upper part of one plume for 2.2 ± 1 hours had $\Delta O_3/\Delta CO$ ratios of 7.9 ± 2.4 percent, resulting from initial, absolute ozone formation rates of about 50 ppb/hr. Downwind samples obtained well inside another plume, and of similar age, did not have detectable ΔO_3, but did have $\Delta NH_3/\Delta CO$ ratios about one-third of the initial value. $\Delta HCOOH/\Delta CO$ (formic acid) and $\Delta CH_3COOH/\Delta CO$ (acetic acid) usually increased about a factor of 2 over the same time scale in samples from both plumes. NO_X was below the detection limit in all the downwind samples. These data provided the first precise in-plume measurements of the rate of O_3/CO increase and suggested that this rate depended on relative position in the plume. The apparently rapid disappearance of NO_X is consistent with the similar early observation, and the drop in NH_3 was consistent with a reaction with HNO_3 to form ammonium nitrate, which is a NO_X sink. Secondary sources of formic acid relevant to polluted air have been described (Finlayson-Pitts and Pitts 1986). Jacob and others (1992, 1996) discussed several gas-phase sources of acetic acid that could occur in biomass burning plumes. These experiments provide the first experimental indication of the approximate time scale of secondary organic acid production in actual plumes.

A large number of photochemical modeling studies of biomass burning plumes have been published (Chatfield and Delaney 1990; Chatfield and others 1996; Crutzen and Carmichael 1993; Fishman and others 1991; Jacob and others 1992, 1996; Koppmann and others 1997; Lee and others 1998; Lelieveld and others 1997; Mauzerall and others 1998; Olson and others 1997; Richardson and others 1991; Thompson and others 1996). Nearly all these studies conclude that the net production of ozone occurs either in the original plume, or as a result of the plume mixing with the regional atmosphere. Several studies have shown a strong dependence of the final modeled results on the details of the post-emission-processing scenario such as the timing between production of the emissions and their convection to the free troposphere (Chatfield and Delaney 1990; Jacob and others 1996; Lelieveld and others 1997; Pickering and others 1992; Thompson and others 1996).

Factors Affecting Plume Chemistry

The specific chemical composition of the plume depends on many factors: the details of post-emission atmospheric reactions including dilution rates, photolysis rates, position within the plume, altitude, and smoke temperature, which varies by time of day and combustion stage. Equally important is the chemistry of the downwind air that mixes with the plume, which could be clean air or contain aged plumes from urban areas or other fires. In addition, the physical aspects of the plume mixing are important. For example, at the relatively low temperatures typical of higher altitudes in the troposphere, peroxyacetyl nitrate (PAN) is a stable molecule, which can be transported. At lower altitudes, PAN can thermally decompose and rerelease NO_X. Nitric acid (HNO_3) can also be an important, transportable reservoir species for NO_X at high altitudes but for a different reason. HNO_3 has a narrower absorption cross-section at lower temperatures and therefore is less susceptible to photolysis. The rate of bimolecular reactions among smoke components usually decreases with temperature (thus typically with altitude or at night). Reaction rates depend even more strongly on the dilution rate, at least initially. Dilution by a factor of 2 will decrease a bimolecular reaction rate by a factor of 4.

Emission Factors for Reactive Species

Emission factors for hydrogen oxide (HO_X, a collective term for OH and HO_2) precursors, NH_3, and NO_X have been estimated with the Missoula, MT, open-path spectroscopic system (Yokelson and others 1997). These experiments reveal that smoke contains high levels of oxygenated organic compounds, methanol (CH_3OH), acetic acid (CH_3COOH), and formaldehyde (HCHO). These compounds typically oxidize or photolyze within hours in a smoke plume to release HO_X that is important in sulfate aerosol formation processes. Under clear-sky conditions typical for noon on July 1 at 40°N latitude, the formaldehyde photolysis lifetime is about 3.8 hours (Yokelson and others 1997). Since the HCHO/CO source ratio for fires is typically near 2 percent, this process clearly injects large quantities of HO_2 into fresh plumes (Yokelson and others 1997). HO_X emissions from fire may become a critical input to regional haze models that simulate secondary sulfate formation processes.

The H_2O_2 is soluble in cloud droplets where it would play a major role influencing reaction rates during aqueous-phase sulfate formation chemistry (NRC 1993).

Particle Formation in Plumes

A number of processes are important in plume particle formation and growth. Many of these processes involve interaction with the trace gases in a plume originating from nucleation in which two gases react to form a solid nucleus for subsequent particle growth. An example of nucleation is the reaction of ammonia and nitric acid. In addition, condensation can create new particles when gases cool or through particle growth when a trace gas collides with and condenses on an existing particle. The second condensation process is quite common because biomass burning aerosol is hydrated. Soluble nucleilike ammonium nitrate promotes this process. There is a little evidence that organic gases also condense on particles. Nucleation and condensation are both examples of trace-gas-to-particle conversion, which will increase the mass of particles in a plume, decrease the concentration of certain trace gases in the plume, and, in the case of condensation, contribute to an increase in average particle diameter. Andreae and others (1988) measured particle-NH_4^+/CO_2 ratios of 0.7 to 1.5 percent in slightly aged biomass burning plumes. Measurements of NH_3/CO in fresh smoke are typically near 2 percent. Thus, there is probably rapid conversion of gas-phase NH_3 to particle NH_4^+ either through nucleation or dissolution in the surface water of other hydrated particles.

Coagulation is when two particles collide and combine. This increases the average particle diameter, reduces particle number, and does not effect total particle mass. Coagulation probably contributes to the increase in average particle diameter that occurs downwind from fires (Reid and others 1998).

At any given point in its evolution a particle may impact the trace gas chemistry in a smoke plume. For instance, it is known that NO_2 reacts on the surface of soot particles to yield gas phase HONO. This and other heterogeneous reactions such as ozone destruction may occur on smoke aerosol. Some recent research suggests that oxygenated organic compounds emitted from fires could also be important in heterogeneous processes. Hobbs and Radke (1969), Desalmand and others (1985), Andreae and others (1988), and Roger and others (1991) found that a high percentage (25 to 100 percent) of fire aerosol particles from fires could be active as condensation nuclei (CCN). Radke and others (1990) observed that cumulus clouds greater than 2 km in depth scavenged 40 to 80 percent of smoke particles. The high concentrations of CCN in smoke plumes can contribute to the formation of clouds with smaller than "normal" cloud droplet size distributions. This type of cloud is more reflective to incoming solar radiation and less likely to form precipitation. Some work suggests that absorbing aerosol can reduce cloud formation. Finally, clouds can evaporate and leave behind chemically altered particles.

All of these mechanisms alter both the chemical nature and number of particles contained within smoke plumes from fires. In addition, reactive species emitted from fires (see previous section) may alter the conversion rate of gaseous precursors of secondary sulfate and nitrate particles, affecting regional haze modeling results.

Although the regulatory implications of reactive species emissions from fire are yet to be determined, much more attention to these issues will occur once fire is including in regional haze and ozone modeling efforts.

Notes

Chapter 7: Estimating the Air Quality Impacts of Fire

State-of-the-science methods used to determine the impact of fire on air quality and visibility include: (1) emission inventories; (2) air quality monitoring instruments to measure smoke concentrations in real-time; and (3) filter-based monitoring techniques and receptor-oriented methods that quantify wildfire smoke contribution to air pollution based on the chemical characteristics of smoke particles or the spatial and/or temporal variability. Fire also contributes to ground-level ozone. These topics have become increasingly important to both air quality regulators and land managers as efforts to identify, or apportion, the contributions that fire makes to particulate air pollution, regional haze, and ground-level ozone come under increased scrutiny.

Because the health effects of air pollution are so difficult to measure in the broad population, there has been little effort to regulate or manage those effects directly. Many smoke management decisions are made on the basis of nuisance complaints as an indicator, rather than on quantitative measurements of impacts to health and welfare. Close to the source, efforts are being made to keep the exposure of firefighters to hazardous air pollutants within the standards set by the Occupational Safety and Health Administration. Hazard assessment describes the nature, concentration, and duration of pollutants. Exposure assessment quantifies the population exposed and the degree of exposure. Risk assessment describes the probable result for a population from all exposures. Integrated health risk assessments and economic assessments are still rare.

Modeling and data systems are needed to predict, measure, and monitor the ultimate effects of air pollution from fires on human or ecosystem health, on the economy, and on the comfortable enjoyment of life and property. Risk assessment methods are needed to compare these effects with those from other sources.

Emission Inventories

An emission inventory is an estimate of the mass of emissions by class of activity within a specified geographic area in a specified amount of time. Usually, an inventory is compiled by multiplying the appropriate emission factor (see chapter 4) by the estimated level of activity (in other words, tons of fuel consumed).

Development of emission inventory methods for fires was recently reviewed in detail by Battye and Battye (2002). The report considers prior attempts at emission inventory, describes approaches to estimating emissions from fires, and reviews the scientific information available as components of an inventory. The report also reviews emission reduction strategies and smoke management techniques.

