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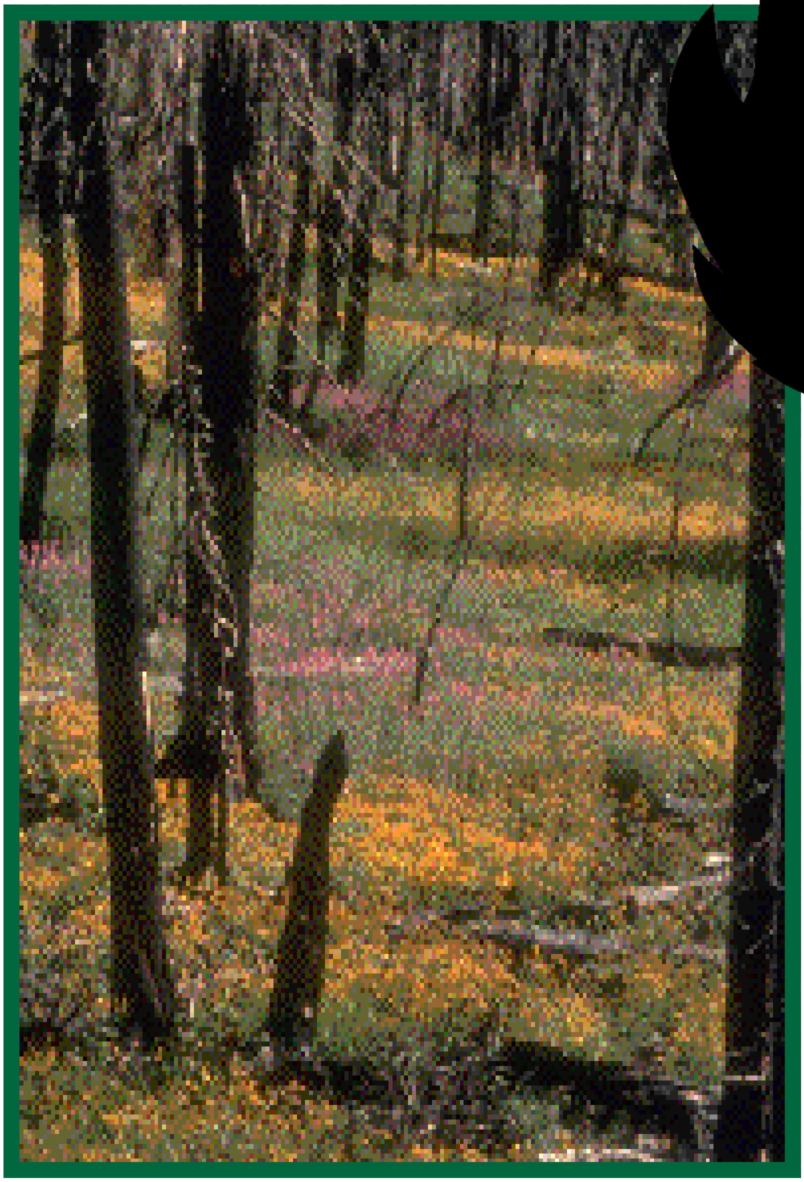
Wildland Fire in Ecosystems

Effects of Fire on Flora



Abstract

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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

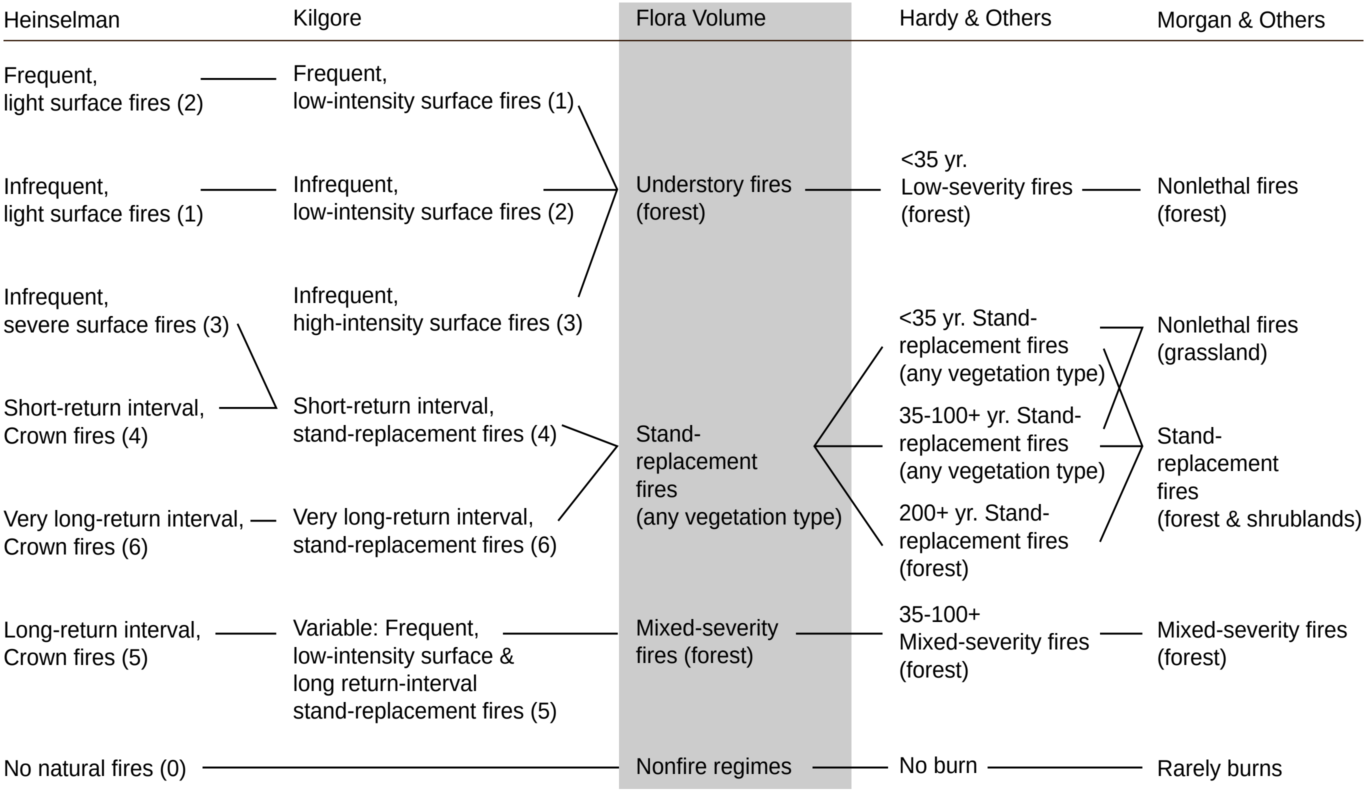

Figure 1-1—Comparison of fire regime classifications by Heinselman (1978), Kilgore (1981), Hardy and others (1998), Morgan and others (1998), and the Flora and Fuel Volume. Lines connect similar fire regime types. In parentheses, forest includes woodlands and grassland includes shrublands.

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.

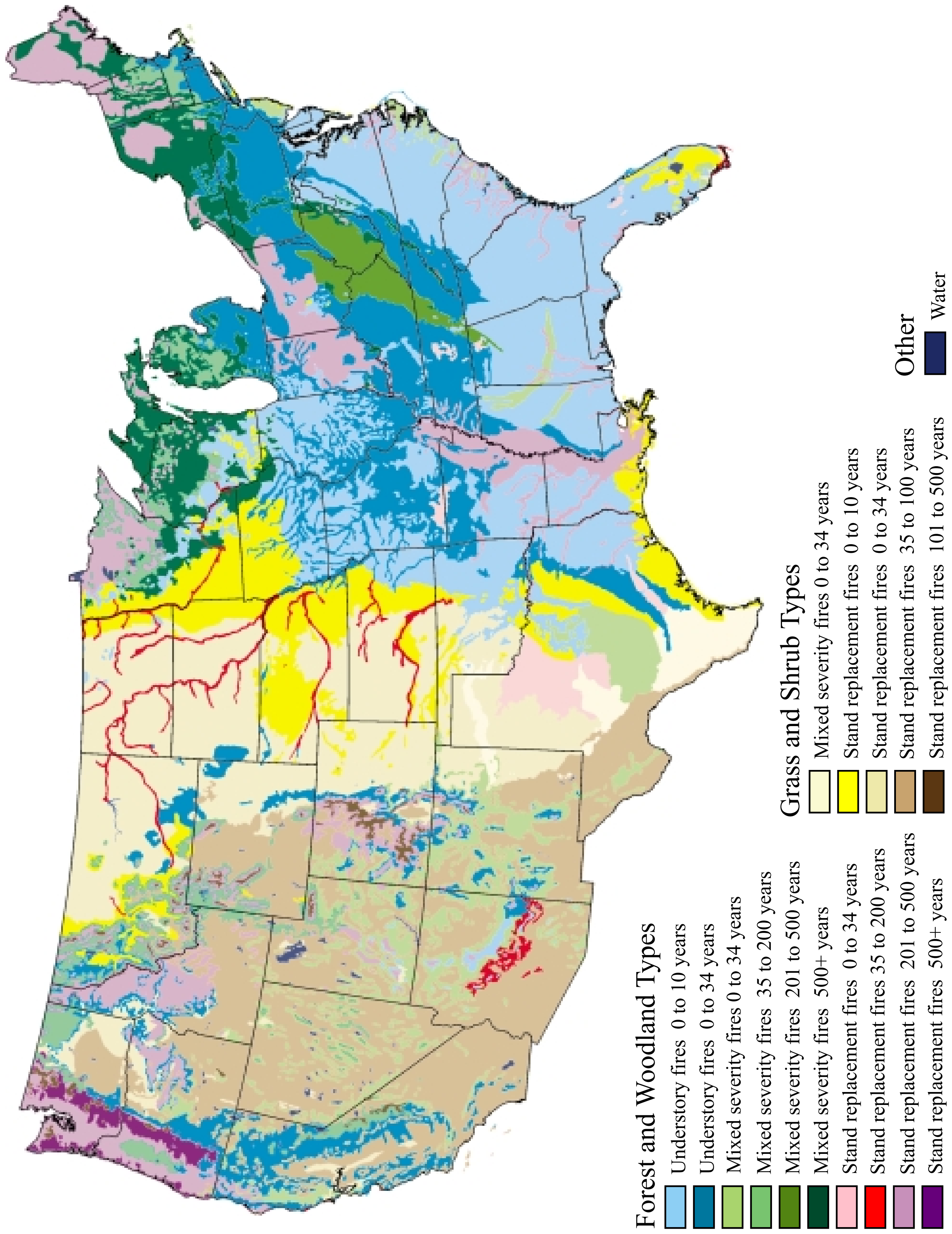

Figure 1-2—Fire regime types based on Kuchler's Potential Natural Vegetation types (prepared by Jim Menakis).

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,	Mature Stump sprouts, Roots	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,	Sapling Stump Sprouts	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).

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

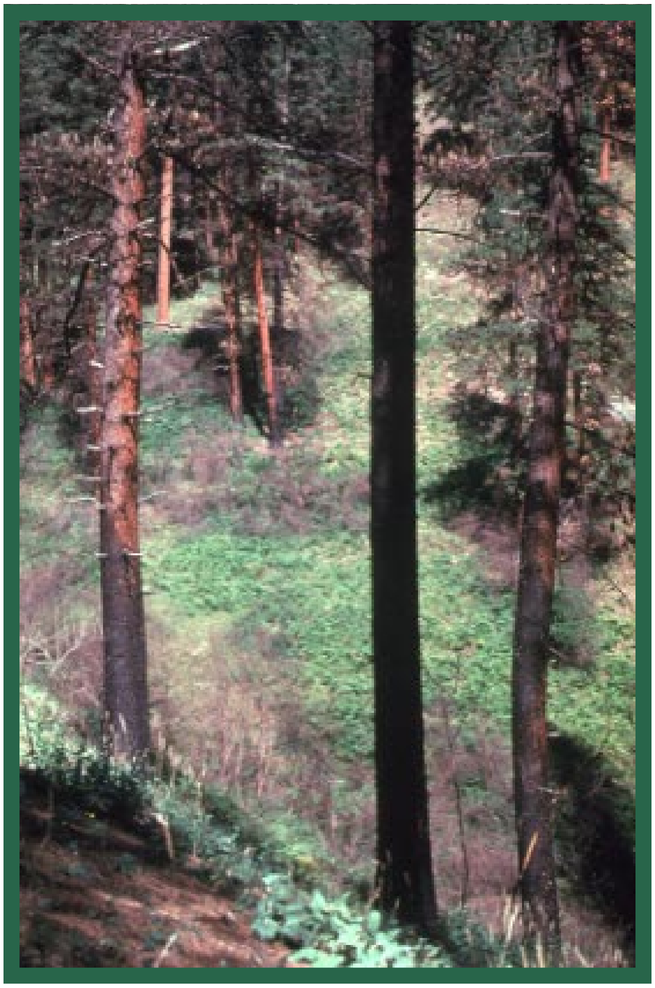

Figure 2-1—Scorched boles on surviving ponderosa pine, Selway-Bitterroot Wilderness, Idaho.

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.

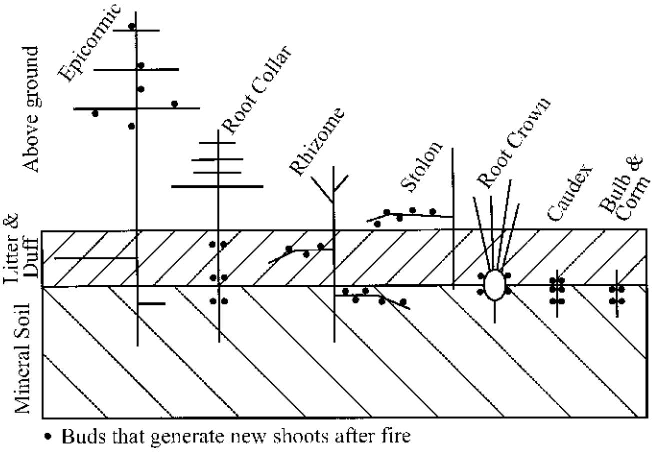

Figure 2-3—Various plant parts that regenerate new shoots and their location in and above the soil.

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.

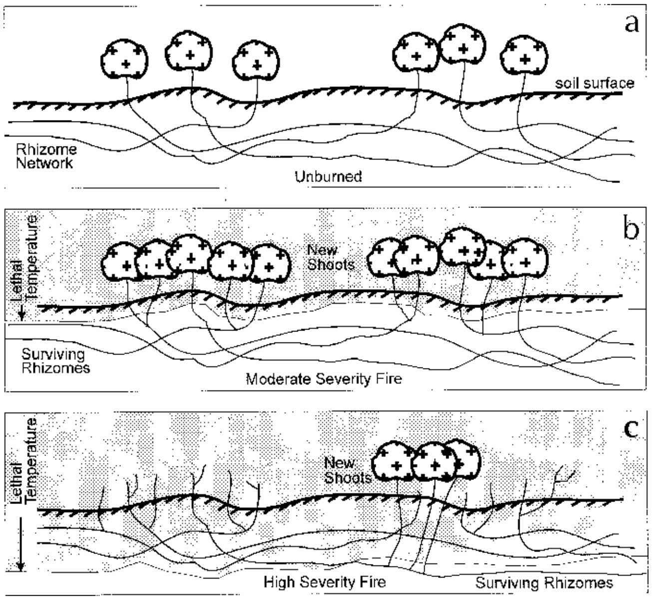

Figure 2-8—Effects of subsurface heating on postfire sprouting of a rhizomatous shrub: (a) Network of unburned rhizomes. (b) Moderate severity fire lethally heated the rhizomes near the surface, but many postfire sprouts grew from more deeply buried rhizomes. (c) High severity fire killed most of the 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.

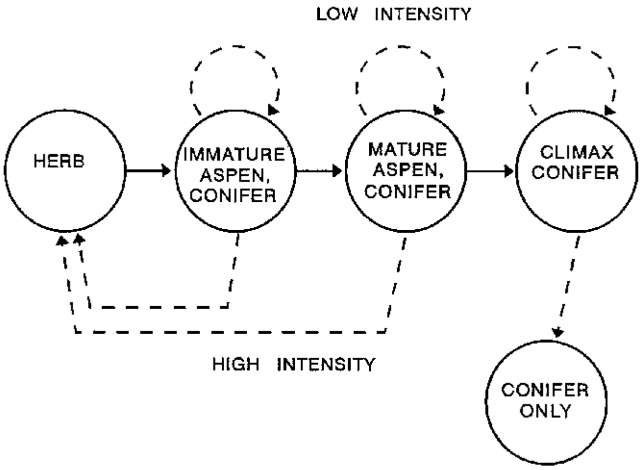

Figure 3-1—Successional pathways involving seral aspen, fire, and conifers (Brown 1985).

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.

FRES	Kuchler	SAF	Fire regime types: Understory Occur[a]	Understory Freq[b]	Mixed Occur	Mixed Freq	Stand-replacement Occur	Stand-replacement 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

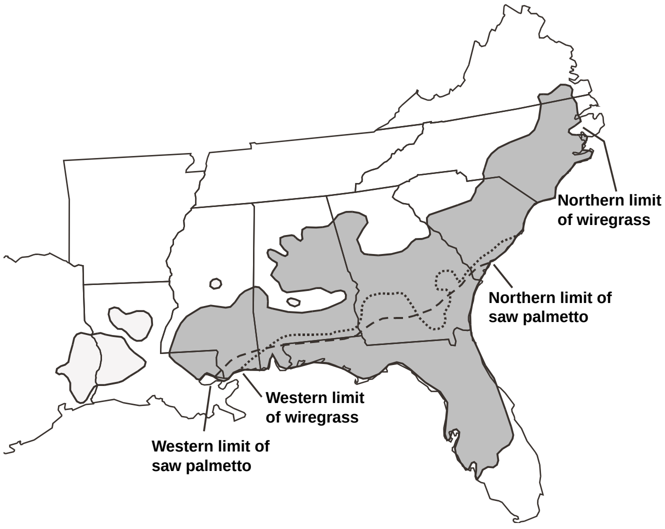

Figure 4-9—The longleaf pine forest region (after Stout and Marion 1993). The shaded area shows the distribution of longleaf pine (Critchfield and Little 1966). A dotted line indicates the landward extent of wiregrass and the western and northern limits of its distribution. A dashed line indicates the 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.

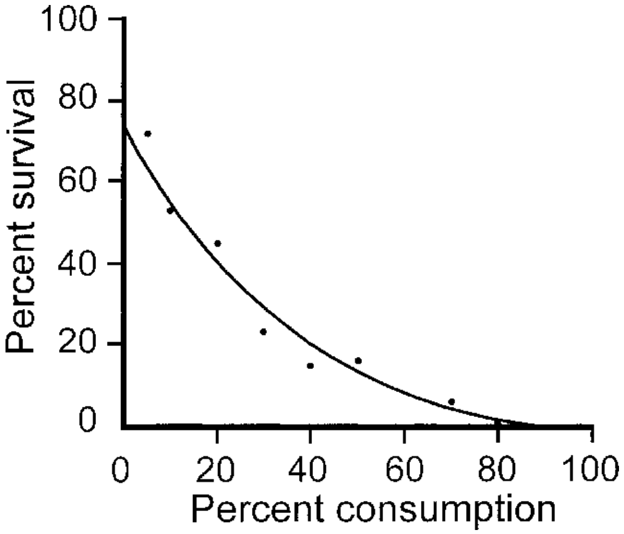

Figure 4-12—Percent survival of 1 to 8 year old planted loblolly pine after the first postfire year relates to percentage of live foliage consumption (Wade 1985).

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).

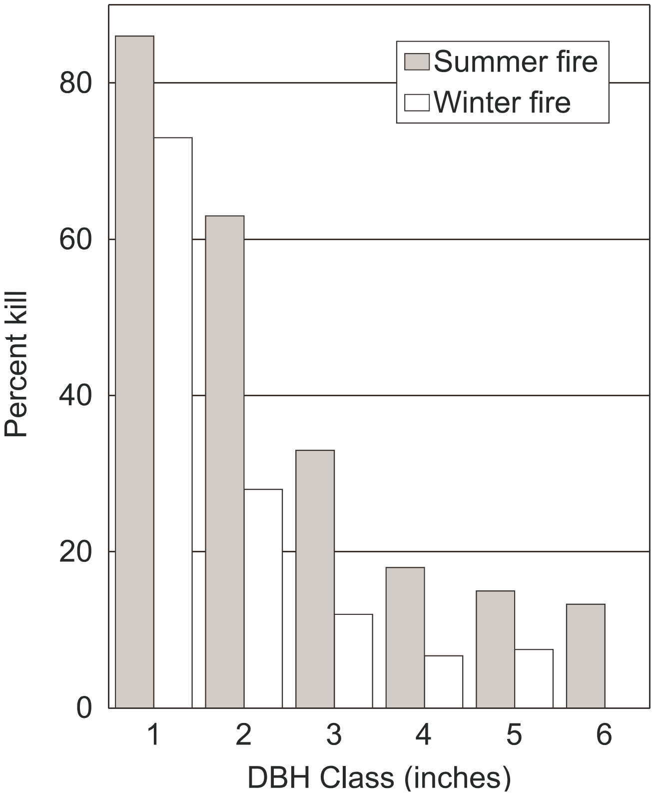

Figure 4-14—Growing-season prescribed fires topkilled more hardwoods and larger ones than dormant-season fires. Data collected on Brender Demonstration Forest, Georgia.

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

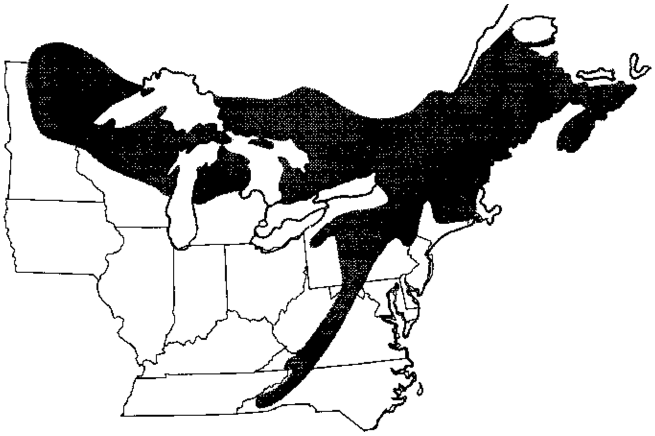

Figure 4-16—The extent of northern hardwood forests in northeastern North America. Redrafted from Bormann and Likens (1979).

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).

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.

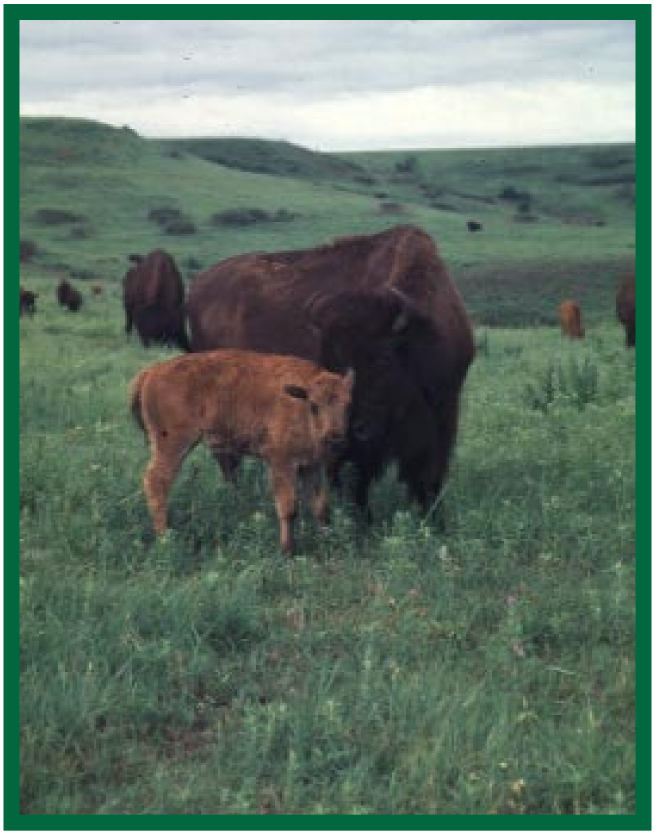

Figure 4-20—Grazers are attracted to freshly burned areas because of increased forage quality, Konza Prairie Biological Station, Riley County Kansas. Fire management can control the distribution of grazers on the landscape. Photo by Greg Hoch.

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						Nonfire
			Understory		Mixed		Stand-replacement		
			Occur[a]	Freq[b]	Occur	Freq	Occur	Freq	
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).

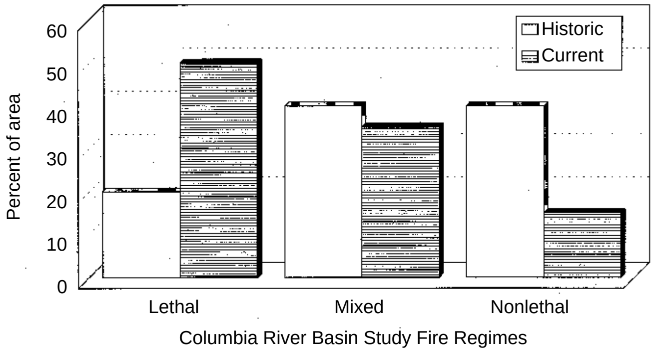

Figure 5-1—Change in fire severity between historic (pre-1900) and current conditions for Forest Service and Bureau of Land Management administered forested potential vegetation groups in the Interior Columbia Basin and portions of the Klamath and Great Basins (Quigley and others 1996).

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

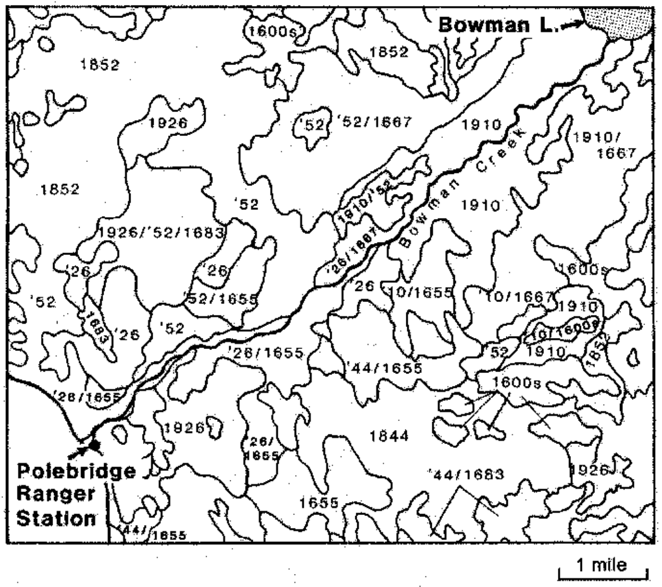

Figure 5-6—This mosaic of age-classes resulted from a mixed-severity fire regime in Glacier National Park, Montana. Dates indicate fire years that resulted in establishment of seral western larch and lodgepole pine age classes. In addition, some low-intensity underburns were detected in fire scars, but 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.

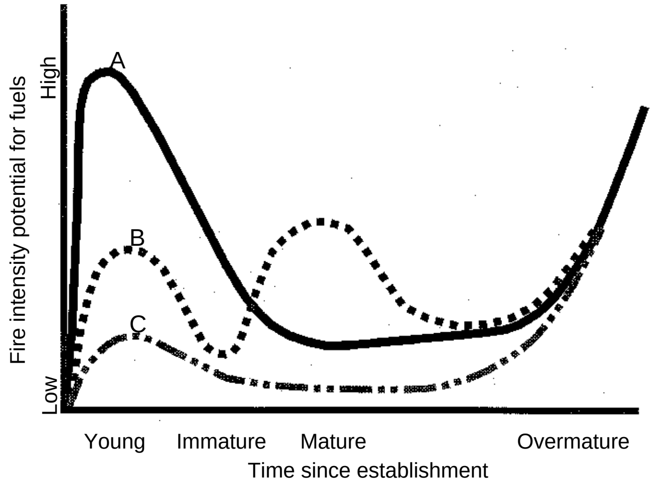

Figure 5-9—Fuel cycles and fire intensity potential in a lodgepole pine stand-replacement fire regime (Brown 1975).

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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Chapter 6:
Fire in Western Shrubland, Woodland, and Grassland Ecosystems

Western shrubland, woodland, and grassland ecosystems lie west of the eastern humid temperate zone, which begins a short distance east of the 100th meridian. Shrublands include sagebrush, desert shrub, southwestern shrub steppe, Texas savanna, and chaparral-mountain shrub ecosystem types. Woodlands include southwestern ponderosa pine, pinyon-juniper, and oak types that at times can be considered either forests or woodlands. The woodland/forest dichotomy can depend on phase of stand development and on the realization of natural site conditions that can form savannas with tree overstories. Grasslands include plains, mountain, desert, and annual grassland ecosystems (table 6-1).

Understory Fire Regimes

Major Vegetation Types

Southwestern United States ponderosa pine consists of two varieties: (1) interior ponderosa pine found over most of Arizona and New Mexico, and (2) Arizona pine found in the mountains of extreme southwestern New Mexico and southeastern Arizona, and extending into northern Mexico (Little 1979) (fig. 6-1, 6-2). Based on stand physiognomy (as in Paysen and others 1982), many stands of this vegetation type can be considered woodlands (relatively open grown), and many are classical closed forests. Differences may be due to inherent site conditions or to expressions of a developmental phase; fire frequency seems to play an important role as well.

Fire Regime Characteristics

Fires were frequent and of low intensity. Light surface fires burned at intervals averaging less than 10 years and as often as every 2 years (Dieterich 1980; Weaver 1951). The short fire-interval was caused by warm, dry weather common to the Southwest in early summer, the continuity of grass and pine needles, and the high incidence of lightning. Two fire seasons usually occurred each year, a major fire season after snow melt and just before the monsoon season in midsummer

Table 6-1—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
Ponderosa pine 21	SW ponderosa pine[c]	Interior ponderosa pine 237	M	1a:2-10	m	1			
	Arizona pine forest K019		M	1a:2-10	m	1			
	Pine-cypress forest K009	Arizona cypress 240			M	1,2	m	1	
Pinyon-juniper 35	Juniper-pinyon K023	Rocky Mountain juniper 220			M	1			
	Juniper-steppe K024	Western juniper 238			M	1			
		Pinyon-juniper 239			M	1			
		Arizona cypress 240			M	1			
Southwestern oaks[d]	California oakwoods K030	Canyon live oak 249			M	1			
		California coast live oak 255			M	1			
		California black oak 246			M	1			
		Blue oak-digger pine 250	M	1	M	1			
	Oak-juniper K031	Interior live oak 241			M	1			
Shinnery 31	Shinnery K071	Mohrs oak 67			M	1			
Texas savanna 32	Ceniza shrub K045				M	1			
	Mesquite savanna K060	Mesquite 68, 242			M	1			
	Mesquite-acacia savanna K061				M	1			
	Mesquite-live oak savanna K062	Western live oak 241			M	1			
	Juniper-oak savanna K086	Ashe juniper 66			M	1			
	Mesquite-oak savanna K087				M	1			
Sagebrush 29	Sagebrush steppe K055						M	2a:20-70	
	Juniper steppe K024	Rocky Mountain juniper 220					M	2a	
	Great basin sagebrush K038	Western juniper 238					M	2a:20-70	
	Wheatgrass-needlegrass shrubsteppe K056						M	2a	
Desert shrub 30	Mesquite bosques K027	Mesquite 68, 242					M	1,2a	
	Blackbrush K039						M	1,2a	
	Saltbrush-greasewood K040						M	1,2a	
	Creosotebush K041						M	1,2a	
	Creosotebush-bursage K042						M	1,2a	
	Paloverde-cactus shrub K043						M	1,2a	
	Cresotebush-tarbush K044						M	1,2a	
SW shrubsteppe 33	Grama-tobosa K058						M	1,2a	
	Trans-pecos shrub savanna K059						M	1,2a	
Chaparral-Mountain shrub 34	Oak-juniper woodland K031						M	1,2a	

(con.)

Table 6-1—Con.

			Fire regime types						
			Understory		Mixed		Stand-replacement		
FRES	**Kuchler**	**SAF**	**Occur**[a]	**Freq**[b]	**Occur**	**Freq**	**Occur**	**Freq**	**Nonfire**
	Mountain mahogany-oak scrub K037						M	1,2a	
	Transition of K031 & K037						M	1,2a	
	Chaparral K033						M	1,2a	
	Montane chaparral K034						M	1,2a	
	Coastal sagebrush K035						M	1,2a	
Plains grasslands 38	Grama-needlegrass-wheatgrass K064						M	1	
	Grama-buffalograss K065						M	1	
	Wheatgrass-needlegrass K066						M	1	
	Wheatgrass-bluestem-needlegrass K067						M	1	
	Wheatgrass-grama-buffalograss K068						M	1	
	Bluestem-grama prairie K069						M	1	
	Mesquite-buffalograss K085	Mesquite 68, 242					M	1	
Desert grasslands 40	Grama-galleta steppe K053						M	1,2a	
	Grama-tobosa prairie K054						M	1,2a	
	Galleta-three-awn shrubsteppe K057						M	1,2a	
Annual grasslands 42	California steppe K048						M	1,2a	
Mountain grasslands 36	Fescue-oatgrass K047						M	1	
	Fescue-wheatgrass K050						M	1	
	Wheatgrass-bluegrass K051						M	1	
	Foothills prairie K063						M	1a	
	Cheatgrass[c]								

[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 was not defined by Kuchler.

[d] Added subdivision of FRES.

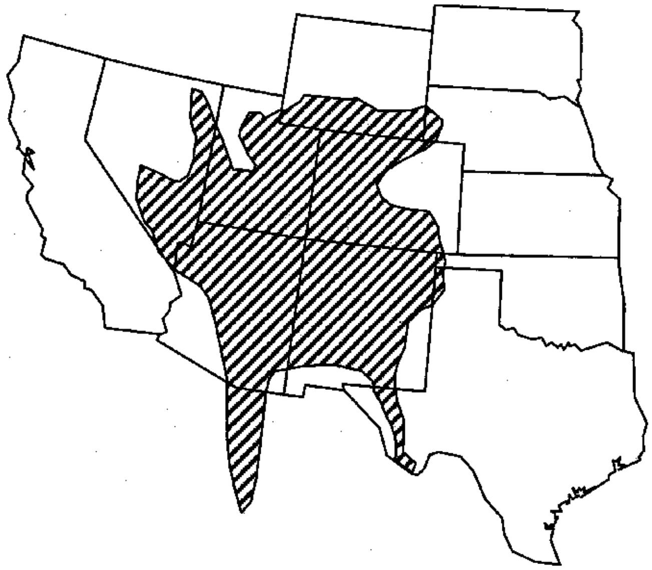

Figure 6-1—Southwestern ponderosa pine distribution.

and a secondary season in the fall. Once a fire started, the forest floor was generally consumed, but the damage to trees was highly variable. Low intensity surface fires predominated and were probably large where dry forests and adjacent grasslands were extensive such as on the gentle topography of high plateaus in Arizona and New Mexico. Damage to trees was highly variable but mortality to overstory trees was generally minor.

Fuels

The structural and compositional changes in Southwestern ponderosa pine over the past 100 years or more have been repeatedly documented (Biswell and others 1973; Brown and Davis 1973; Cooper 1960). What was once an open, parklike ecosystem, maintained by frequent, low-intensity fires, is now a crowded, stagnated forest. In addition to stand changes, general fire absence has led to uncharacteristically large accumulations of surface and ground fuels (Kallander 1969).

The natural accumulation of pine needles and woody fuels is exacerbated by the slow decomposition rates characteristic of the dry, Southwestern climate (Harrington and Sackett 1992). Decomposition rate (k) (Jenny and others 1949) is the ratio of steady state forest floor weight to the annual accumulation weight. Harrington and Sackett (1992) determined k values of 0.074, 0.059, and 0.048 for sapling thickets, pole stands, and mature old-growth groves, respectively. Decomposition rates this slow, which Olson (1963) considers quite low, border on desertlike conditions. Humid, tropic conditions would have k values approaching 1.0 where decomposition occurs in the same year as the material is dropped on the ground.

Fuel loading estimates can be obtained from predictions based on timber sale surveys (Brown and others 1977; Wade 1969; Wendel 1960) and using Brown's (1974) planar intersect method for naturally accumulated downed woody material. Forest floor weights

Figure 6-2—Typical Southwestern ponderosa pine fuels near Flagstaff, Arizona.

have been studied extensively in Arizona and New Mexico; results show high variability between sites. Ffolliott and others (1968, 1976, 1977), Aldon (1968), and Clary and Ffolliott (1969) studied forest floor weights in conjunction with water retention on some Arizona watersheds. These and other works included prediction equations relating forest floor weight to stand basal area (Ffolliott and others 1968, 1976, 1977), age (Aldon 1968), height and diameter (Sackett and Haase 1991), and forest floor depth (Harrington 1986; Sackett 1985).

The forest floor consists of a litter (L) layer, recently cast organic material; a fermentation (F) layer, material starting to discolor and break down because of weather and microbial activity; and the humus (H) layer, where decomposition has advanced. The loosely packed L layer and upper portion of the F layer provide the highly combustible surface fuel for flaming combustion and extreme fire behavior during fire weather watches and red flag warnings (fig. 6-3). The lower, more dense part of the F layer and the H layer make up the ground fuel that generally burns as glowing combustion.

Forest floor fuels (L, F, and H layers including woody material ≤1 inch diameter) were sampled in 62 stands in Arizona during the 1970s in Arizona and New Mexico (Sackett 1979). Throughout the Southwest, unmanaged stands of ponderosa pine had from 4.8 tons/acre (10.8 t/ha) in a stand on the Tonto National Forest to more than 20 tons/acre (45 t/ha) in a stand on the north rim of the Grand Canyon National Park. The next two heaviest weights (18.3 and 18.0 tons/acre) also occurred on the north rim of the Grand Canyon. Mean forest floor loading for the entire 62 stands measured was 12.5 tons/acre (28.0 t/ha). When woody material greater than 1 inch diameter was added, the average increased to 21.7 tons/acre (48.6 t/ha). The heavier material does not have much to do with extreme fire behavior, except as a spotting potential; these fuels do contribute to localized severity when burned. A range of forest floor fuel loadings is summarized in table 6-2.

Of the 12.5 tons/acre (28.0 t/ha) average of forest floor fuel load found in the Southwest, about 1.0 ton/acre (2.2 t/ha) was L layer material, 3.8 tons/acre (8.5 t/ha) was in the F layer, and 6.1 tons/acre (13.7 t/ha) was H layer. Small diameter woody material and other material comprised the remaining 1.8 tons/acre (4.0 t/ha). The large woody material that accounted for 42 percent of the total fuel loading, consisted of 1.4 tons/acre (3.1 t/ha) of material 1 to 3 inches (2.5 to 7.6 cm) in diameter, 5.0 tons/acre (11.2 t/ha) of rotted woody material 3+ inches in diameter, and 2.8 tons/acre (6.3 t/ha) of sound wood 3+ inches in diameter. See Sackett (1979) for complete summary.

Not only is there wide variation from site to site in the Southwestern ponderosa pine ecosystem, but vast differences exist within stands with respect to overstory characteristics (Sackett and Haase 1996). Experience indicates four separate conditions: sapling (doghair) thickets, pole stands, mature old growth (yellow pine) groves, and open areas in the groves

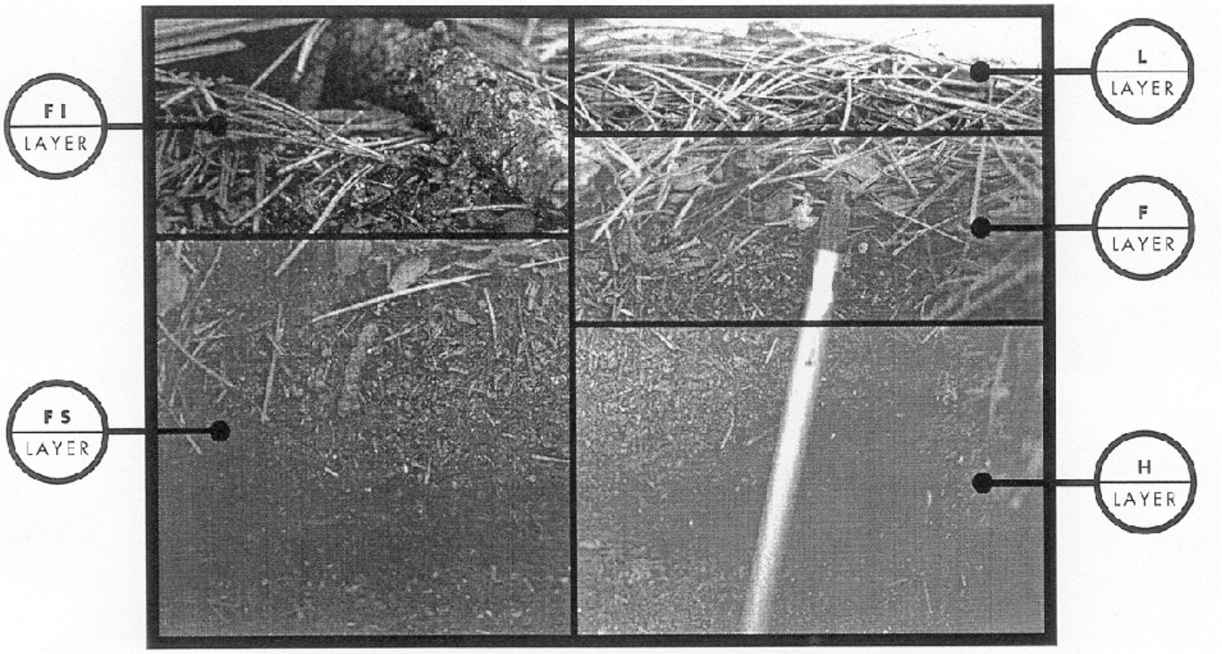

Figure 6-3—Section of ponderosa pine forest floor showing the fire intensity (FI) layer of fuel and fire severity (FS) layer of fuel in relation to the L, F, and H layers of the forest floor.

Table 6-2—Average ponderosa pine surface fuel loadings (ton/acre) in the Southwestern United States by location (Sackett 1997).

Location	Number of sites	Forest floor and 0 to 1 inch diameter wood	Woody fuel >1-inch diameter	Total fuel
Kaibab NF	4	15.5	8.6	24.1
Grand Canyon NP	4	17.5	5.6	23.1
Coconino NF	4	14.7	19.8	34.5
Tonto NF	2	6.5	2.7	9.2
Apache-Sitgreave NF	14	11.3	11.2	22.5
San Carlos Apache IR	3	14.4	8.4	22.8
Fort Apache IR	2	15.1	20.5	35.6
Gila NF	10	11.2	7.3	18.5
Navajo IR	1	9.4	4.9	14.3
Cibola NF	3	8.8	8.8	17.6
Santa Fe NF	3	13.2	14.6	27.8
Carson NF	4	13.3	4.3	17.6
Bandalier NM	1	11.6	3.0	14.6
Lincoln NF	2	13.9	7.1	21.0
San Juan NF	5	11.9	4.8	16.7

without crowns overhead. Sapling thickets produce as much as 1.1 tons/acre per year of litter and woody fuels, pole stands 1.5 tons/acre per year, and mature, old-growth groves as much as 2.1 tons/acre per year. A substantial amount of forest floor material remains after an area is initially burned (Sackett and Haase 1996). The amount remaining varies due to the original fuel's configuration and the fire intensity and behavior, which are affected by the overstory condition. This amount persists even with repeat applications of fire. The charred condition of the remaining forest floor material resists re-ignition from the newly cast needles that are consumed quickly.

Postfire Plant Communities

Southwestern Ponderosa Pine

Pre-1900 Succession—Chronicles from 19th century explorers, scientists, and soldiers described a forest type quite different than what is seen today. The open presettlement stands, characterized by well-spaced older trees and sparse pockets of younger trees, had vigorous and abundant herbaceous vegetation (Biswell and others 1973; Brown and Davis 1973; Cooper 1960). Naturally ignited fires burning on a frequent, regular basis in light surface fuels of grass and pine needles maintained these forest conditions. Light surface fuels built up sufficiently with the rapid resprouting of grasses and the abundant annual pine needle cast. Large woody fuels in the form of branches or tree boles, which fall infrequently, rarely accumulated over a large area. When they were present, subsequent fires generally consumed them, reducing grass competition and creating mineral soil seedbeds, which favored ponderosa pine seedling establishment (Cooper 1960). These effects created an uneven-age stand structure composed of small, relatively even-aged groups.

The decline of the natural fire regime in these ecosystems started with extensive livestock grazing in the late 19th century when fine surface grass fuels were reduced (Faulk 1970). Subsequently, pine regeneration increased because of reduced understory competition, less fire mortality, and more mineral seedbeds (Cooper 1960).

Post-1900 Succession—In the early 1900s forest practices, and reduced incidence of fire, led indirectly to stagnation of naturally regenerated stands and unprecedented fuel accumulation (Biswell and others 1973). Stand-stagnation exists on tens of thousands of acres throughout the Southwest (Cooper 1960; Schubert 1974) and still persists where natural or artificial thinning has not taken place.

For several decades, trees of all sizes have been showing signs of stress with generally poor vigor and reduced growth rates (Cooper 1960; Weaver 1951). This condition is likely due to reduced availability of soil moisture caused by intense competition and by moisture retention in the thick forest floor (Clary and Ffolliott 1969). Thick forest floors also indicate that soil nutrients, especially nitrogen, may be limiting because they are bound in unavailable forms (Covington and Sackett 1984, 1992).

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 drastic increase in high severity wildfires in recent decades

(Biswell and others 1973; Harrington 1982). Fire report summaries (Sackett and others 1996) show a great increase in the number of acres burned by wildfire since 1970 (fig. 6-4). Of all the years since 1915 with over 100,000 acres burned, almost 70 percent occurred between 1970 and 1990.

Another characteristic of today's Southwestern ponderosa pine stands is the sparseness of the understory vegetation, including pine regeneration. The thick organic layers and dense pine canopies have suppressed shrubby and herbaceous vegetation (Arnold 1950; Biswell 1973; Clary and others 1968). Natural regeneration is also limited to areas where the forest floor material has been removed either by fire or by mechanical means (Sackett 1984; Haase 1981). This condition has reduced the wildlife, range, and timber values of these forests and has generally minimized biodiversity.

Management Considerations—The need to alleviate the stagnated and hazardous forest conditions is a primary consideration in the management of Southwestern ponderosa pine stands. The restoration of forest health to the Southwest also needs to address the following concerns:

- Dwarf mistletoe, once held in check by periodic fires, is now a major cause of mortality in localized areas.
- Bark beetle outbreaks are evident in overstocked stands that are stressed from the high competition for limited soil moisture, especially during drought years.
- Some amount of fire injury to the overstory is almost assured from the application of fire into an area. This may be in the form of crown scorch to the smaller trees and belowground injury to roots and root collars of the larger trees.
- Fuel conditions that contribute to the elimination of whole stands from wildfire need to be reduced. These conditions include heavy forest floor accumulations and ladder fuel conditions created from dense, stagnated sapling thickets.

Although the extent of these conditions will vary throughout the region, the combination of any of these situations on a particular forest creates a major concern and problem for the manager. Forest management objectives in the Southwest need to include the maintenance or improvement of existing old-growth stands and actions that promote the creation of future old growth stands.