An emission inventory provides an understanding of the relative burden on the air resource from particular air pollution source categories. Emission inventories help explain the contribution of source categories to pollution events, provide background information for air resource management, provide the means to verify progress toward emission reduction goals, and provide a scientific basis for State air program development. An accurate emission inventory provides a measured, rather than perceived, estimate of pollutant production as the basis for regulation, management action, and program compliance. Emission inventories should include all important source categories including mobile, area, and stationary, and the inventories are not complete unless difficult-to-quantify sources such as agricultural burning, backyard burning, rangeland burning, and wildland and prescribed burning are addressed. Emission inventories are a basic requirement of State air resource management programs and are a required element of State implementation plans (SIPs). Emission inventories are also compiled annually at the national level and for specific geographic regions (sub-State, multi-State, or multi-jurisdiction) to address a particular regional air quality issue.

The science necessary to accurately estimate emissions from prescribed burning is quite good for most fuel types in the United States if good quality information about several critical variables is known. Area burned, fuel type, fuel loading, fuel arrangement, fuel consumption, and emission factors are all needed to accurately estimate emissions. Some of these require onsite reporting for reasonable accuracy including area burned, fuel type, and fuel arrangement. Other factors can be defaulted or estimated with reasonable accuracy if some other information is known. Fuel loading can be defaulted with knowledge of the fuel type and arrangement. Fuel consumption can be calculated with knowledge of the fuel type, fuel loading, and fuel moisture. Emission factor assignment is made with knowledge of the fuel type.

The science of predicting emissions from wildland fire is much weaker than for prescribed fire. In addition, it is generally far more difficult to obtain decent quality information about individual wildland fires.

In most cases, the information gap that makes fire emissions prediction a difficult endeavor is good quality, consistent, and regular reporting of the specific onsite variables needed for emissions estimation. Data collection systems that are supported and utilized by fire managers need to be developed for every State where a reasonable estimate of prescribed fire emissions is desired. Data collection for wildfire emissions estimation will be more difficult because some of the needed information is not currently available in a way that is compatible with emissions estimation requirements. For example, a single wildfire often burns through many different fuel types, but current reporting requirements request the fuel type at the point of ignition. This fuel type may or may not be representative of the majority of acres burned in the wildfire. Also, acres burned in wildland fires may be the area within the fire perimeter rather than the actual acres blackened by fire as is needed for emissions estimation. Similarly, the area reported as burned in prescribed fires is often the area authorized for burning whether or not the entire burn was completed.

State Emission Inventories

High quality Statewide inventories of daily emissions from prescribed fire have been developed by Oregon and Washington since the 1980s (Hardy and others 2001). Eleven other States (Alabama, Alaska, Arizona, California, Colorado, Florida, Idaho, Montana, Nevada, South Carolina, and Utah) estimate annual prescribed fire emissions from records of acreage burned by fuel type and fuel loading at the end of the burning season. Many other States (such as Michigan, New Mexico, and Tennessee) currently have no annual reporting program.

No State has a reporting system for wildland fires that is based on actual, reported data from individual wildland fires events. Any estimate a State may have of wildland fires emissions is based on gross assumptions about fuel loading and consumption, and on an area-burned figure that may systematically overestimate the true value.

Regional Emission Inventories

Several recent regional inventories compiled in support of regional haze program development have shown new approaches to fire emission inventory development.

The Fire Emissions Project (FEP) calculated an emissions inventory for 10 Western States for a current year (1995) using actual reported data, plus two future years (2015 and 2040) using manager projections of fire use. Fourteen vegetative cover types were chosen to characterize the range of species types within the 10-State domain. Within each vegetative cover type, up to three fuel loading categories (high, medium, and low) could be specified by field fire managers. Fuel consumption calculations relied on expert estimates of fuel moisture believed to be most frequently associated with a particular type of burning. Emission factors were assigned based on the vegetative cover type. The FEP inventory was used during the Grand Canyon Visibility Transport Commission (GCVTC) effort to apportion sources of visibility impairment in the Western States.

The GCVTC also sponsored the development of a wildland fire emissions inventory for the period 1986 through 1992. The GCVTC wildfire inventory included only wildland fires greater than 100 acres in size (capturing approximately 98 percent of the acreage burned). The variability of wildland fire emissions, which ranged from 50,000 tons per year of PM2.5 to more than 550,000 tons per year over the 7 years studied, indicates the difficulty in selecting a single 1 year period that is representative of “typical” fire emissions (GCVTC 1996a).

In 1998, analysts at the Forest Service’s Missoula Fire Sciences Laboratory, Rocky Mountain Research Station, used the FEP management strategies with new, additional data to estimate emissions from wildland fires in the Western States (Hardy and others 1998). This inventory of potential emissions used a suite of new or improved spatial data layers, including vegetation/cover type, ownership, fuel and fire characteristics, modeled emissions and heat release rates, and fuels treatment probability distributions. These inventories are included in the Environmental Protection Agency’s (EPA) National Emission Inventory (NEI).

Wildland fire frequency and occurrence are highly variable in time and space (fig. 7-1). The impact of wildland fire smoke on Class I area visibility is also expected to be highly variable from year to year with episodic air quality and visibility impact events that are difficult to predict. Seasonal impacts may be many times higher than annual averages.

National Emission Inventories

National emission inventories for prescribed fire have been compiled and reported by several investigators (Chi and others 1979; Peterson and Ward 1992; Ward and others 1976; Yamate and others 1975). Of these, only the Peterson and Ward inventory of particulate matter and air toxic emissions from prescribed fires during 1989 is still useful today, despite the inconsistencies in the information available to compile the emission estimates. The poor data collection and inconsistent or nonexistent reporting systems in use at the time of the 1989 inventory continue today.

Improving Emission Inventories

Significant barriers to compiling better regional inventories include:

- Varying degrees of availability and number of records describing burning activity over multiple States, multiple agencies, ownerships, and Tribes.
- Lack of a national wildland fuel classification system with spatial attributes.
- Limited and inappropriate modeling of fuel consumption and emission characterization for prescribed burning in natural fuels.

Sandberg and others (1999) describe remedies to overcome some of the limitations of data collection and availability. These remedies are intended to guide

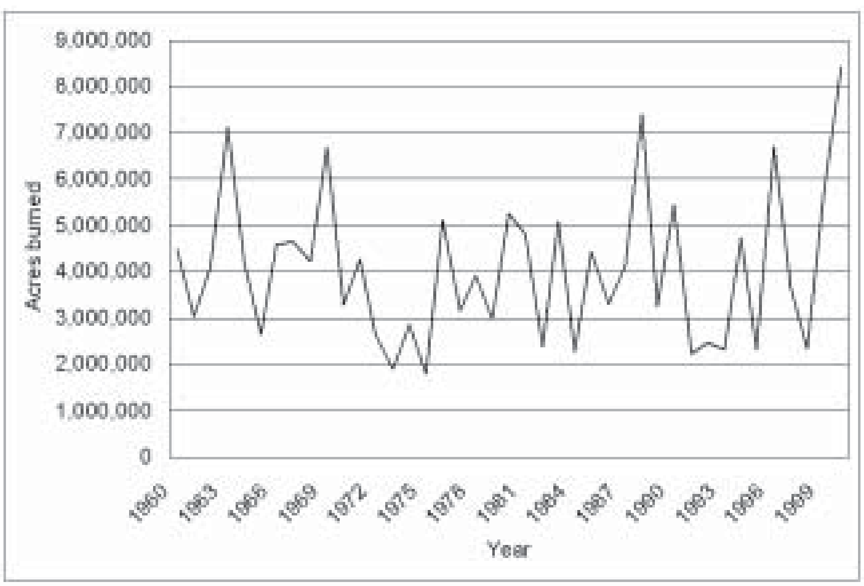

Figure 7-1—Number of acres burned by wildfire between 1960 and 2000 (National Interagency Fire Center 2002).

future inventory development efforts. Significantly, these remedies include adoption of standardized burn reporting protocols to be used by all agencies, Tribes, and ownerships to report daily emissions for each burn, location of the burn, and many other parameters.

The Fuel Analysis, Smoke Tracking, and Report Access Computer System (FASTRACS) is a sophisticated system developed by the Forest Service and Bureau of Land Management in the Pacific Northwest. FASTRACS tracks all the information needed for accurate estimation of emissions from Federal use of prescribed fire in Washington and Oregon including the ability to track use of emission reduction techniques. As long as field fire managers are doing a reasonable job of reporting the information required by FASTRACS, this system provides excellent emissions calculation capabilities and the best data reporting standards in the country. Currently other landowners, such as State and private, are not using FASTRACS in Washington and Oregon although there is an effort under way to bring them into the system. FASTRACS is also being looked at by other regions and may be adopted or emulated across the country. For more information about FASTRACS, see http://www.fs.fed.us/r6/fastracs/index.htm.

Another data reporting system is under development in California. The Prescribed Fire Information Retrieval System (Cal/PFIRS) is a centralized electronic database that allows all users immediate access to detailed information on burns on a day-to-day basis. Cal/PFIRS does not include the kind of detailed reporting of information that could be used to assess the use of emission reduction techniques but does provide a reasonable estimate of the amount of burning taking place. For more information on Cal/PFIRS, see http://www.arb.ca.gov/smp/progdev/techtool/pfirs.htm.