Because recurrent fire was a primary element in sustaining presettlement forest health leading to the establishment and maintenance of old-growth stands, its use should be emphasized when restoring favorable conditions for ancient pine development. These conditions include low levels of dead organic material (fuels) to lessen the potential of high fire intensity and severity, and open stand structure to reduce crown fire

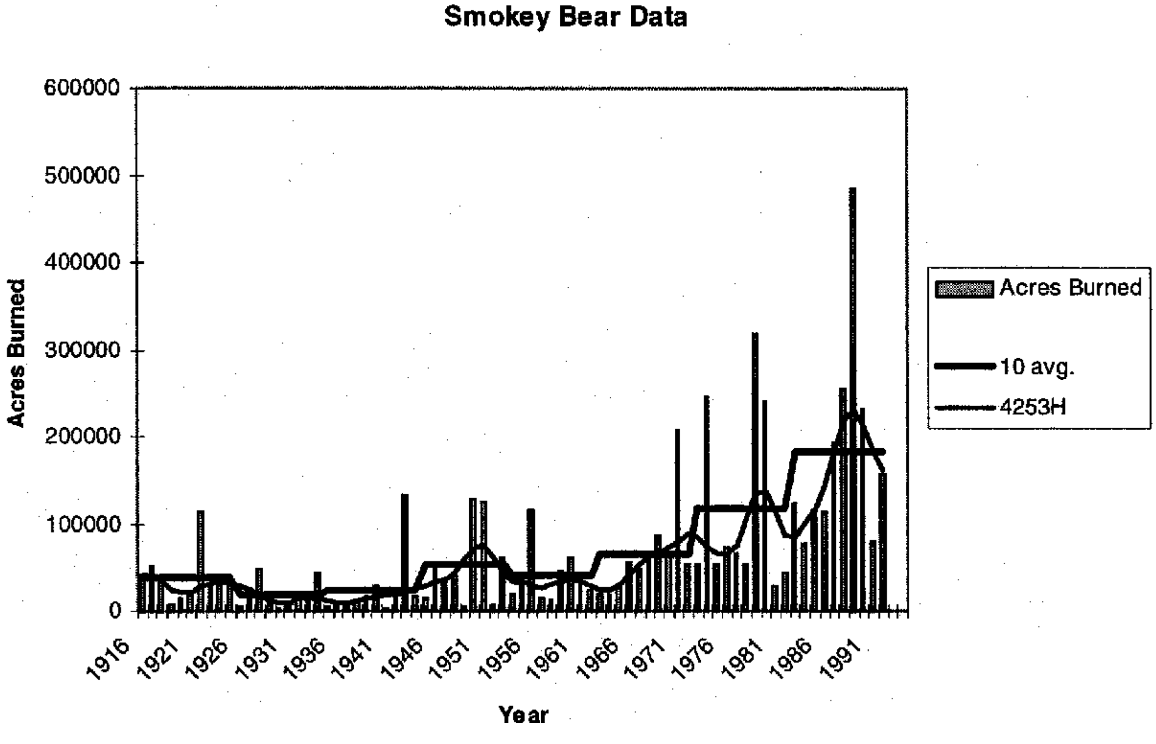

Figure 6-4—The total number of acres burned by wildfires in Arizona and New Mexico from 1916 to 1990, USDA Forest Service Smokey Bear fire summary reports. Heavy line represents 10-year average; light line represents trends using the 4253H mathematical filter, used for smoothing noise in data.

potential and intraspecific competition. Fire can be used to reduce fuel hazard, but its success is temporary. Failures denoted by too little or too much fuel consumption generally result from improper burn prescriptions and by attempting to correct long-term fuel buildup with one treatment. Cooper (1960) questioned whether prescribed fire could be used in the restoration of deteriorated forests. He concluded that planned burning would be too conservative and accomplish little, or would destroy the stand. While this observation has merit, with refined burning techniques as described in Harrington and Sackett (1990), it appears that fire could be applied sequentially to relieve the fuel and stand density condition. However, it is apparent that considerable large tree mortality could result. This seems to be an inescapable cost dictated by years of forest degradation.

Because of these consequences, special attention should be given to the excessive buildup of forest floor fuels in present old-growth sites. Burning of these deep forest floor layers can mortally injure the roots and cambiums of old pines, which previously survived many fires (Sackett and Haase 1996). Options for alleviating this condition are not ideal. Managers could 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. Alternatively, the heavy accumulation of fuels could be manually removed from around the root-collar of the old-growth trees before the fire is applied. Currently, methods are being investigated that will make this mitigation method a feasible option for managers. The use of a burn prescription that removes a portion of the fuel accumulation has not been found for prescribed burning in the Southwest. If glowing combustion is able to begin in the deeper accumulations of material, high moisture content of that material may not prevent total consumption of the forest floor. Nearly complete burnout of duff has been observed in ponderosa pine forests at moisture contents up to 90 percent (Harrington and Sackett 1990) and in mixed conifers up to 218 percent (Haase and Sackett 1998).

In forest regions where old-growth pine groups are absent, designated areas based on site quality and existing stand types should be selected for creating future old growth. The best growing sites should be chosen because old-growth characteristics would be achieved more expeditiously than on poor sites. Moir and Dieterich (1990) suggested that 150- to 200-year-old ponderosa pine (blackjack pine) in open stands with no dwarf mistletoe be selected as the best stands to begin developing old growth. Through sequential silvicultural and fire treatments, the stands should be relieved of wildfire hazards and competition, allowing concentrated growth on a chosen group of trees. A long-term commitment is necessary, because another century may be needed before select old-growth pine is represented (Moir and Dieterich 1990). If younger stands are chosen for old-growth replacement, a greater commitment of time is required for thinning, slash disposal, commercial harvesting, and fire application.

Mixed Fire Regimes

Major Vegetation Types

The pinyon-juniper woodlands (fig. 6-5) cover approximately 47 million acres (19 million ha) in the Western United States (Evans 1988) and are characterized by a large number of diverse habitat types that vary in tree and herbaceous species composition, and stand densities. Climatic and physiographic conditions vary greatly within the range of this vegetation type. Pinyon-juniper woodlands in the United States are commonly divided into the Southwestern and the Great Basin woodland ecosystems based on species composition. True pinyon is common in the Southwest and is usually associated with one or several species of junipers, including one-seed, Utah, alligator, and Rocky Mountain junipers. Singleleaf pinyon is identified with the Great Basin and is generally associated with Utah juniper. Other species of pinyon occur in southern California, Arizona, south of the Mogollon Rim, along the United States-Mexico border, and in Texas (Bailey and Hawksworth 1988). Several other species of junipers also are found in the West; one of the more

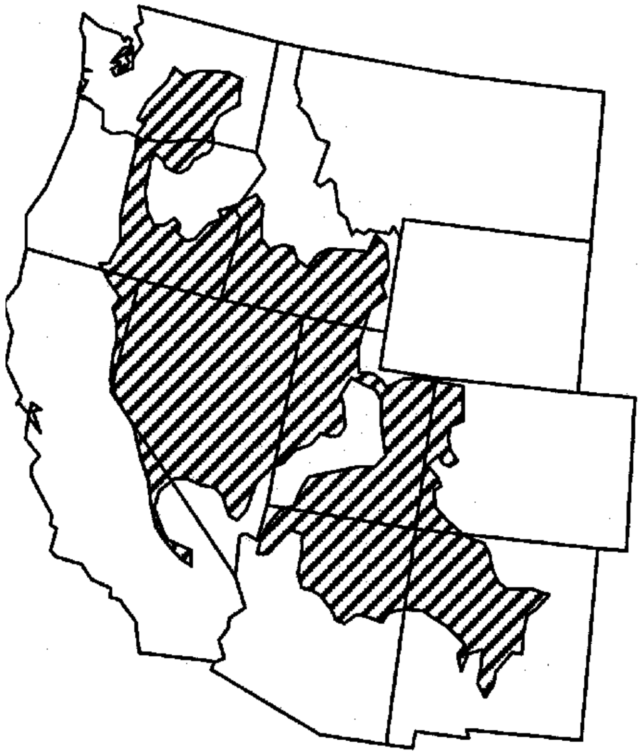

Figure 6-5—Pinyon-juniper woodlands distribution.

common is western juniper, which is found mainly in Oregon and eastern California. Stand densities and composition vary by elevation as it affects available moisture; drier sites tend to be occupied by junipers that are widely spaced and of low stature. Many of these sites are often classified as savannas. Higher elevation sites tend to be dominated by relatively dense stands of pinyon trees of comparatively tall stature and good form.

This report includes western oak species of obvious concern to resource managers but it does not include all oaks found in the Western United States (fig. 6-6, 6-7). Discussion concentrates on the important tree-form deciduous and live oaks of California and of the Southwestern United States (such as Gambel oak and Arizona white oak). These are generally addressed as a group. Little information has been documented for these species (McPherson 1992), but their importance to resource and fire management requires a beginning. Shinnery, predominantly composed of sand shinnery oak, is described as a separate ecosystem (fig. 6-8).

The Texas savanna (fig. 6-8) as a mapped ecosystem occupies major portions of the Rio Grande Plains of south Texas, the Edwards Plateau of south central Texas and portions of the Rolling Plains, Grand Prairie, North Central Prairies, Blackland Prairies, and Cross Timbers. It corresponds roughly with Sections 315C, D, and E of Bailey's Ecoregions and Subregions of the United States (Bailey and others 1994) and with major

Figure 6-6—Western oak distribution.

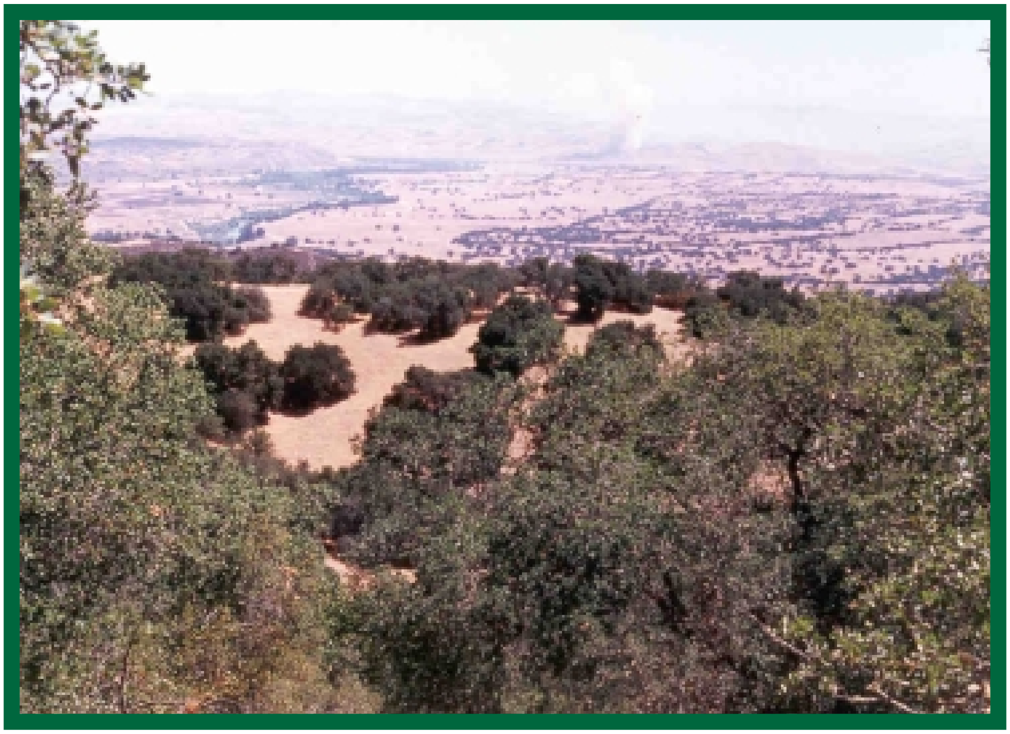

Figure 6-7—Western oak woodlands, Camp Roberts Military Training Reservation, Paso Robles, California.

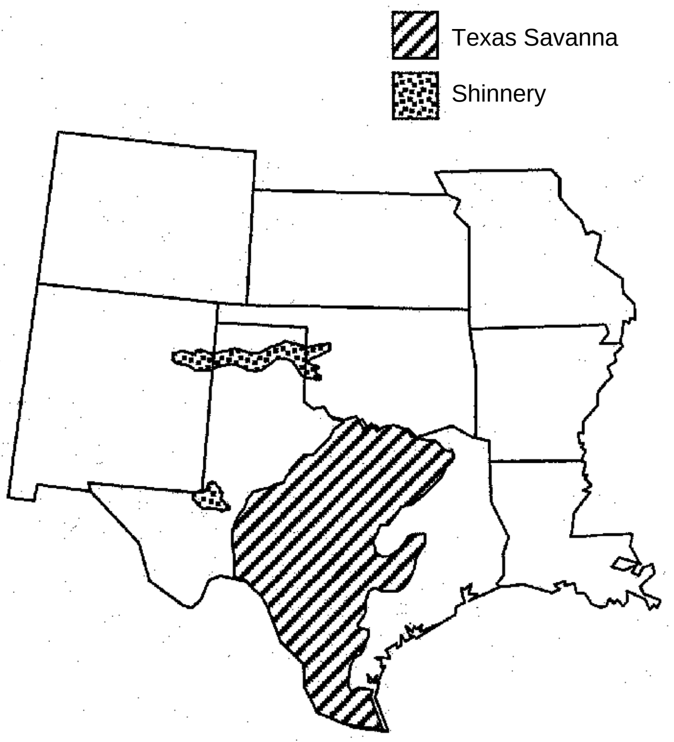

Figure 6-8—Distribution of shinnery and Texas savanna FRES ecosystems.

portions of the Rio Grande Plain and Edwards Plateau vegetation regions found in Box and Gould (1959). As a plant community type, however, it has significant representative elements that extend far north into the southern portion of the Plains Grasslands ecosystem. In fact, in the original Rainbow series volume, "Effects of Fire on Flora" (Lotan and others 1981), the area associated with the Texas savanna was lumped into one huge "Prairie Grasslands" type (which also included the grasslands of the "Great Valley" in California). The vegetation of the "Texas savanna" can be found in the northern portions of Texas in the southern Rolling Plains, the Grand Prairie, the North Central Prairies, the Blackland Prairies, and Cross Timbers, and extends to just south of the Texas Panhandle area (Box and Gould 1959). These other areas will be considered as part of the Texas savanna for the purposes of this publication. These areas receive 20 to 30 inches of precipitation annually—more than half of which falls during the warmest months, and less than a quarter during the period from December through March (Garrison and others 1977).

The vegetation is a savanna with an overstory layer of low trees and shrubs that varies from dense to open. This overstory is of variable composition, having broad-leaved and needle-leaved, deciduous and evergreen species that predominate. These are mesquite, acacias, oaks, junipers, ceniza, and prickly pear species. Honey mesquite is the most widespread woody plant in the Texas savanna type and will receive the most discussion. The grass of the Texas savanna varies from short (<2 inches) to medium-height (2 to 12 inches), and the herbaceous vegetation in general varies from dense to open. These understory plants are mainly bluestems, indiangrass, and switchgrass in the northeast, gramas, buffalograss, Texas wintergrass and *Sporobolus* spp. in the south, central, and northwest, and curly-mesquite and tobosagrass in the west and on the Edwards Plateau. The particular mix of vegetation or specific plant community that one might encounter seems to be well correlated with soil orders, which are variable in the Texas savanna system (Garrison and others 1977).

Fire Regime Characteristics

Long-term fire frequencies for the pinyon-juniper woodlands have not been clearly defined and are the topic of continuing study and discussion. However, there is agreement that fire was the most important natural disturbance before the introduction of livestock, particularly the large herds in the 19th century (Gottfried and others 1995). It is suspected that prior to the introduction of heavy livestock use, large areas of savanna and woodland periodically burned. These fires could have occurred during dry years that followed wet years when substantial herbaceous growth developed (Rogers and Vint 1987; Swetnam and Baisan 1996).

In the Intermountain West, presettlement mean fire intervals of less than 15 years were documented in the sagebrush steppe where western juniper now dominates (Miller and Rose 1999). Other knowledge that clearly documents the fire frequency, extent, and seasonality of long-term fire regimes was developed from a few studies at the upper limit of the pinyon-juniper type where it occurs with ponderosa pine. Fire scars are rare in living pinyon pines due to the tree's susceptibility to damage by fire or to rot fungi that enter resulting wounds. Fire scars have been noted on junipers but most members of this genus are difficult to age because of missing and false rings. Nonetheless, some fire frequencies have been determined for the Southwest. A sample of fire-scarred pinyon trees from three locations in the Sacramento Mountains in New Mexico indicated a mean fire interval of 28 years with a range of 10 to 49 years (Wilkinson 1997). Despain and Mosley (1990), working in the pinyon-juniper and ponderosa pine ecotone at Walnut Canyon National Monument in Arizona, reported a surface fire interval of approximately 20 to 30 years. Other studies by C. Allen and by T. Swetnam and his associates (Gottfried and others 1995), on productive sites in New Mexico, indicated that standwide fires, which covered more than 25 acres, occurred at 15 to 20 year intervals.

Dense pinyon-juniper stands (450 tree/acre or greater) can burn in crown fires under extreme weather conditions, generally low relative humidity and high wind speeds. The key conditions are a closed canopy to allow the spread of fire through the crowns and abundant dead material on the ground and as snags (Gottfried and others 1995). It appears that presettlement fire regimes in dense stands were a mixture of surface and crown fires, and that intensities and frequencies varied depending on site productivity. The Walnut Canyon site probably sustained patchy surface fires at intervals of 10 to 50 years and could carry crown fires at intervals of 200 to 300 years or longer.

On less productive sites with discontinuous grass cover, fires were probably infrequent and burns were small and patchy. Fire frequencies were probably greater than 100 years in these areas, but did occur more frequently under extreme conditions (Gottfried and others 1995). However, where grass cover was more continuous, fire frequencies were probably more frequent (10-year interval or less) and tended to maintain these sites as savannas or grasslands. Surface fires would kill oneseed juniper trees less than 3 to 4 feet (1 m) tall (Johnsen 1962) but would have less of an impact on older, larger trees that have thicker bark and high crown base heights that exceed flame lengths. This relationship between height and susceptibility to fire also has been observed in western juniper stands (Dealy 1990) and in Ashe juniper stands in Oklahoma (Wink and Wright 1973). Fast moving surface fires in the Southwest often do not burn near the trunks of larger trees because the litter layer does not ignite.

In the Great Basin, fire susceptibility depends on the stage of stand development (Meeuwig and others 1990). In young open stands, shrubs and herbaceous cover may be sufficient to carry fire, but this cover declines with time and eventually becomes too sparse as the trees develop. The trees, however, may still be too widely spaced to carry crown fires, except under severe conditions.

In recent centuries, fire regimes in Western oak forests were characterized by frequent, low intensity fires. This was probably due to use of these types by Native Americans, who probably carried out programs of frequent underburning. Higher intensity fires at long intervals have become more likely in the last half of the 20th century.

Few data are available on fire frequencies within the Texas savanna (Fuhlendorf and others 1996). With understory fuels usually exceeding 2,240 lb/acre (2,000 kg/ha) each year under undisturbed conditions, it is quite likely that fire frequencies were less than 10 years, and potentially more frequent in the northeast portion of the Texas savanna. Fires occur most frequently during February and March when most grasses are dormant and lightning strikes occur commonly, and from July to September when grasses are dry. Both winter and summer fires with ample fuel loading in the grass understory can topkill trees resulting in major alteration of the woody physiognomy. However, woody plant mortality and stand-replacements are rare. Winter fires that occur with low understory fuel loadings can result in partial removal of the overstory (Ansley and others 1995, 1996b). Species such as mesquite, redberry juniper, and live oak sprout if topkilled by fire and are rarely removed from the vegetation complex by fire. However, Ashe (or blueberry) juniper, which occurs in south-central Texas, can be killed by fire and replaced by herbaceous vegetation.

Fuels

Pinyon-Juniper—The main fuel consideration is the amount of fine fuels, which varies with habitat type, stand history, and climatic conditions. Fuel loading information for woody material is not readily available; however, Perry (1993) measured an average of 20 tons/acre (45 t/ha) after a pinyon-juniper clearcutting operation in Arizona; this stand produced about six cords/acre of fuelwood. Fuel loadings of more than 11 tons/acre are considered heavy. Slash left in partially harvested woodlands may provide fuel ladders for ground fires to spread into the canopies. Grass understory loadings can range from sparse to abundant (200 to 600 lb/acre). Typical crown fuels are 3.6 tons/acre (8.1 t/ha) for foliage and 1.8 tons/acre (4.0 t/ha) for 0 to 0.25 inch branchwood (Reinhardt and others 1997).

Western Oaks—Fuels are quite variable between stands, depending upon species, site, and stand condition. For example, a closed-canopy canyon live oak forest may have little or no live understory. Surface fuels will be made up of leaf and branch litter and the amount will depend upon the time since last fire in the stand. A more open stand may have an understory of shrubs and nonwoody species. A closed forest of a deciduous species, for example California black oak, may well have an understory of annual grass; but a more open woodland of the same species may have a mix of grass and shrubs as an understory. In the latter case, the combination of grass and shrubs can provide a fuel ladder complex with associated erratic and potentially dangerous fire behavior.

The aerial fuels in these oak stands are variable too. Little information exists to characterize the deciduous species; however, the live oaks can be thought of as roughly comparable to chaparral in terms of crown fuel character—both being sclerophyllous in nature. The green material in these species will burn if fuel moisture is low enough.

Texas Savanna—The predominant fuel that contributes to a fire's propagation is the herbaceous understory. However, if the mesquite overstory has dead stem material, it can be ignited and potentially kill the plant. Britton and Wright (1971) found that up to 24 percent of mesquite that had been sprayed with a topkilling herbicide were killed with fire that occurred 4 years after spraying. The standing dead stems burned into live root crowns. When the overstory is dense—either from a high density of individuals, or from dense resprouted material—a crown fire can be sustained, given the necessary wind and moisture conditions. Such a high density overstory can be found as a phase in Texas savanna stands. Mesquite crown fires would only occur in summer months because the plant is winter deciduous. However, other species of the savanna complex, such as junipers and live oak, could carry crown fire any time of the year.

Herbage production, which indicates potential fine fuel loading in the understory, was divided into four major productivity classes (Garrison and others 1977):

Class	Productivity (lb/acre)
1	2,250 to 3,000+
2	1,500 to 2,250
3	750 to 1500
4	0 to 750

Postfire Plant Communities

Pinyon-Juniper

Pre-1900 Succession—The pinyon-juniper woodlands are diverse, and successional pathways differ by habitat type throughout the West. Traditional succession toward a "climax" vegetation considers the continuous replacement of one community by another. The driving force in the successional process is competition among plant species of different genetically controlled capabilities responding to changes in the environment (Evans 1988). In the woodlands, succession involves the same species but in different amounts and dominance over the landscape. Several successional seres following stand replacing fires have been proposed for the Southwestern or Great Basin pinyon-juniper woodlands. Most of the successional projections are based on stands that had been grazed in the past. Arnold and others (1964), working in northern Arizona, developed one of the first models. A model for southwestern Colorado (Erdman 1970), similar to that of Arnold and others (1964), progresses from skeleton forest and bare ground, to annual stage, to perennial grass-forb stage, to shrub stage, to shrub-open tree stage, to climax woodland. This pattern takes approximately 300 years; however, new fires could set back succession before the climax is achieved. Arnold and others (1964) indicated that tree reoccupation progressed from the unburned stand inward toward the center of the burn. Barney and Frischknecht (1974) reported a sere for a Utah juniper stand in west-central Utah where pinyon was a minor component.

This ecosystem has had a long history of heavy grazing since the late 19th century. The postfire progression went from skeleton forest and bare ground, to annual stage, to perennial grass-forb stage, to perennial grass-forb-shrub stage, to perennial grass-forb-shrub-young juniper stage to shrub-juniper stage, and to juniper woodland. Junipers were well developed 85 to 90 years after a fire. They indicated that the speed of tree recovery would depend on the stage of tree maturity at the time of the fire; older seed producing stands would recover more rapidly than younger, immature stands. They noted the importance of animal transport and storage of juniper seeds in the speed of tree recovery. A new juniper could start producing seed within about 33 years of establishment, hastening tree recovery.

Post 1900 Succession—Data on successional trends apparent in the 1900s show that on similar sites succession may follow several pathways (Everett 1987a; Everett and Ward 1984). Shrubs, rather than annuals, have been the initial vegetation on some burned sites (Everett and Ward 1984), while the shrub stage may be reduced or absent on some New Mexico sites (Pieper and Wittie 1990). Predicting the course of succession is difficult since it depends on a number of factors (Everett 1987a). Specific successional pathways depend on fire severity and related damage to the original vegetation, area burned, available seed sources either in the soil or from adjacent areas, species fire resistance and ability to reproduce vegetatively, site conditions, and climatic parameters throughout the successional process. Everett and Ward (1984) indicated that the "initial floristic model" is appropriate after a burn; initial species composition and density may be as or more important than the progressive succession. Most preburn species returned within 5 years of a prescribed burn in Nevada (Everett and Ward 1984) and in southern Idaho (Bunting 1984).