Research since about 1970 has significantly improved the completeness and accuracy of emission inventory techniques. However, the science is being pressed forward because of new demands for regional scale emission transport information needed to assess the impact of wildland smoke on PM2.5 air quality standards and regional haze. Because of new air regulatory demands, emission inventories, when used in concert with regional models, have become an important means of apportioning fire smoke impacts on air resources.

Air Quality Monitoring

Unlike emission inventories, air quality monitors determine actual pollutant loading in the atmosphere and are therefore the most direct measure of air quality on which air regulatory programs are based. Samples of particulate matter in the atmosphere (PM10 or PM2.5, or both) are also used for source apportionment purposes to identify the origin of the aerosols. Monitoring of smoke from fires, however, presents several unusual technical challenges that affect results. These challenges center on the fact that smoke from fires has several unique characteristics.

Current Monitoring Techniques

The three principal methods of measuring air pollution are samplers, optical instruments, and electrochemical devices. Samplers are most common for long-term monitoring. Data from optical meters and electrochemical devices can be stored in a computer or datalogger on site or transmitted from remote locations to provide real-time information.

Samplers—Samplers collect aerosols on a filter or chemical solution. A simple gravimetric measure of mass concentration may be obtained, or different types of filters or solutions can be used, to help define chemical species and particle sizes. For chemical speciation, filters must be sent to a laboratory for analysis. For this reason, sampling information usually is delayed by days to weeks after the sampling period. Active samplers are the most accurate as they use a pump to pass a known volume of air through the collector. Passive samplers are the least expensive, allowing air to reach the collector by some physical process such as diffusion. Tapered Element Oscillation Microscales (TEOMs) are a special class of samplers that provide a gravimetric measure of mass concentration at the studied site without having to transport filters to a laboratory.

All sampling devices lose some degree of semivolatile fine particulates (Eatough and Pang 1999). Positive and negative organic carbon artifacts are just two of several factors that contribute to variability between different colocated instruments. To minimize this variability, consistent sampling methods are used throughout a sampling network to help recognize such artifacts.

The analytical technique used to quantify carbon concentrations from filters also can cause discrepancies between measurements (Chow 2000). For example, the NIOSH 5040 method (Cassinelli and O'Conner 1994) is a thermal-optical transmittance method of speciating total, organic, elemental, and carbonate (inorganic) carbon being adopted by the EPA's PM2.5 program. This method is a departure from the thermal-optical reflectance method that has been used in the IMPROVE program. Recent comparisons between ambient samples have identified differences as great as 17.5 ± 15 percent (EPA 2000a), which can be significant when monitoring for National Ambient Air Quality Standards (NAAQS) violations.

Because filters can become overfull, they must be changed regularly and are not suitable for sites close to fires where particulate concentrations are heavy.

Optical Instruments—Optical instruments use a light source to measure the atmosphere's ability to scatter and absorb light. Common devices are photometers, which measure the intensity of light, and transmissometers, which are photometers used to measure the intensity of distant light. Photometers and transmissometers have a direct relation to visual range. Nephelometers measure the scattering function of particles suspended in air. They can be used to determine the visual range, as well as the size of the suspended particles, by changing the wavelength of the light source. Wavelengths of 400 to 550 nm are common for monitoring smoke from biomass fires, while wavelengths of 880 nm are more common for road dust measurements. Because the instruments have increasing application for both long-term and real-time monitoring of smoke, Trent and others (2000) evaluated the accuracy of several different optical instruments by comparing their output to gravimetric samples.

Investigators have found some problems in field reliability and temperature drift among photometers and nephelometers (Trent and others 1999, 2000). While Davies (2002) recommends a general coefficient for relating scattering coefficient to drift smoke from a DataRAM nephelometer, a precise relation between a nephelometer's measured scattering coefficient and particle concentration depends on the wavelength of the instrument and the particle distribution of the medium, which varies by combustion stage and fuel type.

Electrochemical Devices—Electrochemical devices have been used in industrial applications for many years. Their small size and ability to measure criteria pollutants, such as carbon monoxide, make them suitable for personal monitoring or monitoring in extremely remote locations. Thus, they are gaining value for monitoring wildland smoke impacts. For example, Reinhardt and Ottmar (2000) recommend the use of an electrochemical dosimeter for monitoring exposure levels experienced by wildland fire fighters (Reinhardt and Ottmar 2000).

States, Tribes, and local air agencies use a variety of instruments to monitor long-term and real-time smoke impacts for both NAAQS and visibility to suit their local interests and regulatory needs. The Interagency Monitoring of Protected Visual Environments (IMPROVE) program is one of few nationally coordinated monitoring projects.

IMPROVE was established in 1985 in response to the 1977 amendment of the Clean Air Act requiring monitoring of visibility-related parameters in Class I areas throughout the country (fig. 7-2). The IMPROVE network uses a combination of speciation filters on active samplers to measure physical properties of atmospheric particles (PM2.5 and PM10) that are related to visibility. Many sites also include transmissometers and nephelometers optical devices. Also, cameras are used document the appearance of scenic vistas. Because the samplers collect for 24 hours every 3 days, their information is used for determining long-term trends in visibility. The optical and camera devices can monitor more frequently and can help define short-term or near real-time changes in visibility impact

Source Apportionment

Most air monitoring programs are designed to measure particulate mass loading to provide data for PM10 and PM2.5 NAAQS and visibility. Because these sizes of particles can come from many sources, they are not useful for apportioning to one source or another. While the IMPROVE program provides speciated aerosol data that are helpful in source attribution analysis, the averaging periods of samples and sparse location of sites make IMPROVE measurements difficult to use for source attribution without supplemental measurements or modeling tools.

Wotawa and Trainer (2000) found that 74 percent of the variance in the average afternoon carbon monoxide levels could not be attributed to anthropogenic sources during the 1995 Southern Oxidant Study (Chameides and Cowling 1995). Analysis of weather patterns indicated that transport of wildland fire smoke from Canada could explain the elevated carbon monoxide levels. Also, they discovered a statistically significant relationship between the elevated carbon monoxide and ground-level ozone concentrations.

Characterization of organic carbon compounds found within the organic carbon fraction of fine particulate matter coupled with inclusion of gaseous volatile organic compounds (VOCs) holds substantial promise in advancing the science of source apportionment (Watson 1997). The key to the use of chemical mass balance methods is the acquisition of accurate data describing the chemical composition of both particulate matter and VOCs in the ambient air and in emissions from specific sources. Several organic compounds unique to wood smoke have been identified including retene, levoglucosan, thermally altered resin, and polycyclic aromatic hydrocarbons (PAH) compounds. These compounds are present in appreciable amounts and can be used as signatures for source apportionment if special precautions are taken during sampling to minimize losses (Standley and Simoneit 1987). Inclusion of these aerosol and VOC components in the speciation

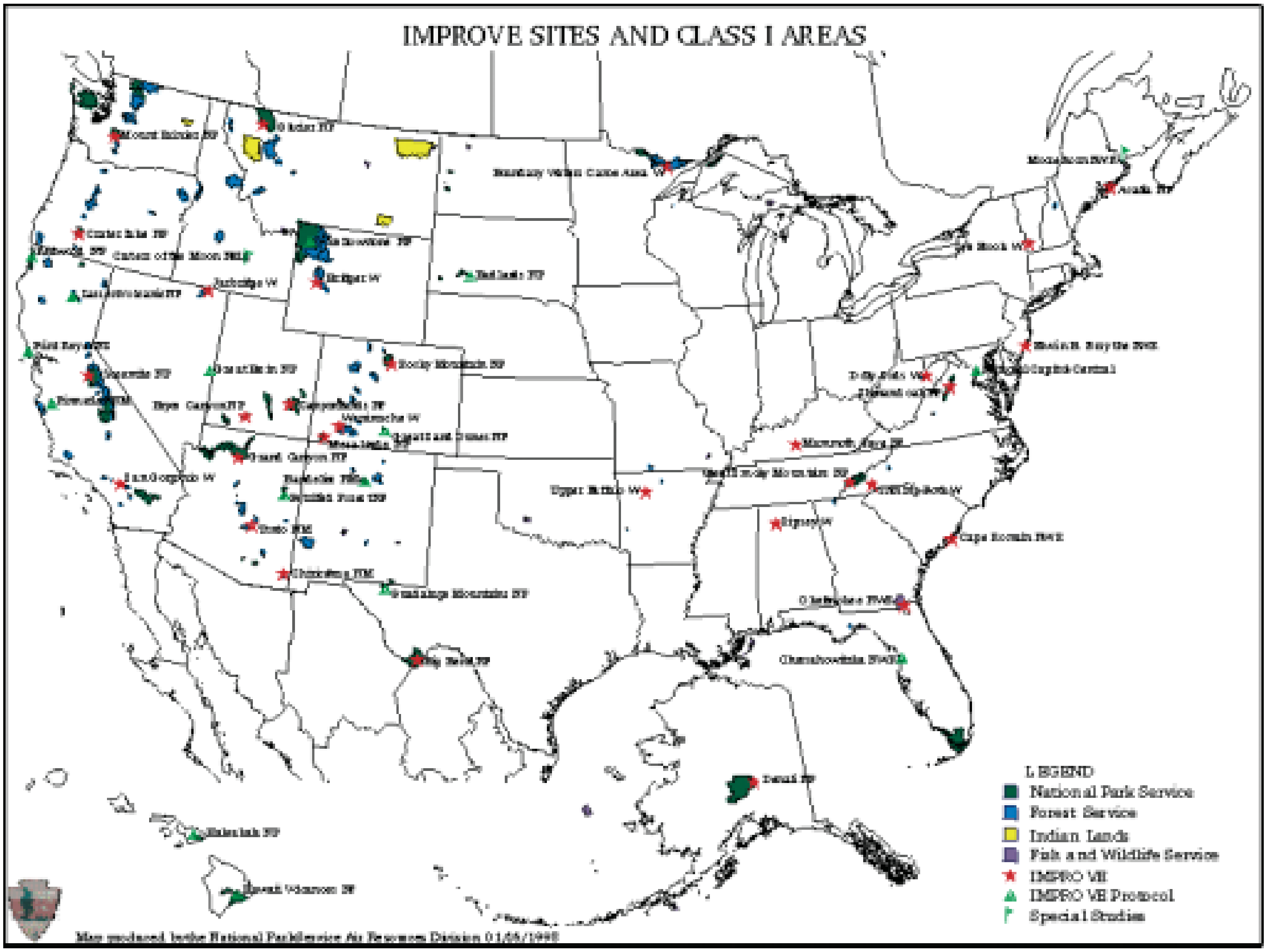

Figure 7-2—IMPROVE monitoring network in 1999 (http://vista.cira.colostate.edu/improve/Overview/IMPROVEProgram.htm).

analysis appears worthwhile but would increase monitoring and sample analysis costs.

Source Apportionment Methods

Apportionment of particulate matter mass to the respective contributing sources is done through both mechanistic models (dispersion models) and receptor-oriented techniques that are based on the characteristics of the particles collected at the receptor. The best approach is through the use of both techniques, applied independently, to develop a "weight of evidence" assessment of source contributions of smoke from fire. A third approach is through the use of visual and photographic systems that can document visibility conditions over time or track a plume from its source to the point of impact within a Class I area.

Receptor-Oriented Approaches

Receptor-oriented approaches range from simple signature applications to complex data analysis techniques that are based on the spatial, temporal, and chemical constituents ("fingerprint") of various sources.

Simple signature applications for smoke from fire are based on chemically distinct emissions from fire. For example, methyl chloride (CH_3Cl) is a gas emitted during wood combustion that has been used in this manner to identify impacts of both residential woodstove smoke and smoke from prescribed fires (Khalil and others 1983).

Speciated Rollback Model —The speciated rollback model (NRC 1993) is a simple hybrid model that uses aerosol data collected at the receptor with emission inventories to estimate source impacts. It is a spatially averaged model that disaggregates major particle components into chemically distinct groups that are contributed by different types of sources. A linear rollback model is based on the assumption that ambient concentrations (C) above background (C_b) are directly proportional to total emissions in the region of interest (E):

$$C - C_b = kE \qquad (1)$$

The proportionality constant, k, is determined over a historical time period when both concentrations C and C_b as well as regional emissions E are known. Once k is determined, new concentration estimates can be derived for other emission levels of interest assuming that meteorological conditions are constant over the same averaging time. Because the anthropogenic components in the particle mass consist almost entirely of sulfates, nitrates, organic carbon, elemental carbon, and crustal material, a maximum contribution from fire can be made based on the assumption that all of the organic carbon or elemental carbon is from primary fire emissions. Various complexities can be added to this model; components can be disaggregated by particle-size fraction (coarse versus fine particles) as well as by chemical composition. Additional distinctions can be made between primary and secondary particles, and nonlinear transformation processes can be approximated to account for atmospheric reactions.

Simple proportional speciated rollback models require data on the chemical composition of airborne particles, knowledge or assumptions regarding secondary particle components, an emission inventory for the important source categories for each particle component and each gaseous precursor, and knowledge or assumptions regarding background concentrations for each component of the aerosol and each gaseous precursor.

The speciated rollback model was applied by the NRC Committee on Haze in National Parks and Wilderness Areas to apportion regional haze in the three large regions of the country (East, Southwest, and Pacific Northwest) by including extinction coefficients to the estimated mass concentrations (NRC 1993). The percentage of anthropogenic light extinction apportioned to forest management burning was estimated at 11 percent in the Northwestern United States on an annual basis assuming that about one-third of the measured organic carbon is of natural origin. The 1985 National Acid Precipitation Assessment Program (NAPAP) inventory was used in this analysis, which also assumed that the elemental carbon and organic carbon fractions of the PM2.5 emissions for forest management burning were 6 percent and 60 percent, respectively.

The limitations of the speciated rollback model are several:

- Deviations from the assumption of spatially homogeneous emissions are likely to occur when air quality is most critical at a single receptor where a single emission source can have an inordinate impact.
- Secondary particle formation is assumed to be linear to changes in precursor emissions.
- Meteorological conditions do not change from year to year.
- Emission inventory errors have a direct, proportional effect on the model estimates.

The model can be applied to any temporal concentration such as annual average, worst 20th percentile, or worst daily average scenarios in any region that meets the constraint on the spatial distribution of emission changes. It is straightforward, necessary input data are available, and the model assumptions are easily understood. It makes use of chemical speciation data collected from the IMPROVE network but cannot apportion contributions made from source classes not included in the inventory.

Chemical Mass Balance Model—The chemical mass balance model, CMB7 (Watson 1997; Watson and others 1990), infers source contributions based on speciated aerosol samples collected at a monitoring site. Chemical elements and compounds in ambient aerosol are "matched" to speciated source emission profiles "fingerprints" by using least-squares, linear regression techniques to apportion the aerosol mass. CMB7 has been widely used within the regulatory community to identify and quantify the sources of particles emitted directly to the atmosphere. The model is based on the relationship between characteristics of the airborne particle (c_i), the summation of the product of the ambient mass concentration contributed by all sources (S_j), and the fraction of the characteristic component in the source's fingerprint (f_{ij}).

$$c_i = \sum_j S_j f_{ij} \qquad (2)$$

Given detailed information about the chemical speciation of the ambient aerosol and similar information about all of the emission sources impacting the receptor, the CMB7 model can apportion the aerosol mass among the sources if certain assumptions are met.

To minimize error, there must be more aerosol components than sources to be included in the least-squares linear regression fit. If there are more components measured than sources, then the comparison of model-estimated concentrations of these additional components provides a valuable internal check on model consistency.

The chemical components in the source "fingerprint" must be conserved and not altered during atmospheric transport — a rather large limitation.

Model resolution is typically limited to five or six source types, and separation of two sources with similar emission profiles (for example, prescribed burning and residential woodstove smoke) is difficult if both sources are active at the same time.

Systematic error analysis procedures have been developed for the CMB7 model, and the results have been published in model validation studies (NRC 1993). However, the model cannot apportion secondary aerosols (sulfate and nitrate); it is limited in its ability to apportion all of the mass to specific sources.

The ability of the model to apportion smoke from fire depends on several factors:

- The presence or absence of smoke from other forms of vegetative burning (woodstoves, agricultural burning, open burning, and others).
- The magnitude of the smoke impact at the receptor (for example, well-dispersed smoke that contributes small amounts of aerosol mass is more difficult to distinguish).
- The uncertainty in both the ambient aerosol and the source "fingerprint" components that the model most heavily weights in the regression analysis, typically organic carbon, potassium, and elemental carbon. The greater the uncertainty of these measurements, the less "fitting pressure" they have in influencing the regression solution.
- Inclusion of multiple aerosol components that are as nearly unique to smoke from fires (endemic signatures) as possible. These include organic compounds such as retene and levoglucosan, as well as gaseous signature such as carbon monoxide and methyl chloride. The more the source profile distinguishes prescribed or wildland fire smoke from other sources, the more accurate the source apportionment is likely to be.

Factor Analysis and Multiple Linear Regression

When many ambient samples are available, linear regression and factor analysis techniques can be applied to the dataset to obtain empirical insights into the origin of the particles. Factor analysis is based on the assumption that chemical components of the aerosol that covary are emitted from a common source. Cluster patterns can then be matched to the source profiles of known sources to identify the degree of covariance associated with a specific source category. Source profiles can be recovered from the ambient data by using two special forms of factor analysis (VARIMAX rotation) or, when the profiles are approximately known, target transformation factor analysis (Hopke 1985). Factor analysis can therefore serve to refine the source profile information used in chemical mass balance analysis. In the context of wildfire smoke apportionment, investigators have historically looked for a high degree of covariance between organic carbon, elemental carbon, and potassium (total, water soluble, and/or nonsoil potassium) as the cluster components that signal particles emitted from vegetative burning of all kinds. Unfortunately, these components of the aerosol are not necessarily unique to smoke from vegetative burning.