The major human influence on the pinyon-juniper woodlands and fire's role in these ecosystems has been ranching. Most of the Western rangelands were overgrazed, especially in the period following the 1880s. Some areas around the Spanish controlled areas of New Mexico have been heavily grazed since the 16th century. Overgrazing has had an important effect on the role of fire in the woodlands. The reduction of cover of herbaceous species resulted in insufficient fuels for fires to spread and to control tree establishment. Fires ignited by lightning or humans tend to be restricted in space. Fire suppression activities by land management agencies also reduced the occurrence of fires.

Woodland and savanna stand densities have increased throughout most of the West. Some people believe that the woodlands have invaded true grasslands because of the lack of fire, but this is open to debate (Gottfried and Severson 1993; Gottfried and others 1995; Johnsen 1962; Wright and others 1979). Climatic fluctuations, such as the drought in the Southwest in the early 1950s, and global climate change also have affected the distribution of woodlands in the West. In the Intermountain West, Miller and Rose (1999) quantitatively established that the co-occurrence of wet climatic conditions, introduction of livestock, and reduced role of fire contributed to the postsettlement expansion of western juniper. Prior to 1880, fire was probably the major limitation to juniper encroachment. Other human influences related to the harvesting of wood products by early American Indians (Gottfried and others 1995) and the harvesting of large quantities of fuelwood to make charcoal for the mines and domestic wood for supporting populations in Nevada (Evans 1988) and near Tombstone in Arizona.

Management Considerations—During the 1950s and 1960s, large operations were conducted to eliminate the pinyon-juniper cover in the hope of increasing forage production for livestock (Gottfried and Severson 1993; Gottfried and others 1995). Other objectives were to improve watershed condition and wildlife habitat. Mechanical methods, such as chaining and cabling, were used and resulting slash was piled and burned. Burning these large fuel concentrations generated high heat levels that damaged soil and site productivity (Tiedemann 1987). Many of these piled areas were sterilized and remain free of vegetation after over 20 years. Individual tree burning was used on some woodland areas. Most of the control operations failed to meet their objectives. Many areas failed to develop sufficient herbaceous cover to support renewed periodic surface fires.

A relatively undisturbed site with a rich variety of understory species may recover differently than an abused site with little understory development. Similarly, an older stand of junipers with a less diverse population of perennial species will recover differently than a younger stand (Bunting 1984). Burning in stands with few desirable understory species may worsen the ground cover situation, and depending on the characteristics of the tree component, destroy a valuable wood resource (Everett 1987b). A potential problem exists if the preburn or adjacent vegetation contains undesirable species, such as red brome. Very hot fires can seriously slow initial succession of desirable species (Bunting 1984). Everett and Ward (1984) indicated that relay floristics, the migration of species into the site, is more important for the later stages of development. Wink and Wright (1973) found that soil moisture was important in determining rate of understory recovery; it is more rapid when soil moistures are high. Dry conditions may increase drought stress of surviving herbaceous plants (Wink and Wright 1973) and retard seed germination. Aspect and elevation can be used to predict some general successional trends (Everett 1987a).

Currently, prescribed fire is used to reduce accumulations of slash from fuelwood harvesting or to reduce or eliminate the tree cover in an attempt to increase range productivity and biodiversity. In Arizona, slash is usually left unpiled. Small piles are constructed occasionally and are burned as conditions and crew availability allows. There is increasing interest in managing the pinyon-juniper woodlands for sustained multi-resource benefits including, but not limited to, tree products, forage, wildlife habitat, and watershed protection (Gottfried and Severson 1993). This is particularly true for high site lands that have the ability to produce wood products on a sustainable basis. Prescribed burning to dispose of slash is less desirable in partially harvested stands, where the selection or shelterwood methods have been used to sustain tree product production. Burning tends to damage residual trees, especially where slash has accumulated at the base, and advance regeneration. Established, smaller trees are particularly important for the next rotation because of the difficulty of achieving adequate regeneration of these relatively slow growing species. It may be desirable to move slash away from areas of satisfactory regeneration prior to burning or to avoid burning in them.

Several different slash disposal options may be applicable to any one management area (Gottfried and Severson 1993). Burning of large piles is unacceptable because of soil site degradation (Tiedemann 1987) and no longer recommended in the Southwest (USDA Forest Service 1993). However, small piles of slash may be burned in low intensity fires to encourage floristic richness or to promote temporary increases of nutrient content in herbaceous vegetation. Piled or unpiled slash can also be left unburned to provide habitat for small mammals or to break up sight distances for wild ungulates. It also can be scattered to provide protection for establishment of young trees and herbaceous species, and to retard overland runoff and sediment movement.

Mechanical methods of clearing pinyon-juniper are increasingly expensive, but prescribed fire is an economical alternative. The method used in Arizona is to ignite the crowns from prepared fuel ladders of cut lower limbs that are piled around the base of the tree. Ladders are ignited one season after the limbs are cut. In denser stands, fire spreads into the crown layer and through the stand from fuel ladders that are created below strategically placed trees. A method used in central Oregon on sites converted to juniper from sagebrush/grass is to conduct prescribed fires several years after harvesting trees. The increased production of herbaceous vegetation following cutting provides fuels to carry the fire, which reduces residual slash and kills juniper seedlings.

Research in the Great Basin suggests that fire works best on sites with scattered trees (9 to 23 percent cover) where the trees begin to dominate the understory and in dense stands (24 to 35 percent cover) (Bruner and Klebenow 1979). Wright and others (1979) indicated that prescribed spring burning was successful in sagebrush/pinyon-juniper communities. Bruner and Klebenow (1979) recommended an index to determine if a fire will be successful or if conditions are too dangerous. This index is based on the addition of maximum wind speed (mi/hr), shrub and tree cover (percent), and air temperature (°F). Burning can be successful if scores are between 110 and 130. Dense stands where pinyon is more common than juniper are easier to burn than pure juniper stands (Wright and others 1979). Bunting (1984) indicated that burning of western juniper stands in southwestern Idaho was only successful during the mid-August to mid-September period; burning in the fall did not achieve desired results because of low temperatures, low wind speeds, and lack of fine fuels. Prescribed fire can be used in previously treated areas to control new tree regeneration. This technique works best if the area is ungrazed for one or two seasons prior to burning. Wink and Wright (1973) reported that a minimum of 890 lb/acre (1,000 kg/ha) of fine fuels is needed to burn and kill Ashe juniper seedlings and to burn piled slash. Success where alligator juniper dominates has been limited because of the trees' ability to sprout, so prescribed fire is not recommended (USDA Forest Service 1993).

Ecosystem Management—Reintroducing low intensity fire into the pinyon-juniper woodlands could help meet ecosystem management goals. For example, prescribed fire could be used after harvesting to limit tree regeneration and to maintain overstory stand densities that would promote vigorous understory vegetation for livestock and wildlife. Fire could be used during the earlier part of the rotation period, when crown cover is less, and modified later to protect adequate tree regeneration. The prescription would vary by the amount and condition of woody debris in the stand so that stand replacing crown fires are prevented. Pockets of regeneration could be protected.

Fire could also be used to maintain herbaceous cover dominance in natural savannas and ecotonal grasslands. However, as indicated above, all surface fire options would require that the land be rested from grazing prior to treatment so that sufficient fuels can develop to carry the fire. It usually requires 600 to 700 lb/acre (672 to 784 kg/ha) of fine fuel to carry a fire in the Great Basin (Wright and others 1979).

Fire has also been used to create mosaics of woodland and openings within some Southwestern landscapes. Mosaics are beneficial to wildlife and livestock (Gottfried and Severson 1993) and can create an aesthetically pleasing landscape. Aerial and ground firing techniques have resulted in mosaics on some juniper/mesquite grasslands in southern Arizona.

Western Oaks

Pre-1900 Succession—There is little doubt that western oak trees evolved over a time when climatic change was occurring and when disturbance including fire was common. The deciduous or evergreen habit probably is related to environmental moisture—evergreen oaks belonging to more arid systems (Caprio and Zwolinski 1992; Rundel 1987). Postfire succession during pre-Euro-American settlement was probably much like the dynamics that we see today, but there were probably more oaks than we find today. Some species were easily top-killed; many species sprouted in response to fire.

Post-1900 Succession—The current reduction in the occurrence of the oaks in many areas may be due to a number of factors, including increased fire severity, grazing, overt removal to provide more pasture land, and urban encroachment. Fire is probably not the primary factor, but it can kill a stand of oaks outright. Some oaks are more easily top-killed than others, which is generally a function of bark thickness. See the categorization of oak sensitivity to fire by Plumb and Gomez (1983). Almost all of the oak species sprout after fire, if root crown or underground portions are still alive (Plumb 1980).

Management Considerations—In some parts of the West, oaks have become subjects of intense resource management interest. The ranges of some species have become severely reduced; some species do not seem to be reproducing at a desired rate (Bartolome and others 1992). Competition to seedlings from understory vegetation may be hampering seedling survival (Adams and others 1992); grazing may play a part as well. Effective management of these species has yet to be established. The use of prescribed fire as a means of reducing competition and opening up closed canopy stands is being attempted (Clary and Tiedemann 1992). Although results are not definitive yet, it shows promise. For now, the use of prescribed fire in western oaks should be approached with caution and patience. Some species are sensitive to fire (table 2-1) but may survive under certain conditions (Paysen and Narog 1993). Many oaks seem to be prone to disease, such as heart rot. Injury from fire or other treatment may not kill a tree, but might conceivably inflict damage that could provide a port of entry for disease. Much research remains to be done on these species. For now, management treatments should be carried out carefully.

Texas Savanna

Pre-1900 Succession—Historical accounts differ as to original density and distribution of mesquite in Texas. Bartlett (1854) described much of Texas rangeland as open grasslands with scattered large mesquite (a mesquite savanna). Marcy (1866) described some upland areas of central Texas as "covered with groves of mesquite trees," and an area in the lower Texas Panhandle as "one continuous mesquite flat, dotted here and there with small patches of open prairie." These observations suggest that honey mesquite was a natural part of the northern Texas vegetation complex prior to Euro-American settlement and, apparently in some instances, occurred as dense stands. There is no indication as to the growth form of mesquite trees prior to Euro-American settlement. Fire was a part of the environment when these explorers traveled through Texas (Wright and Bailey 1980), but the specific role it played in shaping the scenes they observed is difficult to know. However, biological agents and fire are credited with having limited mesquite densities on rangelands before Euro-American settlement in the Southwest (Jacoby and Ansley 1991).

Post-1900 Succession—Honey mesquite density increased in the Southwest during the 20th century. It is likely that most of the multistemmed thickets that occur in Texas today have greater stem and foliage density because of increased anthropogenic disturbance of the canopy (including use of fire to topkill shrubs, which induces sprouting) than would have occurred naturally. Individual shrub densities have increased since the late 19th century as well. This has also been attributed to human influence—either through suppression of natural fires, or dissemination of mesquite seed by the herding and migration of domestic livestock (Brown and Archer 1989).

Much of the vegetation in the Southwest is in a state of flux and may have been changing for centuries in many areas. This seems to be particularly true for the Texas savanna type. Its dynamics, however, may have been accelerated by the influence of recent human activities.

Intensive animal grazing coupled with extremes of climate may be instrumental in causing active fluctuation of vegetation composition and physiognomy. Domestic livestock have played a major role in dissemination of mesquite seed into mesquite-free areas (Archer 1995; Brown and Archer 1989). Observations of recently seeded Conservation Reserve Program (CRP) stands on cropland near mesquite stands indicates that in the absence of cattle grazing, mesquite seeds were probably deposited by wildlife (coyotes, hogs, birds). However, this appears to be restricted to the margins of already existing mesquite stands. Early settlers accelerated dissemination into mesquite-free areas first via the cattle drives that occurred about 1900, and second with continuous grazing within fenced areas.

Current landscape patterns may reflect a trend that has been ongoing for centuries, or phases in a pulse equilibrium that may exist in much of the Southwest. The current pattern may depend upon recent combinations of weather and human activity. Mesquite encroachment, or encroachment of other woody species would probably occur in the absence of domestic livestock grazing, but such grazing has probably accelerated this process.

Management Considerations—Historically, the Texas savanna has provided a home to an abundance of wildlife. But, in recent times, land clearing for agricultural purposes has reduced the habitat for some of these species (Garrison and others 1977). Livestock grazing has been a predominant factor in managing this vegetation type. The woody overstory plants of the savanna, especially mesquite, have been viewed as pests by most landowners. Mesquite's thorny branches, increasing density on rangeland, and perceived competition with forage grasses have made it the target of eradication efforts over recent years. Chemical and mechanical controls have been the primary agents used in this effort (Fisher 1977). More recently, fire has gained increased acceptance as a management tool (Wright and Bailey 1982).

Mesquite now has an emerging image as a resource that should be managed rather than eradicated (Ansley and others 1996a; Fulbright 1996; Jacoby and Ansley 1991). Unfortunately, decades of control attempts have destroyed many mature stands of mesquite that contained single to few-stemmed trees. These trees were desirable in that they occupied far less surface area than multistemmed growth forms that resulted from destruction of aerial tissue and subsequent resprouting. Complete elimination of mesquite has been a goal that few landowners have achieved, and the concept of complete removal is questionable, both economically and environmentally (Fisher 1977).

Mesquite has many potential benefits to the ecosystem when maintained at controlled densities such as in a savanna. Such benefits include nitrogen fixation, livestock shade, habitat for nesting birds, and the potential as firewood or wood products. Mesquite has the potential to produce commercial hardwood in some regions with higher rainfall (Felker and others 1990). In lower rainfall areas, shrubby growth forms of mesquite can have other benefits, such as wildlife habitat. A mesquite savanna offers a pleasant landscape and may improve the value of a property over either an unmanaged woodland or a treeless grassland.

Recent research suggests that mesquite savannas can be sustained as long as the herbaceous understory is maintained at sufficient densities to out compete mesquite seedlings (Archer 1989; Brown and Archer 1989; Bush and Van Auken 1990). A savanna of this nature can be created and maintained in large part by using prescribed fire—one of the more environmentally acceptable and most economically sustainable options for managing woody plants (Ansley and others 1996a). In the initial stages of stand treatment, herbicides may be a useful supplement to the use of prescribed fire. However development and maintenance of the desired savanna growth form can often rely on the use of low-intensity fire, which can be achieved by burning under certain fuel loadings, humidities, and air temperatures. Creating a savanna from thickets using low-intensity fires will take time and should be part of a long-term management plan.

Response of honey mesquite to fire is highly variable and is a function of fine understory fuel loading and condition and of season of the year (Ansley and others 1995; Lotan and others 1981; Wright and others 1976). Abundant fine fuels tend to produce hotter fires and result in more topkill of the woody plants than lighter loadings (Wright and Bailey 1982). Summer fires will produce more topkill than winter fires (Ansley and others 1998). Fine herbaceous fuel loading and season of the year can work in various combinations to produce partially defoliated mesquite, or completely topkilled mesquite that quite often produces abundant sprouts from the root crown. Mesquite age also affects survival of individual plants after fire. Individual trees 1.5 years of age or less are easily killed by a fire when the soil surface temperatures are above 500 °F (260 °C) (Wright and others 1976). At 2.5 years of age, they can be severely harmed, and if older than 3.5 years, they are seemingly fire resistant at these soil temperatures.

Stand-Replacement Fire Regimes

Major Vegetation Types

The major vegetation types within this fire regime type are varied. Broadly, they include grassland and shrubland vegetation types (fig. 1-2).

Grasslands

The grassland types (fig. 6-9) include:

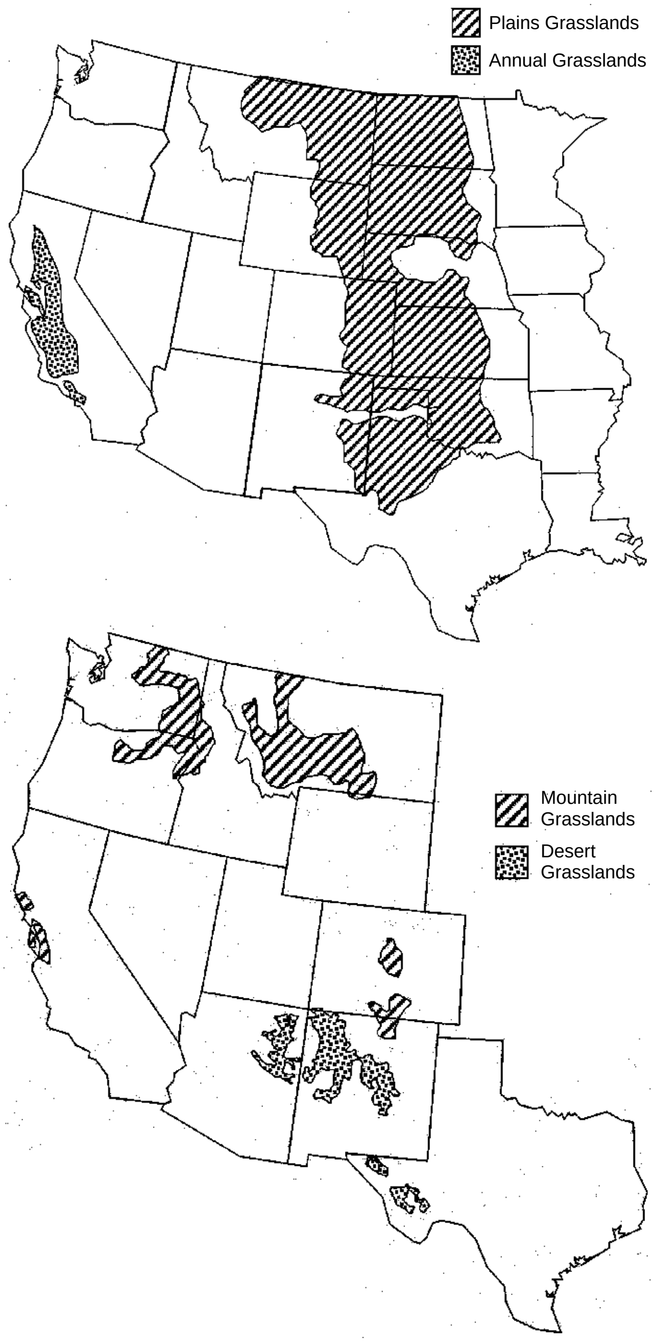

Figure 6-9—Distribution of plains, mountain, desert, and annual grassland FRES ecosystems.

- The **plains grasslands**, which range from Canada south to northern Texas in a broad swath that covers much of the Mid-Western United States.
- The **mountain grasslands**, which consist of open, untimbered mountainous areas from Canada south through the Northern and Central Rocky Mountains and the Coastal Range.
- The **desert grasslands**, which occur in the Southwestern States and in the Great Basin.
- The **annual grasslands**, which are concentrated for the most part in the valleys and foothills of California and along the Pacific coast.
- **Cheatgrass** (fig. 6-10), which has invaded and gained dominance in many plant communities in the Intermountain and Columbia Basin regions (Monsen 1994).

Figure 6-10—Cheatgrass.

Shrublands

Shrublands are described here as desert shrubland types and the chaparral-mountain shrub type. Desert shrublands transcend North America's four major deserts—Mojave, Sonoran, Chihuahuan, and Great Basin (fig. 6-11, table 6-3). These deserts encompass about 500,000 square miles (1,717,000 km^2) within the physiographic Basin and Range Province, surrounded by the Rocky Mountains and Sierra Nevada in the United States, and the Sierra Madre Occidental and Sierra Madre Oriental in Mexico (MacMahon 1988; MacMahon and Wagner 1985). They are characterized by low but highly variable rainfall, 10 inches/year (25 cm/year), and high evapotranspiration. Each desert differs in precipitation patterns, temperature variables, and vegetation structure (Burk 1977; Crosswhite and Crosswhite 1984; MacMahon 1988; MacMahon and Wagner 1985; Turner and Brown 1982; Turner and others 1995).

Bailey's (1978) Desert Division includes Mojave, Sonoran, and Chihuanhuan Deserts, considered warm deserts because their precipitation is mostly rain. The Mojave receives winter rainfall, the Chihuahuan summer rainfall, and the Sonoran both. Winter rainfall tends to be of long duration, low intensity, and covers large areas, whereas summer rainfall is of short duration, high intensity, and covers limited areas (MacMahon 1988). The Mojave Desert has greater elevation and temperature variations than the Sonoran

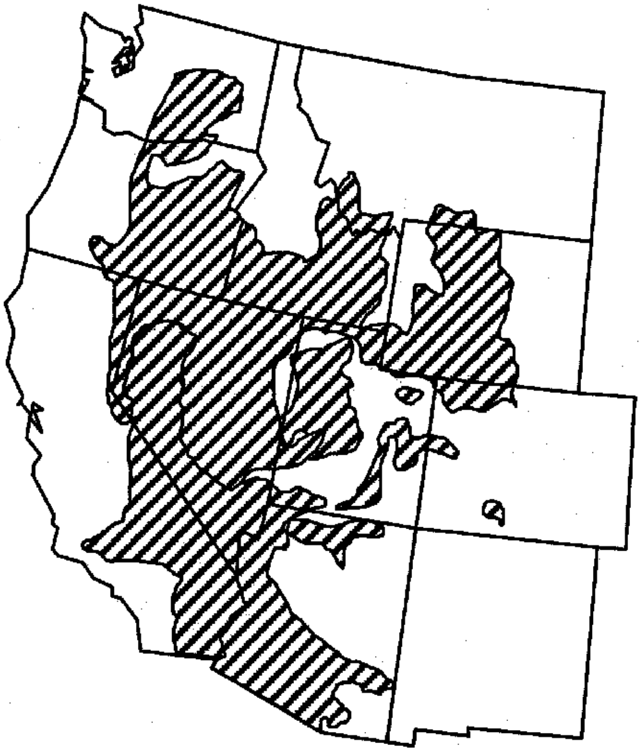

Figure 6-11—Distribution of desert shrub FRES ecosystems.

Table 6-3—Physiognomic fuel types for desert shrublands[a] associated with the four North American deserts.

Desert shrublands	North American deserts			
	Chihuahuan	Sonoran	Mojave	Great Basin
Sagebrush F-29				
Great Basin sagebrush K-38			X	X
Desert shrub F-30				
Blackbrush K-39			X	X
Saltbush/greasewood K-40	X	X	X	X
Creosotebush K-41	X	X	X	
Creosotebush/bursage K-42		X	X	
Mesquite bosques K-27	X	X	X	
Paloverde/cactus shrub K-43	X	X		
Southwestern shrubsteppe F-33				
Grama/tobosa shrubsteppe K-58	X	X		
Trans-Pecos shrub savanna K-59	X	X		

[a] FRES (F) shrubland ecosystems and the Kuchler Potential Vegetation System (K) equivalents (Garrison and others 1977).

Desert, which is lower, flatter, and warmer. Although the Chihuahuan Desert lies south of the Sonoran, it varies more in elevation and has colder winters. The Mojave Desert is considered transitional between the Sonoran and Great Basin Deserts, respectively, sharing components of each at its extreme southern and northern ends. The Great Basin desert is considered a cold desert because its precipitation is primarily snow (MacMahon 1988).

Vegetation in these regions varies from predominantly shortgrass prairie, consisting of sparsely distributed bunch grasses, to predominantly shrubs, sometimes with scattered small trees, and often with exposed areas of soil (fig. 6-12). Desert and desert shrubland vegetation has been classified in numerous ways (Shreve and Wiggins 1964; Turner 1982; Turner and Brown 1982; Vasek and Barbour 1977). We focused on desert shrublands within the United States (table 6-1).