Linear regression analysis is a well-established statistical procedure for estimating unknown coefficients in linear relationships where a large dataset of observations of both the dependent and independent variables are present. In the terminology of regression analysis, c in equation (2) is the variable, Sj is the independent variable, and fj the regression coefficients. In practice, the independent variable is taken to be proportional to source strength rather than the source strength themselves. Multiple linear regression has been widely used to apportion total particle mass, the most common approach being use of signature concentrations taken directly as the independent variable, fj. Significantly, gaseous pollutant data can be included in the regression to increase the model's ability to resolve sources. Although carbon monoxide would greatly enhance the success of the model as it is emitted by wildfires in large quantities and is stable in the atmosphere, carbon monoxide is not routinely included in nonurban monitoring programs.

Regression analysis has been used successfully to apportion the total carbon portion of the aerosol mass between wood smoke, vehicle exhaust, and other sources by using nonsoil potassium . The regression-derived estimates were then validated by ^{14}C isotope analysis, which is a direct indicator of "contemporary" versus fossil fuel carbon sources. The ^{14}C measurements nicely confirmed the source apportionment results by regression analysis (r=0.88) (NRC 1993).

Summary

Receptor-oriented methods of particle mass source apportionment have proven successful in a large number of urban studies worldwide. A number of these studies have attempted to apportion wildfire smoke on the basis of a set of aerosol and source emission trace elements and compounds. The experimental design of these studies has limited the ability of receptor models to resolve wildfire smoke from other sources. With improvements in speciation of the organic carbon component of the aerosol, and inclusion of carbon monoxide, methyl chloride, and other endemic signatures, the ability of these techniques to resolve sources and minimize uncertainties will increase. Sensitivity studies are needed to determine which additional components beyond the standard array of trace elements, ions, and carbon fractions would be most beneficial to include in future monitoring programs.

Mechanistic Models

As noted in chapter 6, multiple dispersion models have been used to estimate air quality impacts of single or multiple fires at local and regional scales. Eulerian regional-scale models have been principally used for source apportionment application both to estimate contributions to particulate air quality and regional haze. The suitability of such models for apportionment applications largely depends on the completeness and accuracy of the emission inventory inputs used by the model. Unfortunately, few field validations are available.

Notes

Chapter 8: Consequences of Fire on Air Quality

The potential impacts of fire-induced degradation of air quality on public health and welfare range from occupational exposure of smoke on firefighters to broader economic and social impacts and highway safety.

Health Effects

National Review of Health Effects

In 1996, the Environmental Protection Agency (EPA) conducted an extensive review of the science relating human health effects to particulate matter (PM), the principal pollutant of concern from fires (EPA 1996). The review found that (1) epidemiological studies suggest a variety of health effects at concentrations found in several U.S. cities and (2) ambient particles of greatest concern to health were those smaller than 10 micrometers in diameter. Results of efforts to trace the physiological and pathological responses of the body to PM are unclear, and demonstration of possible mechanisms linking ambient PM to mortality and morbidity are derived from hypotheses in animal and human studies. It is known, however, that PM produces physiological and pathological effects by a variety of mechanisms, including:

- Increased airflow obstruction by PM-induced narrowing of airways.
- Impaired clearance of lung pathways caused by hypersecretion of mucus caused by PM exposure.
- Lung responses to PM exposure including hypoxia, broncho-constriction, apnea, impaired diffusion and production of inflammatory mediators.
- Changes in the epithelial lining of the alveolar capillary membrane that increase the diffusion distances across the respiratory membrane, thereby reducing the effectiveness of blood gas exchange.
- Inflammatory responses that cause increased susceptibility to asthma, chronic obstructive pulmonary disease (COPD) and infections.

Recent information also suggests that several subgroups within the population are more sensitive to PM than others. Children are more likely to have decreased pulmonary function, while increased mortality has been reported in the elderly and in individuals with cardiopulmonary disease. Asthmatics are especially susceptible to PM exposure. In addition, coarse (2.5 to 10μm) particles from road dust or windblown soil were found to have less toxicity than fine particles (less than 2.5μm) that include acid aerosols, diesel emissions, smoke from fires, and potentially carcinogenic PAH compounds.

Occupational Exposure to Wildland Fire Smoke

Wildland firefighters and fire managers have long been aware that smoke exposure occurs during their work (Reinhardt and Ottmar 1997; Sharkey 1997). Although the long-term health effects from occupational smoke exposure remain unknown, the evidence to date suggests that brief, intense smoke exposures can easily exceed short-term exposure limits in peak exposure situations such as direct attack and holding firelines downwind of an active wildfire or prescribed burn. Shift-average exposure only occasionally exceeds recommended instantaneous exposure limits set by the American Conference of Governmental Industrial Hygienists (ACGIH), and rarely do they exceed Occupational Safety and Health Administration (OSHA) time weighted average (TWA) limits (fig. 8-1) (Reinhardt and Ottmar 2000; Reinhardt and others 2000). Overexposure increases to 10 percent of the time if the exposure limits are adjusted for unique aspects of the fire management workplace; these aspects include hard breathing, extended hours, and high elevations, all factors which intensify the effects of many of the health hazards of smoke (Betchley and others 1995; Materna and others 1992; Reinhardt and Ottmar 2000; Reinhardt and others 2000). It could be argued that few firefighters spend a working lifetime in the fire profession, and thus they should be exempt from occupational standards that are set to protect workers over their careers. But this argument is irrelevant for irritants and fast-acting health effects such as eye and respiratory irritation, headache, nausea, and angina. An exposure standard specifically for wildland firefighters and appropriate respiratory protection may need to be developed (Reinhardt and Ottmar 2000).

Figure 8-1—Firefighters being monitored for smoke exposure. Monitoring equipment seen includes a red backpack that collects gas samples from the breathing zone of the firefighters and a white-colored particulate matter filter sampler attached to the chest. (Photo by Roger Ottmar)

In spite of the studies that have been done, major data gaps remain:

- In the area of health hazards, not enough evidence is available to defend the commonly cited "inert" classification of total and respirable particulate in dust and smoke; there is little knowledge of the occurrence of crystalline silica in dust at fires; and there is incomplete characterization of aldehydes and other respiratory irritants present in smoke (Reinhardt and Ottmar 1997, 2000).
- The differences in smoke exposure between large and small wildland fires have not been characterized in spite of the fact that one or two crews extinguish the vast majority of wildfires (Reinhardt and Ottmar 2000).
- The long-term health experience of wildland firefighters is unknown, although anecdotal reports and the biological plausibility of cumulative health effects indicate a potentially greater incidence of disease and death than in the general population of workers (Booze and Reinhardt, in press; Sharkey 1997).

Although data gaps remain, enough information has been gathered to chart a course to alleviate many of the overexposures. Respiratory protection is available for irritants such as aldehydes and particulate matter but not for carbon monoxide. Respirators can be heavy, hot, and impede the speed of work, but some new models are light, simple and could be worn only when needed (Beason and others 1996; Rothwell and Sharkey 1995). The entire costly process of medical evaluations, fitness testing, maintenance, and training must be employed if respirators are to be used. But there are immediate benefits to reducing respiratory irritant exposure. Small electrochemical dosimeters can provide instant warnings about carbon monoxide levels in a smoky situation, and fire crew members equipped with respirators and carbon monoxide monitors have all the protection necessary to stay and accomplish objectives safely and withdraw when the carbon monoxide levels become the limiting factors (Reinhardt and others 1999). In the future, a respirator for use during wildland fires may be developed that offers warning and protection against carbon monoxide as well. Although some work has been done in this area, we need more significant development. Smoke exposure is a hazard only a small portion of the time and is manageable because the situation where it occurs can be predicted. A long-term program to manage smoke exposure at wildland fires could include (1) hazard awareness training, (2) implementation of practices to reduce smoke exposure such as rotating crews and providing clean air sites, (3) routine carbon monoxide monitoring with electronic dosimeters, (4) improved recordkeeping on accident reports to include separation of smoke related illness among fireline workers and fire camp personnel, and (5) improved nutritional and health habits. Fire management practices such as crew rotation, awareness training, and carbon monoxide monitoring can mitigate the hazard and allow firefighters to focus on the job of fire management, lessening the distraction, discomfort, and health impacts of smoke exposure (Reinhardt and Ottmar 2000).

Research Issues

A number of wildland fire health effect research issues flow from the EPA staff report (Clean Air Scientific Advisory Committee1995) and occupational health exposure studies.

Research into the health effects of particulate matter is largely based on epidemiological studies conducted over long periods in urban centers with high hospital admittance or large air quality databases, or both. Consequently, inadequate information is available that relates short-term, acute smoke exposure (such as would be experienced by a visitor to a National Park or to a community near a wildfire) to human health effects. As a result, little or no specific guidance is available to wildland fire managers, air quality regulators, or public health officials who need to responsibly judge the public health risks of exposure to extremely high smoke concentrations. This gap in knowledge was clearly evident during the 1988 Yellowstone fires and later wildfire events when quick decisions had to be made on how best to protect public health in communities near major wildfires (WESTAR 1995). The best available guidelines are those published by EPA (1999) for assessing the risk to health from air pollution (table 8-1). These guidelines may or may not reflect the specific hazards of pollutants from fires, which will have a different chemical composition.

Table 8-1—Pollutant-specific breakpoints for the air quality index (AQI) and accompanying health effects statements (adapted from EPA 1999).

Category	PM2.5 (24-hour)		PM10 (24-hour)	
	Concentration breakpoints	Health effects statements	Concentration breakpoints	Health effects statement
	$\mu g/m^3$		$\mu g/m^3$	
Good	0.0-15.4	None	0-54	None
Moderate	15.5-40.4	None	55-154	None
Unhealthy for sensitive groups	40.5-65.4	Increasing likelihood of respiratory symptoms in sensitive individuals, aggravation of heart or lung disease and premature mortality of persons with cardiopulmonary disease and the elderly.	155-254	Increasing likelihood of respiratory symptoms and aggravation of lung disease, such as asthma.
Unhealthy	65.5-150.4	Increased aggravation of heart or lung disease and premature mortality in persons with cardiopulmonary disease and the elderly; increased respiratory effects in the general population.	255-354	Increased respiratory symptoms and aggravation of lung disease, such as asthma; possible respiratory effects in general population.
Very unhealthy	150.5-250.4	Significant aggravation of heart or lung disease and premature mortality in persons with cardiopulmonary disease and the elderly; significant increase in respiratory effects in general population.	355-424	Significant increase in respiratory symptoms and aggravation of lung disease, such as asthma; increasing likelihood of respiratory effects in general population.
Hazardous	250.5-500.4	Serious aggravation of heart or lung disease and premature mortality in persons with cardiopulmonary disease and the elderly; serious risk of respiratory effects in general population.	425-604	Serious risk of respiratory symptoms and aggravation of lung disease, such as asthma; respiratory effects likely in the general population.

The long-term health effects of smoke exposure to wildland firefighters are unknown in spite of anecdotal evidence that indicates the possibility of a greater incidence of cardiopulmonary disease and death than in the general population. Although carbon monoxide monitoring and respiratory protection can mitigate the hazard, personal protection equipment is still needed that allows firefighters to work effectively without discomfort or distraction (Reinhardt 2000).

Welfare Effects

Air quality-related effects of smoke include the soiling of materials, public nuisance, and visibility loss. Because these and other consequences of smoke have come increasingly into conflict with the public's interest in clean air, an understanding of these effects is important to fire managers.

Soiling of Materials

The deposition of smoke particles on the surface of buildings, automobiles, clothing, and other objects reduces aesthetic appeal and damages a variety of objects and building structures (Baedecker and others 1991). Studies of the effect of aerodynamic particle size on soiling have concluded that coarse particles (2.5 to 10μm) initially contribute more to soiling of both horizontal and vertical surfaces than do fine particles (less than 2.5μm), but that coarse particles are more easily removed by rainfall (Haynie and Lemmons 1990). Smoke from fires is largely within the fine mode, although ash fallout in the near vicinity of a fire is often also a concern. Smoke may also discolor artificial surfaces such as building bricks or stucco, requiring cleaning or repainting. Increasing the frequency of cleaning, washing, or repainting soiled surfaces becomes an economic burden and can reduce the life usefulness of the soiled material (Maler and Wyzga 1976).

Soiling from smoke also changes the reflectance of opaque materials and reduces light transmission through windows and other transparent materials (Beloin and Haynie 1975).

When fine smoke particles (less than 2.5μm) infiltrate indoor environments, soiling of fabrics, painted interior walls, and works of art may occur. Curtains may require more frequent washing because of soiling or may deteriorate along folds in the fabric after being weakened by particle exposure (Yocom and Upham 1977). As in the case of corrosion damage from acidified particles, these same particles accelerate damage to painted surfaces (Cowling and Roberts 1954). Studies of the soiling of works of art at a museum in southern California concluded that a significant fraction of the dark-colored fine mode elemental carbon and soil dust originated from outdoor sources (Ligocki and others 1993). Smoke from fires is one source of elemental carbon.

Public Nuisance and Visibility Loss

Nuisance smoke is the amount of smoke in the ambient air that interferes with a right or privilege common to members of the public, including the use or enjoyment of public or private resources (EPA 1990). The abatement of nuisance smoke is one of the most important objectives of successful smoke management (Shelby and Speaker 1990). Public complaints about nuisance smoke are linked to loss of visibility, odors, and ash fallout that soils buildings, cars, laundry, and other objects. Acrolein (and possibly formaldehyde) in smoke at distances of 1 mile from the fireline are likely to cause eye and nose irritation, exacerbating public nuisance conditions (Sandberg and Dost 1990).

Perhaps the most significant nuisance effect of smoke from fire is local visibility reduction in areas impacted by the plume. While visibility loss within Class I areas is subject to regulation under the Clean Air Act, smoke plume-related visibility degradation in urban and rural communities is not. Nuisance is usually regulated under State and local laws and is frequently based on public complaint or, when highway safety is compromised, the risk of litigation (Eshee 1995). The courts have also ruled that the taking of private property by interfering with its use and enjoyment caused by smoke (and without just compensation) is in violation of Federal Constitutional provisions under the Fifth Amendment. The trespass of smoke may diminish the value of the property, resulting in losses to the owner (Iowa Supreme Court 1998).

Because the public links visibility loss with concerns about the health implications of breathing smoke, smoke management programs have been under increasing pressure to minimize emissions and reduce smoke impacts to the greatest degree possible (Core 1989). Visibility reduction is used as a measure of smoke intrusions in several smoke management plans. The State of Oregon program operational guidance defines a "moderately" intense intrusion as a reduction of from 4.6 to 11.4 miles from a background visibility of more than 50 miles (Oregon Department of Forestry 1992). The State of Washington smoke intrusion reporting system uses a "slightly visible," "noticeable impact on visibility" or "excessive impact on visibility" to define light, medium, and heavy intrusions (Washington Department of Natural Resources 1993). The State of New Mexico program requires that visibility impacts of smoke be considered in development of the unit's burn prescription (New Mexico Environmental Improvement Board 1995).

Economic and Social Consequences

The economic consequences of smoke are principally in the areas of soiling-related losses and costs related to reduced visibility.

Soiling-Related Economic Losses

Economic costs associated with materials damage and soiling caused by airborne particles include reduction in the useful life of the damaged materials and the decreased utility of the object. Losses caused by the need for more frequent maintenance and cleaning are also significant. Amenity losses occur when the increased cleaning or repair of materials results in inconvenience or delays, many of which are difficult to quantify (Maler and Wyzga 1976).

Within the United States, however, the soiling of buildings constitutes the largest category of surface areas at risk to pollution damage (Lipfert and Daum 1992). Soiling on painted surfaces on residential buildings, resulting in a need to repaint exterior walls, has caused damage approaching $1 billion per year (Haynie and others 1990).

Willingness-to-pay estimates developed using the contingent valuation method found that households were willing to pay \$2.70 per $\mu g/m^3$ charge in particle pollution to avoid soiling effects (McClelland and others 1991). No estimates are available for costs specifically associated with smoke from fires.

Visibility-Related Costs

The importance of clean, clear air within the wildlands and National Parks of this nation is hard to overemphasize. People go to these special places to enjoy scenery, the color of the landscapes, and clarity of the vistas. At Grand Canyon, 82 percent of 638 respondents rated "clean, clear air" as very important or extremely important to their recreational experience (Ross 1988). Three National Park Service (NPS) studies determined that air quality conditions affect the amount of time and money visitors are willing to spend at NPS units (Brookshire and others 1976; MacFarland and others 1983; Schulze and others 1983). These studies found estimated onsite use values for the prevention or elimination of plumes that ranged from about $3 to $6 (1989 dollars) per day per visitor party at the park. Based on these results, the implied preservation value for preventing a visible plume most days (the exact frequency was not specified) at the Grand Canyon was estimated at about $5.7 billion each year when applied to the total U.S. population (EPA 1996). Other investigators have suggested that these estimates are overstated by a factor of 2 or 3 (Chestnut and Rowe 1990).

In the studies noted above, park visitors generally responded that they would be willing to spend more time and money if visibility conditions were better and, conversely, less if visibility conditions were worse (Ross 1988). The average amount of time visitors were willing to spend traveling to a vista for every unit change in visibility (.01 km^{-1} extinction coefficient) was between 15 minutes and 4 hours. These results provide evidence that changes in visual air quality can be expected to affect visitor enjoyment and satisfaction with park visits.

Even given the limitations and uncertainties of contingent valuation surveys, economic values related to visibility degradation are clearly likely to be substantial.

Public Perception of Haze—Perceived visual air quality (PVAQ) has been used as a measure of the public's acceptance of haze conditions (Middleton and others 1983). Subjects were asked to judge the visual air quality in several photos depicting vistas under different haze conditions using a scale of 1 to 10, 1 being the worst and 10 being the best. These 1 to 10 scales reflect people's perceptions and judgments concerning visibility conditions. By matching particulate air quality conditions that occurred at the time of the photographs, researchers have been able to develop a relationship between PVAQ and particulate matter concentrations (Middleton and others 1985). Even small increases in particulate concentrations in the atmosphere result in dramatic decreases in PVAQ. Because of the light scattering efficiency of smoke, this relationship is especially applicable to fire emissions.