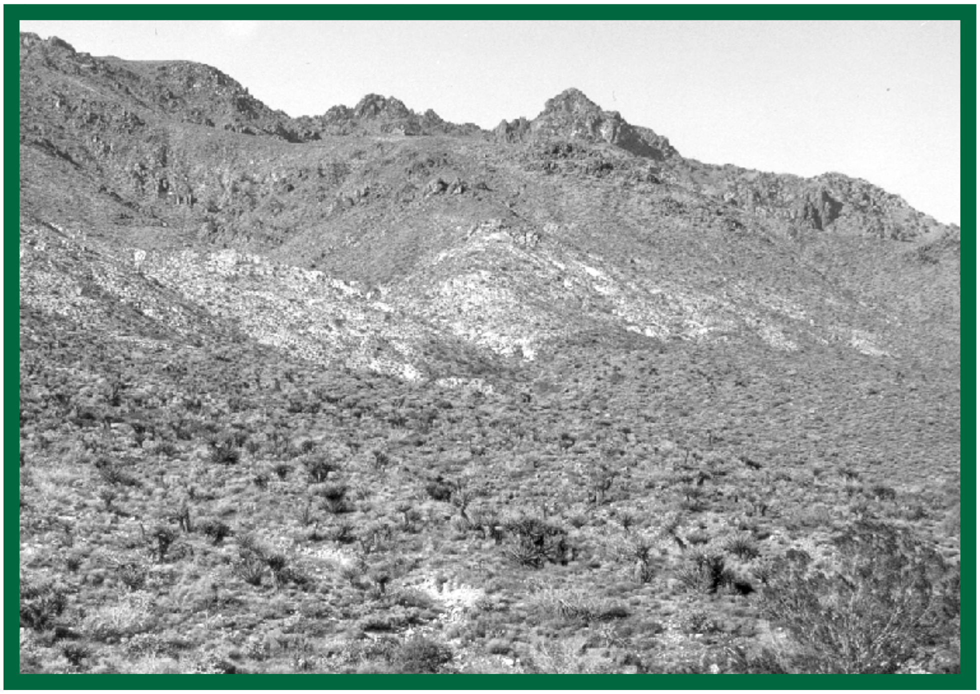

Figure 6-12—Bare soil, evident between shrubs and small trees, is a common characteristic of North American deserts as seen in the Mojave Desert, California.

Although these shrublands are distributed as a continuum of natural ecosystems, the use of vegetation classification systems gives us a convenient functional format for making fire management decisions. For our purposes, desert vegetation will be subdivided according to the FRES ecosystems as organized in table 6-3. We included the FRES sagebrush and Southwestern shrubsteppe types in our description of desert shrublands based on their similar fuels types, geographical proximity, and species integration.

Great Basin Sagebrush—This type characterized by sagebrush species (fig. 6-13) covers plateaus and vast plains at elevations ranging between 1,600 and 11,000 feet (490 and 3,500 m) with varied soils derived from lava flows, ancient lake beds, and alluvium (Garrison and others 1977). The Great Basin sagebrush, the largest range ecosystem in the Western United States, covers about 247 million acres (100 million ha) of arid lands (Blaisdell and others 1982). Sagebrush and associates are valuable for soil stabilization, wildlife habitat, animal feed, and ecosystem stability. There are about 22 species and subspecies; some have been studied extensively (Harniss and others 1981; Koehler 1975; Monsen and Kitchen 1994; Roundy and others 1995; Tisdale and Hironaka 1981). Sagebrush, composed of dwarf and tall sagebrush species, range between 1 and 7 feet (0.3 and 2 m) tall and grow in dense clumps or scattered plants. Shadscale, spiny hopsage, Mormon tea, and milkvetch are important co-dominants in this vegetation type. Understory grasses such as wheatgrass, brome, fescue, and bluegrass, and variable forbs form discontinuous patches with bare soil.

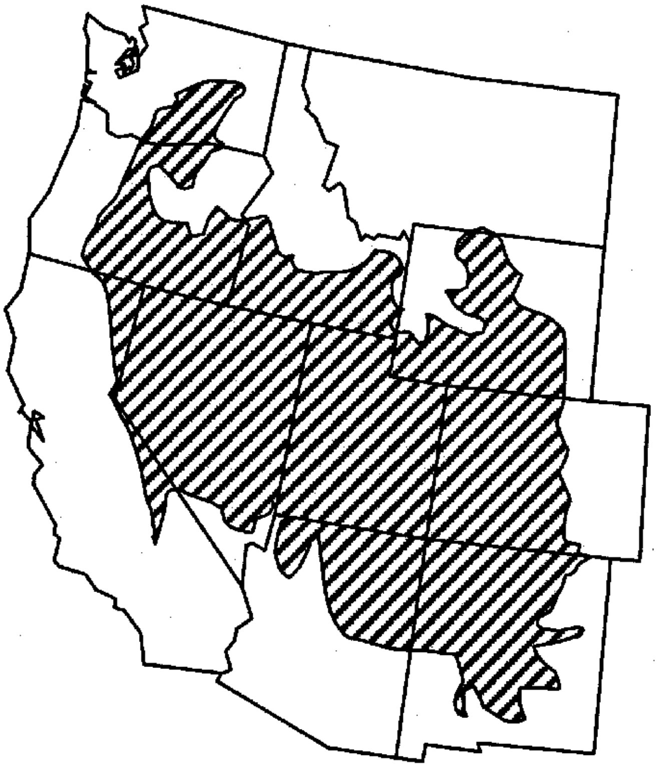

Figure 6-13—Distributiuon of Great Basin sagebrush FRES ecosystems.

Blackbrush—This type is composed of dense to scattered low stature shrubs and dense to open grass at elevations below 6,550 feet (2,000 m) (fig. 6-14). Blackbrush is one of the least studied landscape dominant shrubs in the United States. It prefers level topography and is not common on slopes or in drainages (Lei and Walker 1995). It maintains the highest cover of any desert shrub community. This transitional community between the Great Basin and the Mojave Desert occurs where annual precipitation is about 7 inches (18 cm) (MacMahon 1992). Moisture may limit its range. Blackbrush usually occurs in almost pure stands, although it intergrades with creosotebush and bursage at lower ecotones and sagebrush/juniper ecotones at higher elevations (Lei and Walker 1995).

Saltbush-Greasewood—This shrubland is characterized by halophytes and succulent subshrubs. Vegetation dominants include shadscale, black greasewood, and saltbush with saltgrass, winterfat, and sagebrush also present. This shrubland is common to all four deserts (table 6-3) and occurs on approximately 42 million acres (17 million ha) on heavy depauperate soil, often with underlying hardpan or alkaline flats. It is found below the sagebrush zone, generally below elevations of 6,900 feet (2,100 m). Saltbush and black greasewood are dominant and co-dominant species throughout much of their range from Canada to northern Mexico, eastern California to Colorado and northeast Montana.

Creosotebush—This vegetation consists of low to medium-tall, typically open shrubs (fig. 6-15) that grow on bajadas, valley floors, gentle slopes, sand dunes, and in arroyos below 5,000 feet (1,500 m) in the Mojave, Sonoran, and Chihuahuan Deserts. Creosotebush is a widespread dominant or co-dominant that also forms transitional vegetation between the three warm deserts. Creosotebush occurs in mixed to pure stands of open, low but variable diversity plant communities on about 46 million acres (18.4 million ha) (Cable 1973).

Joshua Tree—In parts of the Mojave Desert, creosotebush is associated with the Joshua tree woodland (fig. 6-16). Joshua trees can resprout after fire, develop fire-resistant bark on trunks, have protected apical meristems usually high above surrounding fuels, and reseed from offsite sources. Resource managers at the Joshua Tree National Monument in California are testing prescribed burning as a tool to create fuel

Figure 6-14—Prescribed burning to reduce blackbrush fuels at the urban wildland interface, Carson City, Nevada.

Figure 6-15—Creosotebush shown growing on the Mojave Desert valley floor may resprout (inset) after fire.

Figure 6-16—Joshua tree clones provide clusters of fuel in otherwise sparse desert shrublands, Mojave Desert, California.

breaks to reduce large-scale destruction of this unique resource by wildfires (fig. 6-17).

Creosotebush-Bursage—This is a transitional plant association found below 5,250 feet (1,610 m) elevation. It merges with the paloverde-cactus shrub association found in the Sonoran Desert. In this region creosotebush-bursage has higher species diversity including a larger tree component (table 6-4).

Paloverde-Cactus Shrub—This type is characterized by open-to-dense stands of low-to-medium tall shrubs, small trees, cacti, and succulents (fig. 6-18). Paloverde, pricklypear, cholla, saguaro, and bursage are dominant species in this vegetation type. These communities are a diverse mosaic of mixed vegetation that occur in the Sonoran Desert at elevations generally below 4,000 feet (1,200 m) (table 6-4).

Southwestern Shrubsteppe—This shrub type or the semidesert grass-shrub type (called desert grasslands in the FRES system) is composed of gently sloping desert plains found below the Rocky Mountains and between the low mountain ranges of the Sonoran Desert, Mexican Highland, and Sacramento section in Arizona, New Mexico, and Texas (fig. 6-19). Annual precipitation in this ecosystem varies from 10 inches (25 cm) in western areas to 18 inches (46 cm) to the east. Despite the fact that half of the rainfall occurs during warm months (frost free periods occur 180 days or more of the year), evapotranspiration is between 80 and 90 inches (203 to 229 cm) per year and may exceed the precipitation by a factor of 10.

Vegetation is composed of short grasses and shrubs of variable composition. Grasses inhabit the more developed Aridisol and Mollisol soils. Shrubs inhabit the shallow soils. Junipers occur exclusively on Entisols, which are predominantly found in the South. Yucca, mesquite, creosotebush, and tarbush are the dominant woody plants, while black grama, tobosa, and threeawn are the dominant herbaceous plants. Curlymesquite and other grama species also contribute significantly to the biomass of these shrubsteppe communities, which are used mainly as rangeland.

Two shrubsteppe types are recognized. The **Grama-tobosa shrubsteppe** occupies areas at elevations between 1,610 and 7,045 feet (488 to 2,135 m) and includes the more shrub dominated communities of the shrubsteppe. Black grama, sideoats, and tobosa are climax indicators occupying arid grassland communities throughout the Southwest. Black grama prefers more gravelly upland sites; sideoats is less selective, while tobosa prefers heavier clay lowland soils. The **Trans-Pecos shrub savanna** is found on

Figure 6-17—Prescribed burning in a Joshua tree forest to reduce fuel loading at the urban/wildland interface, Covington Flats, Joshua Tree National Park, California.

the Stockton Plateau and southwestern portion of Edwards Plateau. It has a higher average elevation (4,000 to 6,000 feet; 1,220 to 1829 m) and greater rainfall than the grama-tobosa shrubsteppe. This is a shrub dominated type characterized by grasses and the common occurrence of junipers (fig. 6-20). Junipers occupy more than 6 million acres (2.4 million ha) of rangeland in dense to open communities with oaks, Texas persimmon, and mesquite.

Chaparral-Mountain Shrub—This ecosystem type (fig. 6-19, 6-21) occupies lower and middle elevation mountain areas in the Pacific States, the Southwestern States, and the Rocky Mountains. The vegetation consists of dense to open shrubs or low trees with deciduous, semideciduous, and evergreen species represented. Some of the types are so dense that understory vegetation is practically eliminated, while other types support a highly productive understory.

Fire Regime Characteristics

Fire frequency was variable in the stand-replacement fire regime types and depended upon ignition sources and plant community development. In the grassland types, fires could occur in any given year, provided the grass was cured and dry enough to burn. Although fire frequencies could not be measured precisely, mean fire intervals probably ranged from about 4 to 20 years depending on climate and ignition sources (Gruell and others 1985a). In the plains and grasslands, Native Americans ignited fires for a wide variety of cultural reasons. This was the predominant source of ignition in heavy use areas particularly at lower and middle elevations. But, an ever-present ignition source was lightning, which was probably more important in valleys surrounded by forests than in plains grasslands due to differences in efficiency of lightning (Gruell and others 1985b). Grasslands, occupying flat to gently rolling terrain, would burn over large areas until a break in terrain or a change in weather stopped the fires. Fires swept over extensive areas sometimes covering several hundred square miles.

Desert shrublands have been influenced over the last 12,000 years by climatic shifts, varying soils, and fire. Prior to Euro-American settlement, fires in these desert shrublands were set by lightning and Native Americans (Humphrey 1974; Komerek 1969). Wyoming big sagebrush experienced fire intervals ranging from 10 to 70 years (Vincent 1992; Young and Evans 1991). Arid land fire history studies report fire intervals between 5 and 100 years (Wright 1986). Griffiths (1910) and Leopold (1924) reported that before 1880

Table 6-4—Physiognomic and taxonomic descriptions of vegetation types modified from Kuchler (1964) showing habitat type[a] fuel, and forage associated with each. Note: Although numerous grass species are not listed for each vegetation type, they have become cosmopolitan throughout each type as a result of anthropogenic disturbance. Their impact on the fire dynamics of these desert ecosystems should be considered in making fire management decisions.

Vegetation[a] **-Fuels (Fu)** **-Forage (Fo)**	**Dominant species** **•Associated genera**	**Tree[b]**	**Shrub**	**Herb**	**Cactus**
Great Basin sagebrush[c]	*Artemisia tridentata*		S		
Dense to open low to medium	• *Artemisia, Atriplex, Chrysothamnus, Coleogyne*		S		
shrubs	• *Ephedra, Eriogonum, Tetradymia*		s		
Fu-0 to 2,000 lb/acre	• *Astragalus, Lupinus, Phacelia*			H	
Fo-0 to 700 lb/acre	• *Agropyron*			G	
Blackbrush		S			
Dense to open broadleaf	• *Artemisia, Gutierrezia, Haplopappus*		S		
evergreen shrubs	• *Ephedra*		s		
± herbaceous understory	• *Hilaria*			G	
Fo-250-500 lb/acre					
Saltbush/black greasewood	*Atriplex confertifolia/Sarcobatus vermiculatus*		S/s		
Open small shrubs	• *Lycium, Artemisia, Atriplex, Grayia, Krascheninnikovia*[d]		S		
Fu-250 to 750 lb/acre	• *Allenrolfea, Menodora, Suaeda*		ss		
Fo-50 to 200 lb/acre	• *Kochia*			H	
	• *Distichlis*			G	
Creosotebush	*Larrea divaricata*		S		
Open dwarf to medium shrubs	• *Yucca brevifolia*[e]	T			
Fu-40 to 100 lb/acre	• *Lycium, Baccharis*		S		
Fo-12 to 40 lb/acre	• *Encelia, Franseria, Sphaeralcea*		s		
Creosotebush/Bursage	*Larrea divaricata/Ambrosia dumosa*		S/s		
Open dwarf to medium shrubs	• *Cercidium, Dalea, Prosopis, Olneya*	T			
Fu-40 to 100 lb/acre	• *Lycium, Acacia, Fouquieria*		S		
Fo-12 to 40 lb/acre	• *Encelia, Franseria*		s		
	• *Hilaria*			G	
	• *Opuntia, Ferocactus*				c
Mesquite Bosques	*Prosopis glandulosa; P. velutina*	T			
Open to dense forest low	• *Cercidium, Olneya, Prosopis, Populus, Dalea, Salix*	T			
broadleaf deciduous trees	• *Acacia, Baccharis, Lycium*		S		
Fu-250 to 1000 lb/acre					
Fo-0 to 500 lb/acre					
Paloverde/Cactus Shrub	*Cercidium microphyllum/Opuntia* spp.	T			c
Open to dense low trees,	• *Cercidium, Olneya, Prosopis*	T			
shrubs, and succulents	• *Jatropha, Larrea, Lycium, Simmondsia,*		S		
Fu-100 to 250 lb/acre	*Acacia, Condalia, Fouquiera, Celtis*				
Fo-30 to 100 lb/acre	• *Calliandra, Ephedra, Franseria, Janusia*		s		
	• *Carnegiea*				c
	• *Ferocactus, Echinocereus, Opuntia*				c
Grama-tobosa shrubsteppe	*Hilaria* spp., *Bouteloua* spp.			G	
short grass with shrubs	*Larrea*		S		
Fo-0-600 lb/acre	*Yucca* spp.		ss		
- - - - - - -					
Trans-pecos shrub savanna	*Juniperus* spp.	T			
shrubs with short grass	*Hilaria* spp., *Bouteloua* spp., *Muhlenbergia* spp.			G	
Fo-0-600 lb/acre					
- - - - - - - -					

[a] Based on Kuchler's classification system
[b] T = tree; S = shrub; s = subshrub; ss = succulent shrub; H = herbaceous; G = grass; c = cactus
[c] Great Basin sage is broken into four productivity classes (Garrison and others 1977)
[d] *Eurotia lanata (Pursh) Moq. = Krascheninnikovia lanata (Pursh) A. D. J. Meeuse & Smit,* (Jepson 1993)
[e] *Yucca brevifolia* (Joshua trees) become a significant tree component in parts of the Mojave Desert and grama-tobosa shrubsteppe

Figure 6-18—Mixed vegetation of the paloverde/cactus shrub in the Sonoran desert near Four Peaks, Maricopa County, Arizona.

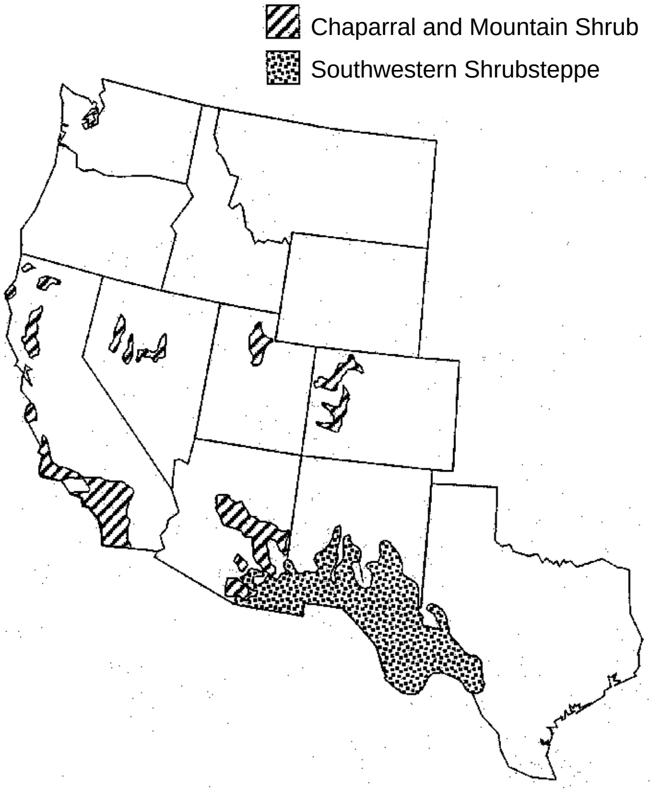

Figure 6-19—Distribution of Southwestern shrubsteppe and chaparral-mountain shrub FRES ecosystems.

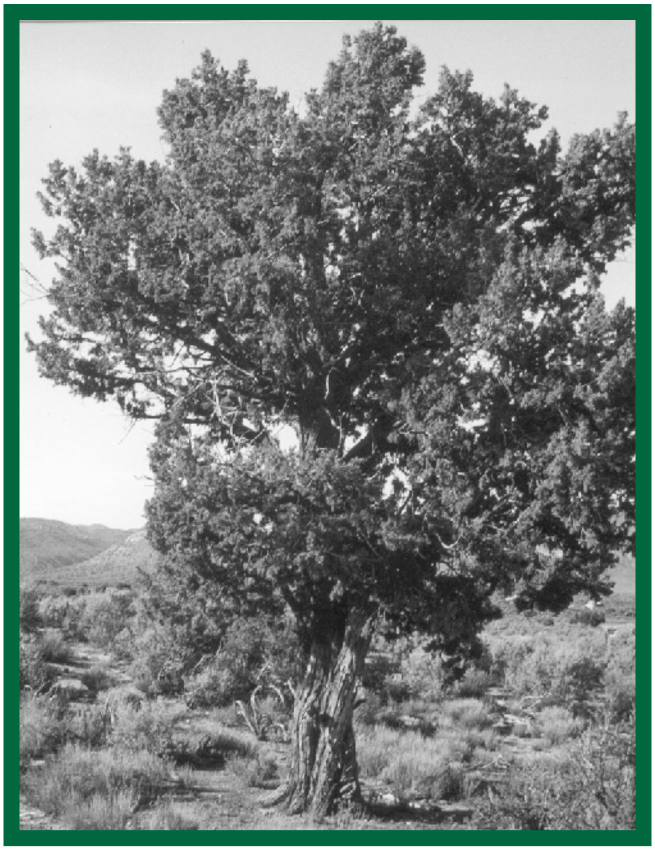

Figure 6-20—Mixed fuels found in juniper shrub savanna (New York Mountains, California).

Figure 6-21—Typical chaparral vegetation (Arctostaphylos, Ceanothus), Mill Creek, San Benardino National Forest, California.

desert grasslands produced more grass and fires recurred at approximately 10-year intervals. Before settlement deserts were characterized by sparse vegetation, broken by barren soil, and were not expected to burn except under unusual circumstances. But when fire occurs in warm desert shrub habitat, a long recovery is expected. This recovery depends on geographical location, species composition, and climatology after the burn. Recovery is more rapid in areas receiving higher precipitation. The various desert shrublands vary in wildfire risk ranging from nonexistent risk of the sparsely vegetated saltflats to high risk associated with heavy fuel loadings often found in the mesquite type. Postfire survival by desert plants may depend on genetic variation (Munda and Smith 1995), resprouting capability, resistant seeds, and delayed mortality.

In California chaparral, fire intervals for large fires (more than 5,000 acres) typically ranged from 20 to 40 years (Wright and Bailey 1982). But at higher elevations and north aspects fire return intervals were longer, perhaps as infrequent as 50 to 100 years. Young stands of chaparral whose canopy has not closed and stands that have not restocked well after disturbance often have a grass component that can burn on any given year, as is the case with the grasslands. These fires may or may not be stand-replacement fires, depending upon the amount of heat transferred from the grass component to the sparse shrub overstory. Fully developed chaparral stands can be difficult to ignite unless there is some component of dead material and good fuel continuity. However, given an ignition and some wind, they will propagate a moving fire even when virtually no dead material exists in them. Because these are crown fires, they are almost always stand-replacement fires. With both the grasslands and chaparral, all or most of the aboveground portion of the plants are killed. Most of the perennial grasses have a perennating bud at or near ground level, often protected by bunched stems that act as insulators; often, tufts of these stems remain after fire. Chaparral shrubs are often killed down to the root collar; sometimes the entire individual is killed outright.

Fuels

Grassland Fuels—When cured and dry, grassland fuels are ideally suited for burning. For the most part, they fall into the fine fuel category; however, the compact arrangement of stems in the "tufts" of bunchgrasses makes these portions of the plant difficult to ignite regardless of their dryness. Once ignited, however, they can smolder for long periods if enough old stem material has accumulated.

Plant density is also a critical factor in a grassland's ability to propagate fire. Heat output is relatively low from grass fuels, so fairly continuous fuels are necessary for fire spread to occur. Light winds can sometimes compensate for moderately sparse fuels by providing required flame bathing. The amount of fuel can vary with site condition, precipitation, and disturbance history. Typical annual productivity in desert grasslands can vary from next to nothing upwards to 1,000 lb/acre (1,120 kg/ha); in plains and mountain grasslands, productivity can be as high as 2,000 lb/acre (2,240 kg/ha) (table 6-5).

The character of a grassland fire is also affected by the overall geometry of the stand, which changes throughout the life cycle of the plants in the stand. The most dramatic example of this can be seen in annual grasslands where the plants germinate, seed, and die in a single season. A stand of recently cured annual grass can be quite dense and tall (up to 6 or 7 feet); its bulk density can be optimum for propagating a fast moving fire. In a relatively short period, a process of stand collapse begins and the bulk density of the stand becomes steadily modified. By the end of the season, the biomass is in a dense thatch on the ground and will begin decomposing—in some localities, fairly completely. Fire can still propagate during these later stages, as long as not too much moisture has accumulated in the thatch, but spread rates will not be as great.

Cheatgrass is a highly flammable fuel because of its finely divided plant structure, long period in a cured condition, rapid response to drying, and a tendency to accumulate litter (Bradley 1986a). Cheatgrass dries 4 to 6 weeks earlier than perennials and can be susceptible to fire 1 to 2 months longer in the fall. It produces large quantities of seed that usually develop into dense stands providing ideal fuel continuity for fast spreading fires. It grows well in areas of low precipitation that frequently undergo severe fire seasons.

Desert Shrublands—Fuels include cacti and other succulents, grasses, shrubs, small trees, and mixtures of these. Fuels occur in discontinuous patches to areas where trees, shrubs, and grasses are contiguous. Fuel loadings may reach 2,000 lb/acre (2,240 kg/ha) (fig. 6-22, 6-23). See table 6-4 for fuel loading and forage production for each associated shrub community.

Table 6-5—Fuel loadings (lb/acre) from FOFEM fuel models (Reinhardt and others 1997) for FRES grassland ecosystem types based on annual productivities.