Cultural Consequences of Visibility Loss—"National parks and wilderness areas are among our nation's greatest treasures. Ranging from inviting coastal beaches and beautiful shorelines to colorful deserts and dramatic canyons to towering mountains and spectacular glaciers, these regions inspire us as individuals and as a nation" (NRC 1993). With these words, the National Research Council (NRC) noted the importance of preserving the scenic vistas of the nation. Congress, in recognition of the scenic values of the nation, adopted the Clean Air Act Amendments of 1977, which established a national visibility protection program. The GCVTC was later established in the 1990 amendments to the act to address visibility impairment issues relevant to the region surrounding Grand Canyon National Park. Following 4 years of study, the GCVTC concluded that smoke from wildland fires is likely to have the single greatest impact on visibility in Class I areas of the Colorado Plateau through the year 2040 (GCVTC 1996c). While difficult to quantify, there is consensus that visibility loss associated with smoke from wildland fire and other sources has important cultural consequences on the nation.

Highway Safety

Smoke can cause highway safety problems when it impedes a driver's ability to see the roadway (fig. 8-2) and can result in loss of life and in property damage at smoke levels that are far below NAAQS. This section focuses on highway safety issues in the Southeastern United States because this is where the foremost forestry-related air quality problem has been in the past. We also describe tools being developed to aid the land manager in avoiding highway safety problems.

Although smoke at times can become a problem anywhere in the country, it is in the Southern States, from Virginia to Texas and from the Ohio River southward, where highway safety is most at risk from prescribed fire smoke, principally because of the amount of burning done in the South and the proximity of wildlands to population centers. Roughly 4 million acres of Southern forests are treated with prescribed fire each year (after Wade and Lunsford 1988). This area is by far the largest acreage subjected to prescribed fire in the country. Prescribed fire treatment intervals, especially in Southern pine (in an area extending roughly from Virginia to Texas), is every 3 to 5 years. These forests are intermixed with homes, small towns, and scattered villages within an enormous wildland/urban interface. During the daytime, smoke becomes a problem when it drifts into these areas of human habitation. At night, smoke can become entrapped near the ground and, in combination with fog, creates visibility reductions that cause roadway accidents. The potential exists for frequent and severe smoke intrusions onto the public roads and highways from both prescribed and wildland fires.

Magnitude of the Problem

Smoke and smoke/fog obstructions of visibility on Southeastern United States highways cause numerous accidents with loss of life and personal injuries every year. Several attempts to compile records of smoke-implicated highway accidents have been made. For the 10 years from 1979 through 1988, Mobley (1989) reported 28 fatalities, over 60 serious injuries, numerous minor injuries, and millions of dollars in lawsuits. During 2000, smoke from wildfires drifting across Interstate 10 caused at least 10 fatalities, five in Florida and five in Mississippi.

As the population growth in the South continues, more people will likely be adversely impacted by smoke on the highways. Unless methods are found to adequately protect public safety on the highways, there exists the prospect that increasingly restrictive regulations will curtail the use of prescribed fire or that fire as a management tool may be altogether prohibited.

Figure 8-2—Smoke can cause highway safety problems when it impedes a driver's ability to safely see the roadway. (Photo by Jim Brenner)

Measures to Improve Highway Safety

Several approaches are being taken to reduce the uncertainty of predicting smoke movement over roadways:

High-resolution weather prediction models promise to provide increased accuracy in predictions of wind speeds and directions and mixing heights at time and spatial scales useful for land managers. The Florida Division of Forestry (FDOF) is a leader in the use of high resolution modeling for forestry applications in the South (Brenner and others 2001). Because much of Florida is located within 20 miles of a coastline, accurate predictions of sea/land breezes and associated changes in temperature, wind direction, atmospheric stability, and mixing height are critical to the success of the FDOF. High-resolution modeling consortia are also being established by the USDA Forest Service to serve clients with interests as diverse as fire weather, air quality, ecology, and meteorology. These centers involve scientists in development of new products and in technology transfer to bring the products to consortia members.

Several smoke models are in operation or are being developed to predict smoke movement over Southern landscapes. VSMOKE (Lavdas 1996), a Gaussian plume model that assumes level terrain and unchanging winds, predicts smoke movement and concentration during daytime. VSMOKE has been made part of the FDOF fire and smoke prediction system. It is a screening model that aids land managers in assessing where smoke might impact sensitive targets as part of planning for prescribed burns. PB-Piedmont (Achtemeier 2001) is a wind and smoke model designed to simulate smoke movement near the ground under entrapment conditions at night. The smoke plume is simulated as an ensemble of particles that are transported by local winds over complex terrain characteristic of the shallow (30 to 50 m) interlocking ridge/valley systems typical of the Piedmont of the South. Two sister models are planned — one that will simulate near-ground smoke movement near coastal areas influenced by sea/land circulations, and the other for the Appalachian Mountains.

Climate Change

Globally, fires are a significant contributor of carbon dioxide and other greenhouse gases in the atmosphere. Fires are also an important mechanism in the redistribution of ecosystems in response to climate stress, which in turn affects the atmosphere-biosphere carbon balance. Currently, there is no policy mandate, nor widely accepted methodology for managing fires, for the conservation of terrestrial carbon pools or mitigation of greenhouse gas emissions. However, we may expect carbon accounting and perhaps conservation to become a part of fire and air resource management if and when global agreements are made to address biomass burning and resultant greenhouse gas emissions.

Notes

Chapter 9: Recommendations for Future Research and Development

Managing smoke and air quality impacts from fires requires an increasing base of knowledge obtained through research and the development of information systems. Fire and air resource managers have had the responsibility since the 1960s to mitigate direct intrusions of smoke into areas where it presents a health or safety hazard, or where it is simply objectionable to an affected population. In more recent years, that responsibility has broadened because of an increase in the use of fire, more people in the wildland/urban interface, tightening of regulatory standards, and decreasing public tolerance for air pollution. More States require smoke management plans, and the plans are increasingly complex due to increased coverage and greater requirements for notification, modeling, monitoring, and recordkeeping.

Established Research Framework

There is ample strategic analysis and workshop output to guide research. The most comprehensive and up-to-date recommendations for research and development are found in *National Strategic Plan: Modeling and Data Systems for Wildland Fire and Air Quality* (Sandberg and others 1999). Workshop sessions, internal discussion, and review comments were compiled into more than 200 proposals from which 46 priority projects were selected that support the nine summary recommendations outlined here.

Recommendation 1: Fuels and fire characteristics—An ability to estimate emissions from all types of fires over the wide variation in fuels in the contiguous United States and Alaska is needed. Expanded models and fuel characteristics data are needed to fill this gap.

Recommendation 2: Emissions modeling systems—Current models to estimate emissions are inadequate in coverage and incomplete in scope. Emissions production models need to be expanded to include all fire and fuel types as well as linked to fire behavior and air quality models in a geographically resolved data system.

Recommendation 3: Transport, dispersion, and secondary pollutant formation—Air quality and land management planners lack spatially explicit planning and real-time systems for assessing air quality impacts. A geographic information system (GIS) based system linked to emissions production, meteorological, and dispersion models is needed.

Recommendation 4: Air quality impact assessment—Better wildland and prescribed fire information is needed to compile emissions inventories, for regional haze analysis and for determination of compliance with National Ambient Air Quality Standards (NAAQS).

Recommendation 5: Emissions tradeoffs and determination of "natural" visibility background assessments—No policy-driven or scientific definition of "natural" background visibility exists for regional haze assessments. The tradeoffs between wildfire and prescribed fire emissions are also not known. To address these issues, the policy community needs to decide what types of fires contribute to natural impairment after which a scientific assessment could be done and tradeoffs evaluated.

Recommendation 6: Impact and risk assessment of emissions from fire—A comprehensive assessment of smoke exposure of prescribed and wildland firefighters and the public at current levels of fire activity should be done to provide a baseline for future risk assessments. Exposures should be periodically reassessed to evaluate increased risks from future increases in fire emissions.

Recommendation 7: Monitoring guidelines and protocols—Guidelines are needed on how best to monitor source strength, air quality, visibility, and nuisance impacts from fires to support consistent and quantitative evaluation of air impacts.

Recommendation 8: National fire and air quality information database—A readily accessible source of information on past, current, and predicted future fire activity levels, emissions production, and air quality impacts from fires does not exist. Such a database is needed to analyze past experiences and replicate successes.

Recommendation 9: Public information and protection—A centralized system is needed to provide information to the public on air quality impacts from fires. Also needed are general criteria for how land managers, air regulators, and public health officials should respond to adverse smoke impacts and emergency notifications of the public to health hazards associated with smoke from fire.

The authors of this plan hoped that these recommendations would be used in future joint agency efforts to advance the fire sciences, minimize duplication of effort, and share information among agencies and the public.