Fuel class	Desert	Plains	Mountain
Sparse	300	600	900
Typical	600	1,250	1,900
Abundant	900	1,900	2,800

Fuel loading in **sagebrush** varies depending on the site and species. Based on shrub height and percent cover, big sagebrush varies from 0.26 to 4.6 tons/acre (0.55 to 10.2 t/ha). For a stand 2.5 feet in height and 20 percent cover, conditions typically found, sagebrush foliage and stemwood averages 1.5 tons/acre (Brown 1982). Herbage production for this vegetation type can vary from about 200 lb/acre (224 kg/ha) under poor growing conditions (Brown 1982) to 1 ton/acre (2.2 t/ha) under favorable conditions (Garrison and others 1977). Forage production generally is one-fifth of the annual herbage production. Humphrey (1974) noted that sagebrush was more subject to burning than any other desert type.

Dwarf sagebrush (14 habitat types) is usually relegated to shallow soils and is not considered a fire management problem because fuel continuity is poor and it generally cannot carry fire. Tall sagebrush (29 habitat types) occurs on deeper soils, often has a substantial grass component, and burns readily (Blaisdell and others 1982). The presence of a herbaceous understory increases the potential for big sagebrush to carry a fire. Threetip, basin, Wyoming, and mountain big sagebrush occupy about 60 percent of the total sagebrush area. This sagebrush association is practical to burn (Blaisdell and others 1982). Techniques for managing sagebrush/grass ecosystems with fire and other means are discussed by Blaisdell and others (1982), Bushey and Kilgore (1984), McGee (1976, 1977), and Onsager (1987) (fig. 6-24). Fuel and fire behavior models were developed by Brown (1982), Frandsen (1981), Reinhardt and others (1997), and Tausch (1989) for burning in Great Basin sagebrush. Fire behavior studies in big sagebrush show that fire intensity and rate-of-spread can be two to three times greater when sagebrush foliage is cured, yet the proportion dead has little effect on predicted fire behavior (Brown 1982).

In **blackbrush** fuel production ranges from 0 to 500 lb/acre (0 to 560 kg/ha), and forage production ranges from 0 to 150 lb/acre (0 to 168 kg/ha). Blackbrush is negatively associated with fine fuels of litter and grasses. In **saltbush-greasewood** fuels production varies from year to year, depending on the amount of

Figure 6-22—During wet years, a herbaceous layer develops in the bare spaces between the dense thorn-shrub of the Sonoran desert, Maricopa County, Arizona, increasing the potential for major fires.

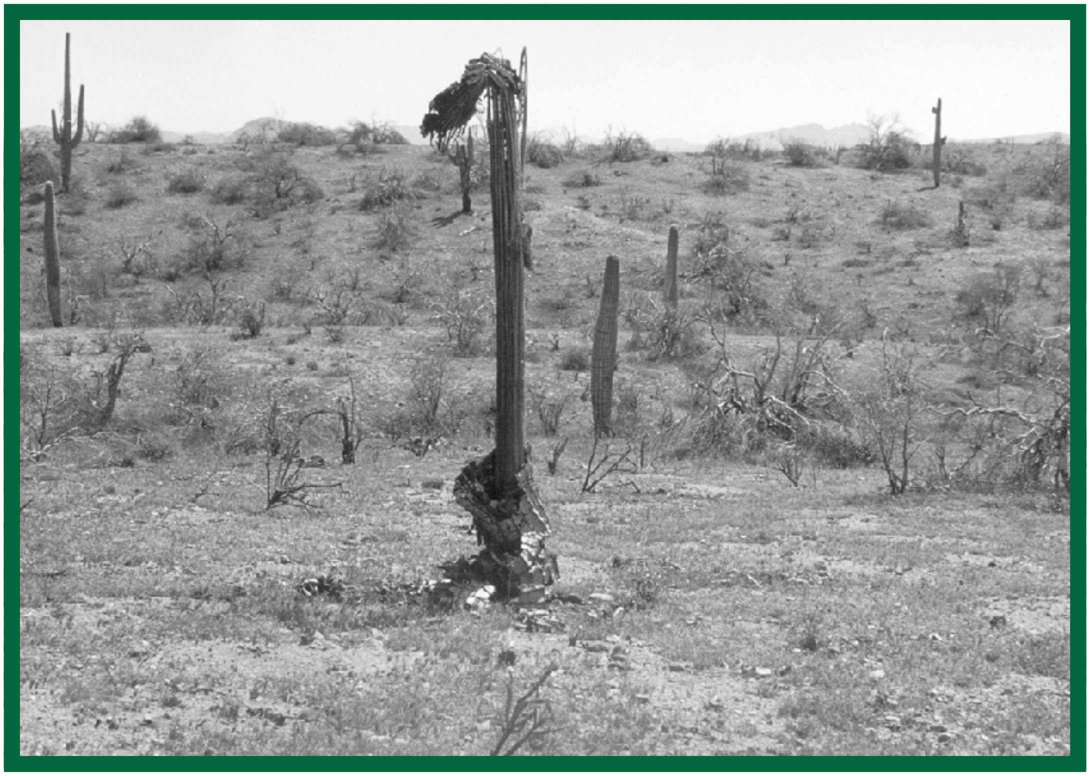

Figure 6-23—A wildfire burned 10,000 acres of this Sonoran desert thorn-shrub in Four Peaks, Tonto National Forest, Arizona.

Figure 6-24—Fire is used as a range management tool for sagebrush found on the Great Basin plains.

precipitation. Production is also related to soil salinity and texture (West 1994). Herbage production is generally 0 to 500 lb/acre (0 to 560 kg/ha).

Creosotebush has low leaf to stem biomass, yet its standing dry biomass may reach about 3.8 ton/acre (8.5 t/ha) and produce about 892 lb/acre (1,000 kg/ha) per annum of new fuels (Chew and Chew 1965). The resinous foliage is flammable, but fire generally will not carry well in this community because the plants are usually surrounded by bare soil. Herbage production ranges from 40 to 100 lb/acre (44 to 112 kg/ha), about one-third of which is considered forage. High species diversity within the **creosotebush/bursage** shrub type produces diverse fuels. In some areas dense stands with herbaceous understory supply contiguous fuels for fire.

Mesquite bosques (fig. 6-25), characterized by low deciduous mesquite trees, are typically found in high moisture areas, and may produce up to 2,000 lb/acre (2,240 kg/ha) of herbage, particularly in areas that flood periodically and where the mesquite has been artificially reduced. Fuels are highly concentrated in mesquite bosques. Herbage production is commonly between 750 and 1,000 lb/acre (840 and 1,120 kg/ha) with forage production from 0 to 500 lb/acre (0 to 560 kg/ha) (Garrison and others 1977). Higher fuel loading on a site will increase the fire mortality of mesquite. Areas with 2.25 ton/acre (5.06 t/ha) of fine fuel sustain up to 25 percent mortality, but only 8 percent mortality for 1.1 ton/acre (2.47 t/ha) (Wright 1980). Dunes may form in association with mesquite thickets.

In **paloverde-cactus shrub** fuels production ranges from 100 to 250 lb/acre (112 to 280 kg/ha); about 35 percent of this vegetation has forage value. Fuels in the **Southwestern shrubsteppe** are mixed grass-shrublands. The Grama-tobosa region has a higher grass component while the Trans-Pecos shrub savanna has a higher shrub component. The variable fuels in the Trans-Pecos shrub savanna produce up to 450 lb/acre (505 kg/ha) of forage. Creosotebush and yucca are present, but grama and tobosa primarily contribute to the maximum 1,500 lb/acre (1,680 kg/ha) herbage production in this type.

Chaparral—Generally, fuels are not as easily ignited as grass fuels, but once ignited will burn readily if conditions are right. Plant density can vary with site, and sometimes with species. This is but one factor that affects fuel continuity in a stand. Another factor is the basic within-plant geometry that varies by species. Geometry and arrangement of the woody fuel portion and the leaves of chaparral plants are key to understanding the ability of chaparral stands to propagate fire. The woody fuel inside a given shrub varies in size class ranging from fine fuel (<0.12 inch diameter)

Figure 6-25—Mesquite thickets form highly concentrated fuels in desert washes, Mojave Desert, California.

to heavy fuel (3 or 4 inches and larger in the case of some manzanita species). The arrangement and distribution of these size classes within a shrub varies by species. Two extremes illustrate this: the arrangement of the woody portions of chamise and manzanita species (fig. 6-26). The smaller woody size classes are quite dominant in chamise and tend to be in proximity throughout the crown; the opposite is true for manzanita. The leaves of chamise are small and needlelike and are often relatively dense on a given twig.

Manzanita is a broadleaf shrub. The leaves of some species are relatively sparse—being held distant from each other by the woody structure of the shrub. Other species of manzanita have dense clusters of leaves—so dense that their thick sclerophyllous structures act like an insulator. Other chaparral species, some members of the *Ceanothus* genus for example, have only a moderate amount of fine woody material and have small broad leaves that are sparsely distributed throughout the shrub crown. In general, the geometry of chaparral shrubs is not well suited to the spread of fire. Chamise is an exception, especially in dense stands with overlapping crowns. The maintenance of crown fires in chaparral almost always requires dry, windy conditions, which commonly occur in this vegetation type.

With few exceptions, fully developed stands of chaparral have no understory layer of vegetation, and therefore no potential for the "ladder effect" to propagate fire. However, when a litter layer exists, which occurs under gentle slope conditions, it can significantly aid fire spread under marginal burning conditions. In this situation, fuel moisture content becomes an important factor.

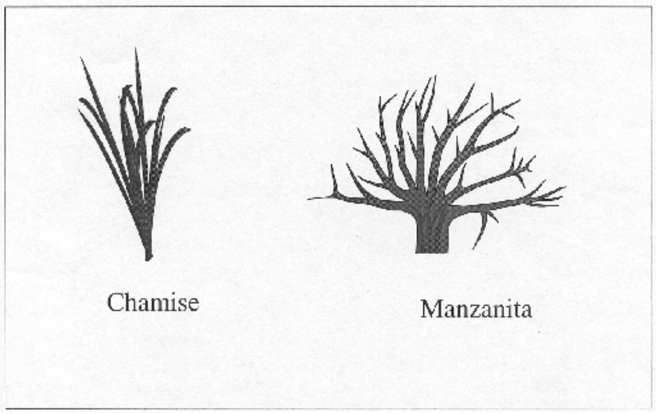

Figure 6-26—Arrangement, distribution, and size of woody fuels can vary by species.

The dynamics of dead fuel production in chaparral remain a mystery. Some suggest that dead fuel production increases with stand age (Rothermel and Philpot 1973). While this is undoubtedly true, age is not the only factor involved (Paysen and Cohen 1990). Complexities of onsite growing conditions and periodic events seem to be important. For example, the authors suspect that an unusual drought can produce fine dead material in chaparral stands that may be only present onsite for a year or so—making the assessment of dead fuel dynamics unclear. Considerable down and dead material can be found in old chaparral stands. The concept of "old," unfortunately, has to remain a relative one for now. The age at which significant amounts of dead material are produced in a given stand of a given species composition cannot be predicted yet.

Postfire Plant Communities

Plains Grasslands

Pre-1900 Succession—The literature on plains grasslands communities is rife with contradictory interpretations of grassland dynamics. A few facts seem to be agreed upon. First of all, pollen records and rat middens indicate that most of the Central Plains was covered with boreal forest dominated by spruce, while much of the Northern Plains was glaciated during the Pleistocene. There are indications that the Southern Plains and the arid grasslands of the Southwest were also dominated by various conifer and broadleaf trees. The climate change that brought about the end of the glacial period ushered in the retreat of the boreal forest and its replacement by grasslands—a kind of vegetation able to cope with the drier climate and soil conditions that predominated.

Fire was not a predominant force in delimiting the extent of the plains grasslands. But given their existence and their flammability characteristics, the presence of fire had to have an impact on the character of the grasslands, their species composition, and the distribution of dominance. Modifications of climate and soil development led to invasion of some grassland areas by woody species. Under these circumstances, fire probably had a distinct role to play in the maintenance, or loss, of these grassland areas. Working in concert with grazing animals, fire could check the advance of more fire-sensitive, woody species, providing enough grass fuel was available. It could also encourage the advance of woody species that were adapted to disturbance and harsh climate conditions. Where invasion by woody species was not an issue, fires could maintain a highly productive mode in some grasslands, and in others cause shifts in grassland species composition; under conditions of drought, it could result in severe site damage.

Clearly, fire was a common element in presettlement times, and there is some conjecture that its frequency might have increased with the arrival of Euro-American settlers (Jackson 1965). For years, attempts to suppress fires in the plains were either nonexistent or not effective. As late as the 1890s, from the Dakotas to the Texas Panhandle, fires would run unchecked for days. During this period, fire, drought, and grazing played a role in maintaining, and at times debilitating, the grassland character. When fire, or any other phenomenon that reduced the vegetative cover, occurred during periods of serious drought, wind erosion often retarded the processes of succession.

Post-1900 Succession—The general set of natural forces affecting succession just prior to 1900 has not really changed in principle. Land use has alternately intensified, and disappeared, and returned again in some cases. Some of the plains grasslands have been converted to agricultural use—producing corn, wheat, barley, and various legumes; some have been put to intensive grazing use—successfully in some instances, and in others with disastrous results. In the Southern Plains, the conjunction of inappropriate farming practices and a devastating drought in the 1930s brought about a perceived ecological disaster and social phenomenon, called the "dust bowl," that shook the fabric of Southwestern culture. In retrospect, no surprises should have existed.

The semiarid climate of the plains grassland area, the ever-present potential for drought, yearly temperature extremes, and the potential for high winds exist today, as they have existed for centuries. They were operative in forming the plains grasslands and continue to drive the processes of succession. The factors relevant today are the firmly entrenched agricultural practices and the use of the grasslands as pasturage for grazing animals. Land use patterns such as these, once terminated, will drive the processes of succession in various directions—dominated by the presence of the existing natural factors. Deviations from successional patterns of past centuries are difficult to predict other than on a case-by-case basis.

Management Considerations—Management of plains grasslands should be undertaken with a view toward maintaining stability under local climate and soil conditions. In the Northern Plains, a temperature range of more than 130 °F between yearly maximum and minimum temperatures can occur (a range of 174 °F has been recorded in one place). The average growing season can range from 116 days in the northernmost portion to 160 days in the southern part (Rogler and Hurt 1948). Native grasses tend to be hardy and drought resistant—such species as blue grama, buffalograss, western wheatgrass, and needlegrass. If the native grasses are to be used as livestock forage, then overutilization should be guarded against. Native range utilization by livestock should be supplemented by locally produced forage and seed crops whenever needed to protect native species.

The Southern Plains are also characterized by temperature extremes and a highly variable climate. Precipitation is comparatively light and infrequent; a major proportion of it falls during the active growing season, from April through September (Savage and Costello 1948). Humidity is low, winds are high, and evaporation is rapid. Hot temperatures and high winds often reduce the effectiveness of precipitation that does occur. Overutilization of rangelands during drought always has to be guarded against.

Fire can be either a disaster or a useful element in the plains grasslands, depending on its timing and severity. A range fire that denudes a large area preceding a drought can set the stage for severe soil movement in many areas of the Great Plains—the high winds and frequently aolean soils indicating the process. When good recovery is favored by adequate precipitation, fire can improve productivity for a while. The effectiveness of fire, both good and bad, can be mitigated by current levels of productivity and by intensity of utilization. Recently grazed grassland, or a year of low productivity, can reduce the impact of fire by minimizing fuel consumption, fireline intensity, and general extent of burning.

The use of fire as a management tool can improve productivity if it is applied in a manner consistent with the grassland's productivity, given climate and soil character. Kucera (1981) contrasted the application of prescribed fire between the more moist, highly productive grasslands and those of lower moisture availability and less productivity (fig. 6-27). Timing of the application centers on the development of thatch. In the higher productivity grasslands, the buildup of thatch tends to suppress productivity after a few years. In the lower productivity grasslands, the development of thatch provides a means of storing moisture and thus increases productivity—at least over a period of a few years. Thus, relatively frequent application of prescribed fire in the high productivity grasslands can be beneficial by removing thatch that has accumulated beyond desirable levels.

Mountain Grasslands

Pre-1900 Succession—Although bunchgrass species vary in their individual susceptibility to fire damage, repeated fires at intervals of about 5 to 40 years (Gruell and others 1986) maintained the bunchgrass community. The abundance of individual species no doubt varied not only by site conditions but by the actual frequency and seasonal timing of fire. A successional process of major importance was the continual checking and reduction of woody plant encroachment. Mountain grasslands were intertwined with forests and shrublands ranging from rose and aspen in Alberta to conifers and sagebrush of Rocky Mountain foothills. Encroachment into grasslands by woody species was an ongoing process kept in check by repeated fires.

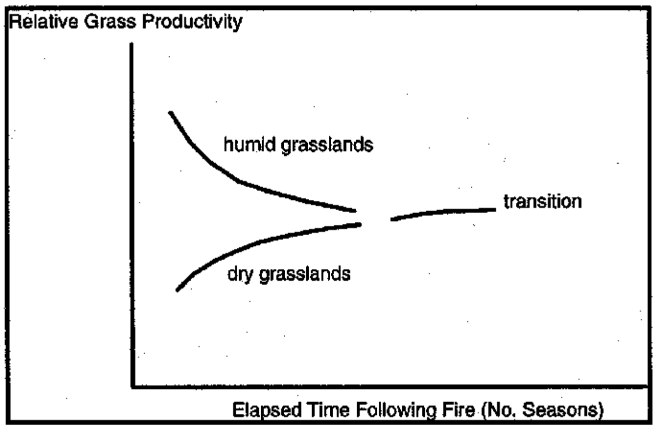

Figure 6-27—The productivity of humid grasslands versus dry grasslands after fire.

Post-1900 Succession—Grazing by livestock, elimination of Native American ignitions, and fire control efforts greatly reduced the amount of fire in these grasslands. As a result tree species such as ponderosa pine, Douglas-fir, and lodgepole pine, and sagebrush have increased substantially along ecotonal boundaries. In some areas dense Douglas-fir forests now dominate sites to such an extent that evidence of former grasslands is lost except by soil analysis (Bakeman and Nimlos 1985). Elimination of periodic burning has apparently reduced diversity of herbaceous species in some areas (Wright and Bailey 1982).

In a study of fire regimes in the Interior Columbia River Basin involving grasslands and other vegetation types, Morgan and others (1994) suggested that human influences have had a variable effect on the nature of fire regimes. Fires tended to be less frequent but not always more severe. For example, where exotic annuals have invaded sagebrush steppe vegetation, fires have become so frequent that sagebrush does not have time to reestablish, and the annuals return quickly. Changes in fire regimes can move in one direction as a result of active fire suppression that results in a buildup of fuel, or in another direction as a result of livestock grazing and other activities that break up fuel continuity. No single successional formula can be offered for grasslands in general.

Management Considerations—Prescribed fire can be effectively used to hold back woody plant encroachment and maintain high levels of productivity in mountain grasslands. The complexity of mountain grasslands, however, requires careful consideration of species composition and site dryness to design prescriptions for successful prescribed fire (Wright and Bailey 1982). For this, knowledge of species response can be helpful.

Idaho fescue is sensitive to fire partly because it is susceptible to smoldering in the clump that can kill plants or reduce basal area. It tends to recover slowly from fire; however, on some sites it can withstand burning (Bradley 1986b). Burning when soils are moist, such as in the spring, helps to minimize damage. Needlegrasses can also be severely damaged depending on severity of fire. Damage from wildfires can be minimized by the grazing of livestock to reduce fuels. Needle-and-thread grass reproduces by seed and can increase markedly in 2 to 4 years after fire (Gruell and others 1986). Bluebunch wheatgrass and Sandberg bluegrass recover quickly from fire (Bradley 1986c; Howard 1997). Rough fescue generally responds favorably to fire even after an initial reduction in basal area. Preburn coverages can be attained in 2 to 3 years (McMurray 1987).

Cheatgrass

Succession—Cheatgrass was accidentally introduced into the United States sometime around the turn of the 20th Century, supposedly through contaminated grain (Pyke and Novak 1994). Cheatgrass did not emerge as a noteworthy element in the Great Basin environment until the period between 1907 to 1930 (Morrow and Stahiman 1984). By 1930, it had achieved its current distribution (Pyke and Novak 1994). In the early 1900s, it had been noted in isolated places—notably embankments, railroads, and highways. During the next 3 decades, it spread rapidly into overgrazed sagebrush rangeland (Billings 1994).

Following disturbance by fire in areas where cheatgrass is present, it reestablishes from abundant seed. Even if fire destroys 90 percent or more of its seed, it can reestablish and compete significantly with native perennials (Bradley 1986a; Monsen 1992). Over a period of years, cheatgrass gains dominance over perennials and increases the flammability of the site (Peters and Bunting 1994). Repeated fire will diminish the perennial seed bank and allow cheatgrass to increase its dominance. Once cheatgrass becomes abundant enough to increase the likelihood of fire, repeated fires may occur frequently enough to eliminate shrubs such as sagebrush and native perennials. As wildfires become more common cheatgrass can essentially dominate a site (Monsen 1994).

Management Considerations—Native species can occupy sites that were dominated by cheatgrass, but this is not a common occurrence. Use of mechanical tillage, herbicides, and properly timed fire can be effective in reducing cheatgrass cover if other species that germinate under cool conditions can be introduced. Prompt rehabilitation of burned areas by seeding accompanied by livestock restrictions is important. Fire usually gives cheatgrass a competitive advantage. However, prescribed fire can be used to reduce cheatgrass and to allow seeded species a chance to establish. The narrow prescription window during which substantial seed can be destroyed is from the time cheatgrass becomes flammable, when it leaves the purple stage, until seed falls a short time later.

During the 1990s a greenstripping program gained favor. The objective was to reduce wildfire frequency and size by establishing strips of fire-resistant vegetation, such as forage kochia, at strategic locations on the landscape to slow or stop wildfires (Pellant 1994). Greenstripping is aimed at effectively disrupting fuel continuity, reducing fuel accumulations and volatility on areas with a high density shrub cover such as sagebrush, and increasing the density of plants that retain higher moisture contents.

Annual Grasslands

Pre-1900 Succession—In California where this type prevails, the Spanish settlers kept poor records, so knowledge of native vegetation types is poor. Many believe that the prehistoric vegetation was perennial (Garrison and others 1977), but meager evidence is available to support this belief. However, evidence from the early 1800s indicates dominance by annual grasses.

Post-1900 Succession—Intensive agricultural development has taken over much of the original annual grasslands. At the lower elevations of the ecosystem, cultivated lands make up one of the richest agricultural areas in the world (Garrison and others 1977). Remnants lie at upper elevations in the Sierra foothills, and many are components of a hardwood savanna or shrub savanna that are quite common in these foothills. The annual grasslands are quite responsive to rainfall, and productivity and species dominance both vary accordingly. Fire is very much a part of the ecosystem and does not seem to have detrimental effects. In fact, it is being used by ranchers to eliminate woody overstory species and enhance productivity of the grasses.

Management Considerations—The most productive portions of the ecosystem are not producing annual grasslands, but rather agricultural crops. Clearly, as long as this activity can be sustained, it will remain the primary management activity in the “bottomland” portions of this system. In the upland portions, grazing and fire can be achieved to attain various management goals. However, they are both system disturbances and must be used judiciously. Annual rainfall is probably the most important consideration in applying management treatments in a manner consistent with ecosystem viability. Drought years should probably not be accompanied by intensive disturbance activities.

Desert Shrublands

During the era of Euro-American settlement, fire frequencies initially increased. Newspaper records between 1859 and 1890 report that settlers engaged in active fire suppression, including deliberate overgrazing of rangeland to reduce fuels. Woody species were favored by the reduction of grass and forb competition caused by overgrazing (Wright 1986). Grazing altered the role of fire in those desert areas once dominated by grasses. The consequent reduction of major fires was followed by shrub invasion into desert grasslands (Bahre 1985). Early 1900s wildland management policies continued to promote historical fire suppression and rangeland use in desert landscapes. A new management strategy was initiated when desert managers recognized that continued shrub encroachment was associated with overgrazing and fire reduction (Komerek 1969; Leopold 1924). Shifts in land management resulted in reduced grazing, increased fuels and, thus, changed the fire dynamics. Currently, burning of thousands of acres is becoming more common, and fire has become a serious management issue in some shrubland areas (Blaisdell and others 1982; Bunting and others 1987; Narog and others 1995; Schmid and Rogers 1988; Wilson and others 1995a).