The technically advanced smoke estimation tools, or TASET, project (Fox and Riebau 2000) was funded by the Joint Fire Sciences Program (JFSP) to develop a structured analysis of smoke management and recommend specific developments for advancing the state of science. The report confirmed and refined the recommendations of Sandberg and others (1999) above, and developed 10 recommendations for research activities:

- Fire community participation in regional air quality modeling consortia.
- Conduct a national smoke and visibility conference and reference guide.
- Develop a national smoke emissions data structure or database system.
- Apply remote sensing for fuels and fire area emissions inventories.
- Develop a fire gaming system to quantify emissions and impacts from alternative fire management practices.
- Improve the CalMet/CalPuff smoke management model.
- Upgrade a nationalized screening model/simple approach smoke estimation model (SASEM).
- Provide onsite fire emissions verification.
- Utilize back-trajectory modeling and filter analysis for fire smoke contributions for nonattainment areas.
- Develop a method to identify the specific sources of organic carbon fine particulate material.

Research priorities established in the *Effects of Fire Air* (Sandberg and others 1979) are unfortunately still valid today, although some progress has been made in every category. We list these here, slightly reworded from the original for brevity and to conform to modern nomenclature:

1. Provide quantitative smoke management systems.
   a. Develop information systems necessary to support smoke management decisions.
   b. Provide a smoke management reporting system for emission rates based on the prediction of fuel consumption, fire behavior, heat release rates, and source control measures.
   c. Provide the data network and modeling scheme to calculate the change in pollution concentrations and character between the source and potential receptors.
   d. Adapt plume rise models necessary to predict the vertical distribution of emissions from fires.
2. Characterize the chemistry and physics of emissions.
   a. Relate emissions and heat release rates to fuelbed characteristics and fire behavior.

b. Advance our knowledge of hazardous and reactive compounds in smoke.
c. Develop field methods to monitor emission rates and smoke chemistry from operational fires.
d. Investigate the potential for secondary reactions of emissions downstream from their source.

3. Model atmospheric transport, diffusion, transformation, and removal mechanisms.
   a. Continue development of winds and dispersion models for boundary layer flow and mesoscale transport of smoke over mountainous terrain.
   b. Investigate the mechanisms of removal; for example, canopy interactions, fallout, and local deposition.
   c. Interact with the wider scientific community to establish the effect of reactive pollutants on the biosphere.
   d. Evaluate the potential contribution of wildland fires to climate change.

4. Identify receptor responses to wildland smoke.
   a. Identify and quantify the visibility needs of wildland users, and recommend standards for particulate and sulfate pollution from all sources affecting Class I visibility areas.
   b. Evaluate the potential impact of wildland smoke on human health.
   c. Investigate the role of wildland ecosystems as a sink and receptor for atmospheric contaminants.

5. Investigate tradeoffs made in the substitution of alternatives to fire use.
   a. Develop simulation models to evaluate interactions of land use policy with air resource management. Incorporate air resource management and fuels management needs into the land use planning process.
   b. Evaluate the effect on wildland fire occurrence and air pollution from changes in the amount of prescribed fire activity.
   c. Describe the resource and economic tradeoff of wildland fire occurrence resulting from a change in prescribed fire activity.
   d. Investigate the effect of changes in fire use on nutrient cycling, successional response, and ecosystem stability.

Emerging Research Needs

Several new responsibilities create the need for additional information systems that require new research and development, including:

- Planning rules that require the consideration of cumulative pollution and visibility impacts of fuel management programs.
- Wildland fire situation analysis requirements that smoke impacts from wildland be anticipated and communicated to the public.
- Increased requirements for emission reduction.
- Policies that require hourly and daily tracking of emissions and the management of smoke from all fires.
- Increased management of wildland fires for resource benefits.
- Increased use of long-duration landscape-scale fires.
- Regulatory concern over secondary pollutants, especially ozone formation and the reentrainment of mercury.
- Questions about the role of fire and global biomass emission on atmospheric carbon and global warming.
- Increased attention to firefighter health effects from exposure to smoke.

Each of these factors requires information systems for planning, operations, and monitoring the effects of fire on air. Using the framework illustrated in figures 1-1 and 1-2 (in chapter 1) and the background of previous chapters, some emerging research needs are outlined below.

Emissions Source Strength and Emissions Inventory

Level of burn activity: Accurately predict, determine, and record the area burned and time of burning for all types of prescribed and wildland fire—Area burned is still the parameter that imparts the greatest error into predictions of source strength and emission inventory. Needed are: a balanced program of new planning models that project area burned and fire residence times; remote-sensing technologies that track fire sizes at hourly intervals; ground based sampling, reporting, and communication systems; and analysis tools. Planning models include those that project fire use and predict wildland fire activity from 1 to 50 years in the future must be included, as well as accurate predictions made a day in advance.

Biomass: Accurately predict, determine, and record the mass, combustion stage, and residence time of fuels burned in all types of fires—Inadequate representation of fuelbed characteristics and the ability to infer fuelbed characteristics and flammability conditions from remote sensing or ecosystem physiognomy is the second greatest remaining source of error. Models of the combustion process, while improving, are still inadequate to predict or characterize emission rates and durations. New classification systems, inference models, inventory and sensing processes, and process models are needed.

Heat release and emissions: Predict and measure physical and chemical characteristics of emissions from all types of fires—Among the greatest advances since about 1980 has been the nearly complete characterization of primary and criteria pollutants from a wide range of fire environments. New models also greatly improve the prediction and characterization of emissions source strength. Emission factors for criteria pollutants are adequate. There is substantial remaining uncertainly in the measurement and prediction of precursors to ozone and other secondary chemical formations, secondary entrainment of mercury, production and stimulation of nitrogen compounds, air toxics, and greenhouse gases. Continuing research on these trace constituents are needed. In addition, we lack models that characterize the complex spatial and temporal distribution of heat release from fires.

Emissions inventory methods: Integrate measurements and reporting from remote sensing, airborne platforms, simulation models, and surface observations into a fine-scale spatial and temporal emission inventory—Emission inventories are a fundamental tool that air resource managers use to calculate the relative importance of air pollution sources and to design control strategies. Hourly, point-specific emission estimates as well as daily, monthly, and yearly summaries are necessary to compare fire with other sources or as inputs to dispersion models. Fire managers currently lack a system of observations and reporting mechanisms required for planning, tracking, and monitoring emissions.

Ambient Air Quality Impacts

Background air quality: Improve the accessibility of girded detail about background air quality and meteorological conditions—Fire emissions are inserted into an already complex atmosphere, and current ability to predict pollutant interactions, transformations, and combined effects are limited by the availability of hourly fine-scale atmospheric profiling.

Plume rise and transport: Improve the prediction, detection, and tracking of plumes from all types and stages of fires—Fire plumes are complex; often splitting into lofted and unlofted portions; plumes that split in two directions at different altitudes, and plumes that change rapidly over time. Plumes are transported long distances, often over complex terrain, and the accuracy and availability of models to predict transport are inadequate. Methods to track plume trajectories and measure pollutant concentrations in near real time using remote sensing are emerging but not yet available.

Dispersion, dilution, and pollutant transformation: Improve the ability on all scales to predict, model, and detect changes in the properties and concentration of pollutants over time and space—Data and models are needed to initiate and predict local, regional, national, and global air quality impacts from individual fires to the cumulative effects of tens of thousands of fires.

Atmospheric carbon balance and climatic change: Develop consistent technologies to assess the contribution of fires to greenhouse gases in the atmosphere and the effect of fire and ecosystem management practices—For a source of greenhouse gas emissions as large as wildland and prescribed fires, there is a regrettable lack of consensus on the magnitude or even the methods for assessment and accountability. This emerging issue requires much of the same research on source characteristics and air quality as do the health, safety, and visibility issues, but also requires integration with the global science and policy communities.

Effects on Receptors

Visibility and other welfare effects: Predict, measure, and interpret the impact of natural and anthropogenic fire sources on visibility, economic, and other welfare effects—The impact of smoke exposure from fires on human health standards is minor relative to the nuisance it creates and the impacts on visibility. New science is required to monitor and predict effects on visibility, and to apportion visibility impacts to specific sources and classes of sources.

Health and safety risk assessment: Develop knowledge and systems to assess the risk of individual and collective fires to personal and community health and safety—This broad topic has received limited attention in recent years, mostly in the prediction of visibility impacts on highway safety and in the assessment of individual firefighter exposure to hazardous air pollutants. But all aspects of risk management, including hazard identification, exposure assessment, dose-response, risk assessment, and mitigation measures are lacking.

Conclusion

Knowledge and information requirements for managing fire effects on air quality continue to increase. Policy advancements require the understanding, modeling, prediction, monitoring, and tracking of fires and their effect on air at greater detail and in greater volume than ever before. Research and development has progressed logically over the past 25 years due to strategic planning and prioritization that has included the needs of the managers of ecosystems and of air quality. Analytical and information transfer capacity has increased dramatically in the past decade, so information is more readily accessible to those who need it. Thanks largely to the National Fire Plan, the Joint Fire Science Program, the Western Regional Air Partnership, and EPA's implementation of the Regional Haze Rule, there is currently more active research and development the effects of fire on air than ever before.

Notes

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