Desert shrubland management traditionally focused on shrub eradication in favor of grasses. The objective was to improve forage for livestock and increase efficient management of range by increasing livestock and wildlife visibility. Fire, disking, herbicides, and heavy grazing were all commonly used. Often, the end result of this heavy range management was to decrease the amount of annual biomass and actually reduce the productivity of these ranges.

The use of fire in desert shrublands is controversial. Experts do not agree on historical fire cycles or what the land-use goal must be. Presently, desert range management practices rely on generalized studies made on limited areas. Anthropogenic influence has changed the vegetation and its dynamics in these dry sensitive areas. High fuel loading, from multiple branching shrubs, and contiguous herbaceous fuels are now common in many of these deserts. Fire can be used to achieve desired objectives in many of these desert shrubland communities (Bunting and others 1987; Lotan and others 1981; McGee 1977; Wright 1990). Fire also may contribute to the loss of desirable fire intolerant species that are sometimes replaced by less desirable fire tolerant species. The present resource management challenge is to determine which species to maintain and what management priorities are suitable for each specific area.

Sagebrush

Pre-1900 Succession—Historical accounts of sagebrush habitat are sketchy, but fires in big sagebrush were set both by lightning and humans. The many species and subspecies of sagebrush are quite susceptible to fire. Typical succession after fire would begin with a grass/forb dominance, and eventually lead to sagebrush recovery in 30 or more years.

Until the mid-1800s, the American bison was the primary herbivore impacting the fuels of sagebrush/grasslands (Young and others 1979). In the late 1800s, overstocked free ranging cattle led to a depletion of perennial grasses and other palatable forage. The subsequent introduction and spread of cheatgrass in the early 1900s corresponds with increased fire frequency and the reduction of big sagebrush. This, in turn, increased erosion and further damaged perennial native grass and forb components (MacMahon 1992).

Post-1900 Succession—Since 1900 the cultivation and abandonment of marginal land, abusive grazing, and widespread recurrent prescribed burning of sagebrush resulted in an imbalance between the numbers and sizes of shrubs, and associated native grasses and forbs (Blaisdell and others 1982). Thus, much of the resource potential of the sagebrush range was depleted. By 1936, 85 percent of sagebrush lands were considered depleted (Tisdale and others 1969). Prescribed fire was used to remove shrubs and replace them with native perennial grass forage (Cornelius and Talbot 1955; Pechanec and others 1954; Pechanec and Stewart 1944; Reynolds and others 1968). This ecosystem readily burns, particularly where there is a contiguous understory of grasses. Habitat changes coincident with increased fire have included plant community composition changes (Blaisdell 1949; Hassan and West 1986), altered soil seed banks (Blank and others 1995), and increased soil repellency (Salih and others 1973). The absence of sagebrush is often an indicator of past burns (Humphrey 1974). Secondary consequences of wildfires in sagebrush can include range deterioration, flooding, erosion, lowered grazing capacity, and reductions in the amount and quality of wildlife habitat. Extensive research has focused on rangeland degradation (Young and others 1979) and loss of productivity (Beetle 1960; Harniss and others 1981).

Management Considerations—Sagebrush land managers are now confronted with recovering its productivity. Sagebrush production loss continues even with recent improvements in management. Currently, the value of the sagebrush rangelands is being reevaluated. Multiple factors need to be incorporated into resource management plans. Big sagebrush can gain dominance over the herbaceous layer in 5 to 30 years after a burn. Season of burn modifies species dominance (White and Currie 1983) and affects postfire sagebrush productivity (Mueggler and Blaisdell 1958). For example, silver sagebrush mortality is higher and regrowth is less after a dry fall burn (White and Currie 1983). After fires, sagebrush mortality is proportional to fuel reduction. Although many sagebrush species are readily killed by fire, at least three species (threetip sagebrush, silver sagebrush, and California sagebrush) are known to resprout (Malanson and O'Leary 1985; Tisdale and Hironaka 1981). Most sagebrush species reseed after fire, but may require fire intervals of up to 50 years to regain their dominance (Bunting and others 1987). Frequent fires can cause type conversion from sagebrush species to rabbitbrush, horsebrush, and snakeweed. Where wheatgrass occurs, the burn season is extended and wildfires are reported to consume more area per burn.

Introduced cheatgrass can outcompete indigenous herbaceous species. This brome is undependable forage because of its large fluctuations in yield from year to year. After two to three reburns, sagebrush sites can be converted to stable cheatgrass; fire return intervals of 5.5 years maintain cheatgrass dominance. Cheatgrass is often accompanied by other invasive, noxious, and undesirable species. Together these pose a serious fire hazard, particularly following wet springs.

Planning prescribed fires in sagebrush should include specific objectives and consider many factors such as species and subspecies of sagebrush, soils (Salih and others 1973; Simanton and others 1990), fuel loading, fuel moisture content, and windspeed (Britton and Ralphs 1979; Brown 1982). Early spring or late summer burns can be used to promote native perennial grasses. There is little postfire recruitment for 3 to 5 years following a fire in perennial grasses, yet surviving grasses and accompanying forbs increase biomass production. Often forbs will dominate an area for several years postburn. Harniss and Murray (1973) found increases in herbage production for 20 years after a burn.

Attempts at restoring sagebrush rangeland to achieve higher biomass yields are being investigated (Downs and others 1995). In general, shrublands that have been converted to grasses by large wildfires are difficult to restore. Fire negatively impacts soil seedbeds important for sagebrush regeneration (Blank and others 1995). Sagebrush seed can be viable up to 4 years. Sagebrush can be restored through reseeding. Cheatgrass seed banks present on sagebrush sites may negatively influence reestablishment of native bunch grasses and shrubs (Hassan and West 1986). If sagebrush is in good "natural" condition an initial postfire influx of cheatgrasss will occur. Given adequate precipitation, perennial native grasses and shrubs can outcompete cheatgrass by the second year (West and Hassan 1985). Postfire rehabilitation efforts can be unsuccessful if other measures such as grazing are not incorporated (Evans and Young 1978). Species and associations of the sagebrush-grass type are influenced by edaphics and microclimate (Meyer 1994). Restoration efforts are complicated by the level of site disturbance and ecosystem variability and specificity (Blaisdell and others 1982; Blank and others 1995). Wildfire in cheatgrass dominated sites may afford managers an opportunity to reseed with perennial grasses and reduce the cheatgrass to lengthen the fire return interval. Presence of woody fuels may provide a hotter fire that can kill more cheatgrass seeds. Herbicide applications may facilitate native shrub and grass reestablishment (Downs and others 1995).

Wildlife such as pronghorn, deer, elk, coyotes, rabbits, rodents, and an endangered prairie dog reside in sagebrush rangelands. Abundant avifauna (over 50 species) that nest and feed in sagebrush include eagles, hawks, owls, doves, chukar, and sage grouse. Wild ungulates and domestic sheep may benefit from the maintenance of high quality sagebrush browse (Rodriguez and Welch 1989). Wildfires have removed large areas of sagebrush and may have destroyed a significant amount of sage grouse habitat (Downs and others 1985). Short- and long-term effects of fire on wildlife in this habitat need further evaluation (Gates and Eng 1984).

Blackbrush

Succession—Historical documentation of blackbrush fire cycles is limited. As late as 1981 (Lotan and others 1981; Martin 1975), land managers did not perceive desert fires as a serious land management problem because of small fire size and minimal damage to resources. Current data refute this perspective (Narog and others 1995; Wilson and others 1995a, 1995b). Cyclic desert precipitation above 10 to 14 inches (25 to 36 cm) may increase biomass and fuel continuity enough to increase fire behavior potential.

Since 1900, it appears that neither fire nor exotic annuals have altered soil microflora apparently required for blackbrush survival or reestablishment. However, burning has promoted succession to grassland by destroying the cryptogamic crust that stabilized the soil. Frequent large fires have eliminated blackbrush from some areas. Some sites show no recovery after almost 4 decades (Wright and Bailey 1982). Currently, burning is not a recommended practice for range enhancement purposes in this shrub type (Callison and others 1985) because blackbrush is often replaced by species of similar forage potential.

Management Considerations—Fire has been used for range improvement by reducing the shrub to grass ratio in areas where shrubs are gaining dominance. Land managers must also focus on protecting cacti and succulents, which will complicate fire management because of their various responses to fire (Thomas 1991). Fire may continue to be a necessary tool to modify fuel buildup. Currently, increases in desert shrubland fires and fire size have become a serious concern particularly with the recent increase in urban encroachment and resource degradation issues on these lands.

Research is needed to develop management and restoration recommendations for blackbrush (Pendleton and others 1995). Fire destroys the short-lived blackbrush seedbanks (produced by masting) necessary for it to reestablish. High temperatures, wind, and low humidity are usually required to propagate fire in blackbrush. If blackbrush becomes decadent or in some way presents a wildfire hazard, removal by burning may be appropriate. In some cases mature shrubs may survive low intensity fires; however, fire generally kills both seeds and mature shrubs. Although blackbrush is somewhat effective for erosion control, it may take more than 60 years to reestablish after a disturbance such as fire (Bowns and West 1976).

Wildlife such as deer, elk, desert bighorn, pronghorn, squirrels, rabbits, and game and nongame birds use blackbrush for cover, browse, and seeds. Livestock are more limited: sheep and goats browse blackbrush, but its low palatability and nutritional value make it unsuitable for cattle and horses.

Saltbush-Greasewood

Succession—Little is written regarding historic fire patterns in the saltbush-greasewood type. In some areas little change has occurred since 1900 in black greasewood dominated vegetation, while in others both saltbush and black greasewood have expanded into areas previously dominated by sagebrush (Sparks and others 1990). Rangeland seeding and invasion of grasses forming a highly flammable understory have increased the fire frequency in the saltbush-greasewood type. Postfire recovery is often rapid due to postfire resprouting and vigorous reseeding strategies used by the various shrub species in this vegetation type.

Management Considerations—In the past fire management was not a concern in saltbush-greasewood vegetation because sparse understory, bare soil frequently found in intershrub spaces, and the low volatilization of many saltbush species made this vegetation type resistant to fire (Tirmenstein 1986). These communities may burn only during high fire hazard conditions. In wet years brought by El Nino, such as 1983 to 1985, fine fuels may become contiguous across otherwise gravelly soils. Recently these fine fuels have become a fire hazard problem (West 1994). Grazing and other disturbance can encourage increases in biomass production, especially in the spring (Sanderson and Stutz 1994). Introduced cheatgrass has increased the fire risk, particularly when the area is ungrazed (West 1994). Disturbance may also allow this vegetation type to increase its range. Many species in this type resprout (West 1994). Black greasewood vigorously resprouts after fire or other disturbance. Season of burn, fire intensity, and fuel loading may be important factors to consider when using fire to regenerate or increase the productivity of this vegetation type (Harper and others 1990). Intense fall fires may increase plant mortality in spite of a species' resprouting potential. Some *Atriplex* species resprout and others produce abundant seeds. Thus postfire reestablishment from onsite and offsite seed sources is possible.

Saltbush-greasewood vegetation provides valuable forage for livestock and wildlife, particularly during spring and summer before the hardening of spiny twigs. It supplies browse, seeds, and cover for birds, small mammals, rabbits, deer, and pronghorn. Saltbush and black greasewood can be used to revegetate mine spoils and stabilize soils. Saltbush concentrates salts in leaf tissue and may be used to reduce soil salts and reclaim degraded land for agriculture. Outplanting methods are being developed for saltbush restoration projects (Watson and others 1995).

Creosotebush

Succession—Historically, creosotebush was restricted to well-drained knolls and foothills. However, by 1858 it had begun to invade the grama grasslands and by the early 1900s creosotebush had encroached into areas dominated by grasslands (Valentine and Gerard 1968). Overgrazing and drought contributed to the expansion of creosotebush range (Buffington and Herbel 1965). Fire suppression may be contributing to this expansion.

Management Considerations—Creosotebush invades desert grasslands. Although creosotebush may suffer up to 80 percent dieback during drought, it still resprouts (Humphrey 1974). On the other hand, it is sensitive to fire, especially in spring (Brown and Minnich 1986; McLaughlin and Bowers 1982). Fire and herbicides have been used to control creosotebush. High fuel loading and spring and summer burning will lead to higher creosotebush mortality from fire (Martin 1966). This indicates that wildfires could have kept it from invading grasslands before

Euro-American settlement (Wright and Bailey 1982). Selective thinning of creosotebush by fire suggests that this ecosystem is not resilient to burning and creosotebush may be replaced by other species, particularly with recurrent fires (Cable 1973). For example, bush muhly growing under creosotebush canopies may out-compete smaller shrubs and become the dominant after fire. Following heavy precipitation, herbaceous fuel increases and may increase fire potential in the creosotebush vegetation type (Brown and Minnich 1986). Creosotebush can withstand some fire exposure (O'Leary and Minnich 1981). Brown and Minnich (1986) report slow recovery for creosotebush after low-severity fire, and limited sprouting and germination were observed after fire in most of the species in the creosotebush associations.

Sheep will use creosotebush for cover, but creosotebush is unpalatable browse for livestock and most wildlife. However, pronghorn, bighorn sheep, mountain goats, game and nongame birds, fox, small mammals, and many reptiles and amphibians are some of the wildlife that use creosotebush for cover and its seed for food. Interestingly, the protected desert tortoise (*Gopherus agassizi*) typically burrows in soil stabilized by this plant (Baxter 1988). Creosotebush can be outplanted to facilitate rehabilitation of disturbed desert areas where it improves microsites for other plants and for fauna.

Creosotebush-Bursage

Fire use prior to 1900 may have limited the range of creosotebush-bursage and kept it from invading desert grasslands (Humphrey 1974). Since the early 1900s white bursage has become dominant to creosotebush on disturbed sites. McAuliffe (1988) reports that creosotebush may use white bursage as a nurse plant. Bursage species are easily topkilled but can resprout. Following a fire, cover of creosotebush and bursage is reduced but then increases over time (Marshall 1994). Because fuel loading can vary seasonally and annually, fire management considerations in the creosote-bursage type requires a site-specific analysis of plant cover, fuel loading, and fuel continuity.

Mesquite

Succession—Mesquite density and distribution increased prior to 1900 with fire suppression and seed dispersal by livestock. After 1900 mesquite continued to increase even though numerous eradication practices such as biological control, herbicides, mechanical removal, and prescribed burning were used to limit its density and spread—with mixed results (Glendening 1952; Jacoby and Ansley 1991; Wright 1990; Wright and Bailey 1982).

Management Considerations—Fire as a management tool for controlling mesquite has its limitations. Mesquite may become more prevalent 5 years following a burn than it was before fire (Martin 1983). Mesquite can root sprout; top-killed individuals may resprout from dormant buds found in upper branches or from the base of the trunk below the ground surface. Mesquite seedlings can survive fire (Cable 1961), but on a burned site mesquite is sometimes reduced (Wright 1980). Fire may kill a good proportion of mature mesquite, particularly the smaller trees (<2 inch diameter) (Cable 1949, 1973). It is most susceptible to fire during the hottest and driest part of the year (Cable 1973). Drought years may increase mortality of mesquite if eradication is attempted. If managers wish to open dense mesquite stands, then roots must be killed, not just aboveground biomass. Fire can be used to reduce the density of young mesquite populations, particularly during dry seasons that follow 1 to 2 years of above normal summer precipitation (Wright 1980). Adequate precipitation, no grazing, and using fire about every 10 years allow grasses to successfully compete with mesquite (Wright 1980). Rehabilitation of mesquite-invaded grasslands requires removal of livestock before burning, otherwise the shrubs outcompete the grasses (Cox and others 1990). Shrub reinvasion depends on grazing management combined with continued use of fire at the desired frequency (Wright 1986).

In managing for mesquite savanna (Ansley and others 1995, 1996b, in Press; James and others 1991), shaded rangeland may be a preferred condition rather than attempting to completely eradicate mesquite (see the **Texas Savanna** section). Low-intensity fire may allow mesquite to retain apical dominance on upper branches while reducing overall foliage. Season, air temperature, relative humidity, and duration and temperature of fire were factors reported to affect mesquite response to fire (Ansley and others in press). Mesquite topkill is related to heat in the canopy, not at the stem bases. Single and repeated summer burns kill mesquite aboveground, but do not kill roots (Ansley and others 1995). Prescribed burning may be used to kill mesquite seedlings while leaving tree sized and shaped older individuals (James and others 1991).

Paloverde-Cactus Shrub

Succession—Prior to 1900, fires in paloverde-cactus shrub were not considered to be important and occurred mainly in the restricted desert grasslands (Humphrey 1963). Conversion of desert shrubland to grassland to enhance forage for livestock and wildlife had been the primary land-use goal during the 1800s (Martin and Turner 1977; Phillips 1962). The high shrub component in this desert is attributed to historic overgrazing and overburning.

Since 1900, increases in ignitions and fire size are evidence of changing land management practices in the paloverde-cactus shrub. Exotic grass invasion now supplies a contiguous fuel source in many areas so that the historical small and infrequent fires were replaced by more frequent and larger fires (Narog and others 1995). Rogers (1986) speculated that finer fuels and higher rates of spread may allow desert fires to become larger than nondesert fires before being controlled. Although many of the species in this vegetation type can resprout (Wilson and others 1995b), postfire communities generally experience changes in species composition, particularly with an increase in the grass component, at the expense of cacti and succulents (Cave and Patten 1984; McLaughlin and Bowers 1982; Rogers and Steele 1980).

Management Considerations—Current management policy for some of the paloverde-cactus shrub vegetation now includes multiple interests with an increasing emphasis on recreation and tourism. This new policy involving reduced grazing, an increasing number of ignitions, and a greater herbaceous component is altering the fire regime (Robinett 1995). Fire dynamics information is required to effectively manage these changing needs. The increase in fire frequency and size may have serious consequences particularly for plant and wildlife species of special interest such as the giant saguaro (Thomas 1991; Wilson and others in press) and the desert tortoise; both may be fire intolerant. Little information exists on maintaining desert species in the presence of fire. Restoration in the paloverde-cactus shrub type needs to be addressed if the thousands of acres recently burned are to be rehabilitated.

Southwestern Shrubsteppe

Succession—Historically, fire suppression and seed dispersal by herbivores have allowed grama-tobosa range to become dominated by creosotebush, tarbush, and mesquite. Tobosa is an early postfire seral component. Since the 1900s fire has been used to regenerate decadent stands of tobosa. Fire may stimulate or damage grama depending on climatic conditions, season, and fire severity. Reestablishment after fire is generally through stolons. Grama species can regenerate by seed, stolons, rhizomes, or tillering; tobosa mainly regenerates by rhizomes.

Management Considerations—Tobosa can be managed with prescribed fire, which causes low mortality, improves palatability, and increases biomass production. Tobosa is one of the few native grasses that have competed well with nonnative grasses. Spring burns produce the best results when precipitation is adequate. Litter of up to 3.0 tons/acre (6.7 t/ha) easily carries fire and is completely consumed. Broomweeds, snakeweeds, and firewhirls are prescribed burning hazards in tobosa. For optimum forage production prescribed burns should be conducted every 5 to 8 years on tobosa stands. Nonbunchgrass species of grama may take 2 to 3 years to recover following fire.

Grama and tobosa supply abundant forage for livestock and wildlife. Grama is palatable all year, but tobosa is poor forage in winter months. Black grama is drought adapted and can be used for restoration to prevent soil erosion.

Trans-Pecos Shrubsteppe

Succession—Historically, junipers were relegated to rock outcrops and upland limestone sites, preferring shallow limestone soils. Fire suppression and overgrazing have allowed the woody species to expand from their historically more limited range onto the mixed prairie, sometimes in dense stands (Sparks and others 1990). Dense juniper stands are highly competitive and reduce understory grassy forage. Junipers dominate over oaks on drier sites, are shade intolerant, and may be succeeded by pinyon pine. Junipers are facultative seral trees with extensive lateral roots that effectively compete for surface moisture in xeric environments. They may or may not root sprout depending upon species. Chemical control, mechanical control, and prescribed burning have been used to reduce juniper density to improve rangeland forage productivity.

Management Considerations—Management techniques to reduce juniper and shrub density to improve rangeland for livestock are employed in many areas. Prescribed burning in junipers is recommended to open dense stands; however, ground fuels are not always adequate to carry fire. Ahlstrand (1982) found that plant response to fire in this community is predominantly by vegetative means. He suggests that prescribed burning can be used to improve the grass component at the expense of the shrubs. Pretreatment with chemicals or mechanical methods is also recommended. A minimum of 1,000 lb/acre (1,120 kg/ha) of continuous fine fuels is needed for prescribed burns (Rasmussen and others 1986). Fire history studies suggest that fire-free intervals of less than 50 years restrict the expansion of junipers, and that nested fire cycles have actually driven the juniper's range (Bunting 1994). Fire rotations of 10 to 40 years are recommended to control junipers. Reburn intervals between 20 to 40 years or when junipers reach 4 feet tall are recommended to maintain converted grasslands (Wright and others 1979). Variable fire effects in this type can be obtained (Tausch and others 1995). For specific fire prescriptions refer to Wright and others (1979).

Mechanical treatment followed 5 years later by burning to kill saplings is recommended to maintain a landscape mosaic of open stands and grassland. Mature junipers in moderate to dense stands are resistant to fire, yet may suffer some mortality. Small stemmed individuals are easily killed by fire. Rapidly burning grass fires occurring at intervals of 10 years or more are adequate to allow juniper saplings to reach sufficient heights 3 to 6 feet (1 to 2 m) to withstand fire injury. Burned areas may be invaded through seed dispersal, and establishment can occur within 10 to 40 years (Rasmussen and others 1986). Dead junipers are volatile fuels, and spot fires from firebrands can be a problem.

Fauna in the Trans-Pecos shrubsteppe ecosystem are similar to the species found in desert grasslands. Pronghorn and deer are widely distributed across the shrubsteppe range as are dove, quail, rabbits, and small rodents. Javelina are common in the south. Common carnivores include coyote, bobcat, eagle, owl, and hawk. Juniper berries and acorns are a favored food by many species.

Chaparral—Mountain Shrub

Pre-1900 Succession—The species that we refer to collectively as chaparral evolved as a component of the understory of Laurentian forest types. They were adapted to harsh conditions and could withstand disturbance. Chaparral development had no particular relation to fire (Axelrod 1989). With warming and drying trends, chaparral species became more opportunistic and were able to fill niches once occupied by species less able to compete under these conditions. A disturbance that chaparral was able to cope with was fire—an element whose presence was probably important in providing opportunity for chaparral to attain status as a recognizable vegetation type. By the end of the 19th century, newspaper accounts of fires burning through this type for days and weeks in southern California became common. By accounts of historic fires, maps of vegetation from the first third of the 20th century, local lore, and by remnants of previous vegetation, a picture of chaparral's ecological amplitude begins to emerge with fire as an important environmental component. This logic continues into the 21st century.

Post-1900 Succession—The benefit of more or less real-time observation allows us the opportunity to fine tune our view of chaparral's role in succession. The dynamics of chaparral's environment have made it difficult to definitively document chaparral's role in the successional process. Several salient points can be made, however, with little fear of argument:

- Chaparral succeeds many forest types after a major disturbance—whether from fire or logging. It is often seral—especially at elevations where we currently consider chaparral as a montane understory type. Given a reasonable number of disturbance-free years, the forest type will regain dominance.
- Chaparral often succeeds chaparral after fire, especially at elevations where we consider chaparral as the dominant vegetation type. Species composition can shift drastically, probably depending on whether the fire occurred before or after seed set for a given species. The concept of an infinite store of chaparral seed in the soil is becoming more and more questionable due to seed predation by rodents, ants, and birds (Quinn 1980).
- The concept of chaparral being a fire climax refers to a delicate balance between characteristics of the chaparral species on a site and the fire regime. Fire frequency and timing can tip the balance so that chaparral can be overtaken by herbaceous vegetation types, such as annual grasses, and in southern California by an allied "**soft chaparral**" type—a highly volatile semiwoody group of shrubs. But the present fire regime appears to be about the same as during the presettlement period. Conard and Weise (1998) presented evidence that fire suppression has offset increased human ignitions during the past century, thus preventing fire frequency from increasing to the point of degrading the ecosystem. Area burned per year, size of large fires, and seasonality of fires in chaparral changed little during the past century.

Management Considerations—Management of chaparral has been directed primarily at concerns about fuel hazard, wildlife habitat, and as a cover type that plays an active role in maintaining slope stability and watershed capability. Some would prefer to manage chaparral as a problem because it occupies potential rangeland that could be used for livestock grazing. All can, in their place, be perfectly good reasons for managing chaparral, but you have to pick one—or maybe two.

Prescribed fire can be used to remove dead fuel for hazard reduction, increase structural diversity for wildlife habitat purposes, and increase the proportion of young biomass in a stand—for both hazard reduction and wildlife habitat improvement. In some areas, but not all, prescribed fire can be used to maintain stands of chaparral in their current state (that is, to maintain a fire climax). For prescribed fire to be successful, species that reproduce only from seed, the presence of seed must be assured. Some chaparral seeds need scarification, which fire often provides. Besides heat-shock scarification, smoke-induced germination is important to many chaparral species (Keeley and Fotheringham 1998). Seeds of chamise can be destroyed if directly exposed to fire. Many chaparral species sprout after fire; reproduction from seed is not as much of an issue for these species as long as individual plants are not killed. However, next to nothing is known about the effects of physiological age on sprouting ability of chaparral species.

Individual shrubs can be killed outright by fire. Shrubs lacking in vigor will probably not respond to fire in their normal fashion. Thus, stresses such as protracted drought might cause an unexpected effect if fire were to be introduced. An extremely severe fire can result in little reproduction from either sprouting or seed germination. A series of fires with short return intervals may result in reduced chaparral shrub density if shrubs burn before they reach seed-bearing age, or young shrubs developing from sprouts are physiologically unable to respond.

Extremely old chaparral stands can be found with little or no dead material in them, and others can be found with a significant down and dead component (Conard and Weise 1998). The difference can be dictated by species composition, site conditions, and history of the site. Management of stands with a lot of dead material in them has to be taken on a case by case basis. From a fuels standpoint, these stands do have an elevated hazard level. Whether or not they present a serious threat should be evaluated in light of their juxtaposition to other resources and the condition of the other resources. Old chaparral stands should not automatically be considered as “decadent.”

Conversion of chaparral to rangeland has to be undertaken with caution. Soil and slope conditions should be evaluated to avoid loss of soil. For this reason, steep slopes and easily eroded soils should be avoided in conversion projects. In all cases, chaparral management should be undertaken with a clear view of species present, site conditions, stand history, fuel situation, and successional potential.

Notes

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Chapter 7:
Fire in Tropical and Subtropical Ecosystems

Apart from savannas and grasslands, wildland fire in tropical environments has received scientific scrutiny only within the past few decades. It is now widely recognized that the vast majority of wildland fires occur in the tropics and subtropics (Goldammer 1993). Because a global treatment of the effects of fire in tropical and subtropical ecosystems is beyond the scope of this volume, the reader is referred to J. G. Goldammer (1990), wherein Mueller-Dombois and Goldammer (1990) outlined generalized tropical and subtropical fire regimes.

In this chapter, we focus only on fire effects in subtropical Florida, Puerto Rico, the United States Virgin Islands, and Hawaii by drawing on appropriate literature from southern Florida, the Caribbean, Mexico, and the Pacific Islands.

Understory Fire Regimes

Major Vegetation Types

In the regions covered here, major vegetation types having woody dominants exposed to surface fires that are generally nonlethal to the overstory occur only in subtropical Florida. They include (1) pinelands and savannas dominated by slash pine (var. *densa*), (2) wetland savannas and woodlands dominated by pondcypress, (3) cabbage palmetto forests and savannas, and (4) southern live oak hammocks (a local term for groves or forests dominated by hardwoods). Close analogs of these vegetation types are the Caribbean pine forests, woodlands, and savannas of the Bahamas, Cuba, and coastal Central America from Belize to Nicaragua; some of the other pine types in Cuba; the mountain pinelands of Hispaniola; and some of the seasonally inundated palm swamps and savannas in the tropical and subtropical Americas. There does not appear to be an obvious analog of the pondcypress type: a subtropical, fire-tolerant, wetland conifer forest or savanna, which occurs in seasonally flooded depressions throughout the Southeastern Coastal Plain of the United States, but reaches its greatest coverage in subtropical Florida. For a synopsis of the role and effects of fire in southern Florida ecosystems see Wade and others (1980) and Myers and Ewel (1990).

All of southern Florida's vegetation types characterized by understory fire regimes are found on low, flat, poorly drained substrates. The landscape consists of a vegetation mosaic where hydrology exercises considerable influence over the availability of fuels and

Table 7-1—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	Subtropical pine forest K116	S. Florida slash pine 111	M	1a: 1-5	m	1b			
		Cabbage palmetto-slash pine 86	M	1a	m	1b			
		Caribbean pine[c]	M	1a	m	1b			
	Palmetto prairie K079						M	1a	
Oak-gum-cypress 16	S. floodplain forest K113	Bald cypress 101			m	2a	M	2b,3	
	Cypress savanna K091	Pondcypress 100	M	1					
	Mangrove K105						M	2	
		Live oak 89	M	1b	m	2a			
Wet grasslands 41	Everglades K092	Marshes [c]					M	1a	
Tropical hardwoods [d]							M	2,3	
Melaleuca [d]			M	1	M	1,2			

[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 vegetation type fits as a subdivision of FRES or Kuchler types, but is not an SAF type.
[d] Special type for chapter 7, not a FRES, Kuchler, or SAF type.

interacts with fire in determining the dominance of species. Along a hydroperiod (period the water table is above the soil surface) gradient, pondcypress occurs on the wettest sites. Relatively higher sites support pine, palm, or oak. The juxtaposition of the pine, palm, and live oak-dominated vegetation is controlled to a large degree by landscape features such as ponds, lakes, and drainages that create fire shadows where the intensity and incidence of fire is reduced. Twentieth century anthropogenic alterations of both fire and hydrologic regimes, coupled with fire exclusion in some areas and the introduction of non-native species, has markedly altered some of southern Florida's historic vegetation patterns and fire regimes.

Fire Regime Characteristics

Fire is essential for existence of the southern Florida woody vegetation types having understory fire regimes. These types are subject to nonlethal, relatively frequent, low-intensity surface fires ignited by either lightning or humans, although lethal long-return-interval ground fires may be important in the development of certain types of pondcypress communities. The shortest fire return intervals of 1 to 5 years are found in the pinelands and savannas. The longest fire intervals measured in centuries are found in bald cypress forests. If fire is removed as a process, or if the normal range of variability of the fire regime is altered, the character of the ecosystem changes.

Lightning is frequently cited as the primary ignition source responsible for southern Florida's fire-maintained vegetation. Florida has the reputation of being the "lightning capitol" of the United States, if not the world, with frequent, intense electrical storms generated during the summer by a double sea breeze system operating from both the east and west coasts of the Florida peninsula. Lightning fires occur during every month of the year, but with peaks in ignitions and area burned concentrated during the summer months (Snyder and others 1990). This period coincides with the season of locally generated convectional storms (fig. 7-1). The greatest area burned from lightning-ignited fires usually occurs at the onset of this season when fuels are dry and wetlands exhibit their lowest water levels.

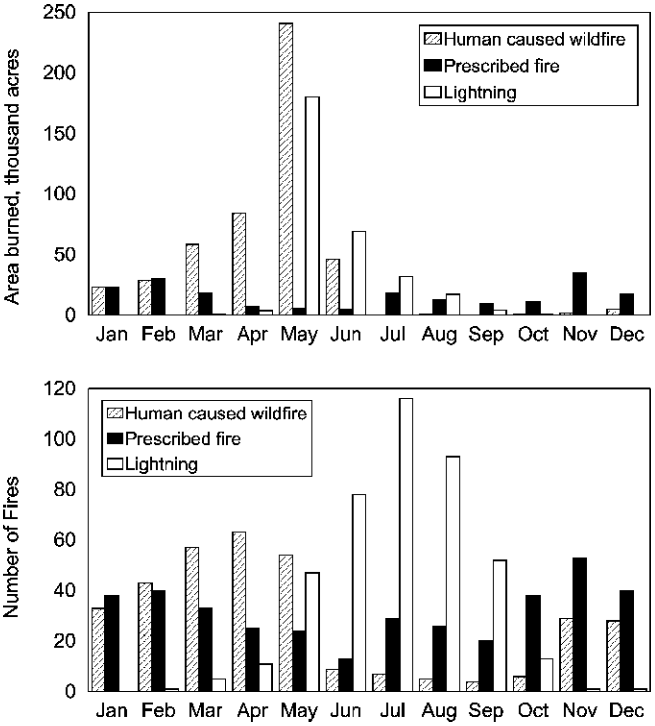

Figure 7-1—Monthly distribution of total number of fires and area burned from 1948 through 1997 in Everglades National Park, Florida, by ignition source. Courtesy of John Segar (Everglades National Park) and James Snyder (Big Cypress National Preserve).

We have little historical documentation of the burning activities of pre-Columbian Native Americans, so it is difficult to assess their influences on southern Florida fire regimes. The Calusa Indians disappeared shortly after contact due to European diseases and Spanish slave raiding parties from Cuba and the Greater Antilles. Because there were no Spanish settlements in, nor expeditions through, southern Florida, there are no known written records of Calusa burning practices. However, the conventional wisdom that lightning ignitions were the sole determinant of southern Florida's fire regimes has been challenged (Myers and Peroni 1983; Robbins and Myers 1992; J. R. Snyder 1991). Extensive, ecologically significant Native American burning in southern Florida can be inferred from archeological surveys, especially the number and distribution of occupation sites; from documented burning practices by the post-Columbian Seminoles, who were forced into southern Florida in the 1700s; by extrapolating from general principles of aboriginal burning elsewhere; and most significantly, from the aseasonal flammability of southern Florida's fuels. The latter means that the landscape could burn at any time of the year, and extensive areas possessed available fuels outside of the summer lightning season.

The most compelling evidence suggesting an important aboriginal influence on fire regimes in subtropical Florida is provided in a vegetation and fire history of Andros Island in the Bahamas (Kjellmark 1995, 1996). Alternating periods of human occupation and depopulation over the past several thousand years are well documented on Andros Island. Pollen analyses indicated that during periods of human occupation, pinelands of Caribbean pine expanded at the expense of tropical hardwoods. During known periods of depopulation, the pinelands contracted considerably, persisting only on the western side of the island where lightning-caused fire may have been more prevalent. The Bahamian pinelands are nearly identical in structure, function, and species composition to some of southern Florida's pinelands.

Florida's present-day human-caused fires are concentrated in January through May with the greatest area burned in March and April. This predictable dry period, occurring before the onset of the lightning season, coupled with the year-round availability of fuels in some vegetation types, suggest that Native American burning likely shifted the seasonal component of fire regimes, increased the incidence of ignitions, and ignited fires at times when wetland vegetation types were likely to burn extensively. This suggests that reliance solely on lightning ignitions in the Everglades might lead to a landscape that may never have previously existed.

Fuels

The fuels controlling Florida's understory fire regimes consist of ground cover of perennial grasses, low woody shrubs, saw palmetto, and surface litter. Due to the presence of volatile oils and waxes, live surface fuels can readily carry fire, contributing to Florida's year-round incidence of fire. Postburn accumulation of fuel is rapid as most grasses, shrubs, and palmetto resprout within a week of the burn regardless of the season. In denser pine stands, needle drop from crown scorched trees can form a continuous litter fuel bed within weeks of a burn. This rapid accumulation of fuel allows for low-intensity reburns on some sites within a year. Sites without pine, or with open stands, may require up to 3 years of fuel accumulation before an effective burn can recur. In circumstances where fuels accumulate for a decade or more, the probability of lethal effects increase, either due to crown scorch and consumption, or through smoldering ground fires in the accumulated duff at the base of trees.

Although many of southern Florida's fuels are available throughout the year, a predictable dry period begins in the winter months and continues until the onset of the summer convectional storms. At times, this dry period is protracted leading to severe drought in March, April, and May. Because water levels have receded and fuel moistures are low, more fuels are exposed, increasing loadings on local areas and on a total area basis. These drought conditions may be exacerbated further if the winter includes widespread frosts or freezes, which top-kill many plant species of tropical origin.

Pinelands—There are two types of subtropical pinelands—flatwoods on sandy soils and pine rockland on limestone. Ground cover in the pine flatwoods consists of a diverse array of shrubs, grasses, forbs, and saw palmetto. The palm fronds, grasses, and pinestraw compose the fuel that carries the fires. Volatile compounds in the leaves of some of the grasses, particularly wiregrasses, the ericaceous coastalplain staggerbush and the aquifoliaceous gallberry and dahoon holly, wax myrtle, and the fronds of saw palmetto, allow for vigorous burning even at high dead and live fuel moistures. For example, the moisture of extinction of the NFFL Southern Rough Fuel Model (Albini 1976), which is representative of these pineland fuels, is a high 40 percent.

Pine rockland fuels comprise grasses, shrubs, and pine needles, which also are represented by the NFFL Southern Rough Fuel Model. In contrast to flatwoods, the shrub layer is composed primarily of species of West Indian origin including white bully, varnish leaf, cocoplum, and myrsine. Some of these shrubs will reach tree stature in the absence of fire. Of about 100 shrub species found in the pine rocklands, only seven come from the pine flatwoods flora (Snyder and others 1990). The herbaceous layer is a diverse mixture of herbs of both tropical and temperate origin, with a high percentage of endemic species.

Postfire Plant Communities

Pine Flatwoods and Pine Rocklands

Vegetation Dynamics—Fire is not a succession-initiating process because the pinelands are fire-maintained vegetation types. Fire is as vital as rainfall in maintaining the vegetation. Postfire species composition is virtually identical to the prefire vegetation composition. Some mortality of the overstory pine may occur in any or all age classes; the soil surface may be exposed, but only for a few weeks; the fuel biomass is reduced and nutrients are released. Released nutrients coupled with exposed soil interact to stimulate flowering in a number of the ground cover species. Some species, particularly the grasses, will rarely if ever flower without fire. The season when a fire occurs can have a strong inluence on the flowering response of some species (Robbins and Myers 1992).

The pinelands of southern Florida fall into two broad categories based on soil substrate and composition of the ground cover: subtropical pine flatwoods and subtropical pine rocklands. Pine flatwoods occur on flat, poorly drained acid sands that were deposited on ancient marine terraces. Across Florida, pine flatwoods compose the most extensive ecosystem type and form the fuel matrix in which many other vegetation types are embedded. Fires originating in the flatwoods have strongly influenced the structure, composition, and juxtaposition of other fire-maintained and fire-influenced habitats, particularly cypress swamps, bays, marshes, and hammocks. See chapter 4 for additional discussion of pine flatwoods and other southern pine forest types.

Similar fire-maintained pine forests and savannas that occur on poorly drained acid sand substrates are found in western Cuba and on the Isle of Pines south of Cuba (Borhidi 1996). The pine species in Cuba are Caribbean pine and *Pinus tropicalis,* which appear to form vegetation complexes maintained by fire regimes identical to the regimes of slash pine and longleaf pine flatwoods, respectively. For example, like longleaf pine, *Pinus tropicalis* has a definite grass stage, increasing the probability that young individuals can survive frequent fires.

The pine rocklands, in contrast to the acid sands of the flatwoods, occur on alkaline limestone bedrock that forms a ridge running from north of Miami south into Everglades National Park. Pine rocklands occur elsewhere in southern Florida on outcrops of limestone, particularly in the Big Cypress National Preserve and on some of the Lower Florida Keys: Big Pine, Little Pine, No Name, Cudjoe, and Sugarloaf Keys. In some areas of the pine rocklands, exposed rock makes up 70 percent or more of the surface (Snyder and others 1990) (fig. 7-2). Although somewhat elevated from surrounding wetlands, the water table in the pine rocklands, like in the pine flatwoods, is at or near the surface during the wet season.

The pine species in both subtropical flatwoods and rocklands is the southern variety of slash pine (var. *densa*), which might be somewhat more fire tolerant than its northern Florida counterpart (var. *elliottii*). South Florida slash pine is described as having a fire-tolerant grass stage, similar to longleaf pine, that provides young individuals some protection from low-intensity surface fires (Little and Dorman 1954). The trait, however, is not nearly as pronounced as in longleaf pine. Seedlings of south Florida slash pine over a year old do develop a thicker stem and bark than the northern variety. This may offer south Florida slash pine relatively greater protection, but they possess nothing like the true grass stage of longleaf pine. The other pine species typical of Florida's flatwoods, longleaf pine and pond pine, do not extend into the subtropical zone.

As a mature tree, south Florida slash pine has thick protective bark that insulates the tree and high open branches that facilitate heat dispersal. Portions of the outer layer of the platelike bark also peel off as they are heated by a passing flame front, serving to dissipate heat. Like all of the fire-tolerant southern pines, south Florida slash pine continuously flushes new needles throughout the growing season. This gives trees the capacity to survive 100 percent crown scorch from burns occurring during the growing season, provided the buds are not killed. In contrast, equivalent levels of scorching from fires during the dormant season are more likely to lead to death of the tree, either directly or through stress-induced beetle infestation.

Figure 7-2—Pine rockland vegetation on Big Pine Key, Key Deer National Wildlife Refuge, Florida, consists of an overstory of south Florida slash pine, a diverse underwood of tropical hardwoods and palms, and a ground cover of grasses and forbs. Photo by Ronald Myers.

Other than the marked floristic differences between pine flatwoods and pine rocklands, they are quite similar: an open overstory of pines and a continuous ground cover of herbaceous and low shrubby fuels. Fire behavior and the historical range of variability of their fire regimes are nearly identical: frequent (every 1 to 5 years), low-intensity surface fire that can occur at any time of the year. Large severe fires are associated with predictable drought, primarily in March, April, May, and June, but can also occur at other times of the year. Smaller, and perhaps patchier, fires occur at more humid and flooded times of the year. Locally intense fires may be associated with fuels accumulated from tropical storm damage.

In general, the majority of the larger pines survive the fires, with mortality occurring in clumps possibly associated with areas of high fuel accumulation or greater intensity generated by interacting flaming fronts. Intensity being equal, pine mortality is greatest after fall and winter burns. Likewise, seedlings and saplings survive fires in isolated clumps that may be associated with gaps created when adult trees were killed by previous fires.

In spite of similar fuel characteristics and dominance by the same pine species, the developmental trajectories of pine flatwoods and pine rocklands in the absence of fire are markedly different. On better-drained sites, long-unburned flatwoods develop into evergreen hardwood hammocks dominated by oaks, particularly live oak. On poorly drained sites they develop into evergreen bays (red bay, sweet bay, loblolly bay, and other genera). The changes are relatively slow because propagules must arrive from pockets of hammock or bay forests persisting in fire shadows, drains, or hollows. In contrast, pine rocklands, in the absence of fire, take on the character of tropical hardwood forests (hammocks) within a few fire-free decades as the species in the extant shrub layer reach tree stature, and as other tropical hardwoods invade from tropical hammocks scattered through the rocklands. Once a significant midstory of hardwood vegetation forms in both pine flatwoods and rocklands, a fire can become lethal and severely damage the pine overstory. In such cases, postfire recovery of the pines depends on seed sources, environmental conditions for regeneration, and subsequent return of frequent fires. In the continued absence of fire, such a site rapidly becomes a dense hardwood forest dominated by either temperate or tropical species.

Unlike many Western forests maintained by understory fire regimes, southern Florida's pinelands did not experience broadscale changes due to 20th century fire suppression efforts. The loss of flatwoods and rocklands habitats has been largely due to urban development and associated fragmentation rather than successional changes brought on by successful fire suppression. Although fire prevention efforts and suppression activities were implemented in southern Florida, woodsburning has a long tradition throughout Florida and the Southeastern United States. Where fuels are uninterrupted across broad landscapes, fires are common. Fires are set by ranchers, hunters, arsonists, and by accident. The exclusion of fire has been more pronounced in fragmented, developed landscapes along the coasts; in relatively small parks, preserves, and refuges; and in areas where suppression equipment had ready access.

Management Considerations—Both the pine flatwoods and pine rocklands in southern Florida have been impacted by the interaction of fire and the spread of non-native species. Remnant rocklands that burn infrequently have been invaded by Brazilian pepper, a subtropical shrub that, once established, will persist in the shrub layer even if fire is reintroduced. Many pine rocklands are also threatened by Burma reed, an exotic grass whose spread is facilitated by frequent burning. The acid sandy soils of wetter pine flatwoods are susceptible to invasion and complete dominance by melaleuca. Melaleuca invasion can lead to a mixed fire regime consisting of some understory burns and some crown fires that are nonlethal to melaleuca but lethal to the pines. Both pineland types are threatened by cogongrass, a pantropical grass known for its close association with fire. In stands of pine that have escaped fire for several cycles and in other woodland vegetation, Old World climbing fern is becoming a problem by creating ladder fuels that carry fire into tree canopies. Fires in the pinelands that normally would stop at the margins of flooded hardwood and cypress swamps can burn into these vegetation types when their canopies are covered with the fern. Burning mats of the light-weight fern break free and are kited away by convection columns, igniting spot fires well downwind from the main fire.

Because of a long history of burning, first indiscriminate then prescribed, fire has been much more readily accepted as an essential land management activity in Florida than it has been elsewhere. Florida law allows the State to conduct hazard reduction burns on unoccupied private wildlands, and it recognizes prescribed burning as a right of the landowner. Prescribed hazard reduction burns and ecological management burns began in the Everglades in the 1960s. Today, the National Park Service's Big Cypress National Preserve burns more acreage annually than any other National Park Service unit in the United States, and more than any other publicly administered wildland in Florida.

Failure to burn either pineland type at frequent intervals (2 to 7 years) leads to rapid fuel buildup, changes in the vegetation structure, changes in species composition, and eventual habitat loss. Selection of an appropriate fire intensity depends on ecological and management objectives. Burns can be conducted at any time of the year. With fires of equal intensity, the pines are more susceptible to stress from scorching, other damage, and death from fall burns. Dry season burns are more effective in removing duff and exposing the mineral substrate, but in the pine rocklands, deep burns may occur in organic matter that has accumulated within cracks in the rock. These may result in root damage and death of the pines, but also they may effectively reduce the density of shrubs on long unburned sites.

Many of the understory species respond favorably to growing season burns, particularly when occurring at the transition between dry and wet seasons. Generally, managers favor winter burns for fuel reduction and growing season burns for ecosystem maintenance.

Pondcypress Wetlands

Vegetation Dynamics—Cypress-dominated vegetation types in southern Florida cover roughly 800 square miles (2,000 sq. km) (Wade and others 1980). Two variants of cypress occur in southern Florida: baldcypresss and pondcypress (var. *nutan*s). Baldcypress grows in floodplain forests, around shores of large lakes, and in the interior of large cypress strands (broad vegetated drainage depressions), all of which burn at intervals of centuries. Pondcyress grows in frequently burned savannas, shallow depression ponds called cypress domes or cypress heads, and on the periphery of cypress strands (fig. 7-3).

Although the role of fire in the dynamics of cypress-dominated wetlands is poorly understood (Ewel 1995), it is known that nonlethal, understory fire regimes prevail in the pondcypress savannas and woodlands rather than the baldcypress forests. The state-transition model (fig. 7-4) illustrates that fire regimes of various types play a role in the dynamics of cypress-dominated wetlands, with understory fire regimes responsible for monospecific pondcypress forests and savannas, including what are known locally as cypress domes, strands, and dwarf forests (Wade and others 1980).

Cypress domes occur in circular depressions or ponds; cypress strands form in elongated shallow drainage channels. Dwarf cypress forests occur on shallow soils over limestone bedrock and may include domes, strands, and savannas. In both domes and strands, trees increase in stature from the periphery to the center or midline of the depression, giving the vegetation a domed or ridged appearance.

Numerous theories involving the interplay of soil depth, hydroperiod, water depth, and fire have been postulated to explain the cause of the domed and ridged appearance. Fires are generally more frequent around or along the periphery of domes and strands, and one study concluded that this "differential marginal fire theory" accounts for the doming and ridging (McJunkin 1977). As one enters a dome or strand, tree height increases, crown closure becomes more complete, and fine herbaceous fuels are either no longer continuous or are submerged. This, coupled with longer hydroperiods toward the interior, limits the influence of fires to those times when protracted drought coincides with fires originating in the surrounding vegetation, such as pine flatwoods, wet prairie, or cypress savanna or woodland. The longer interval between

Figure 7-3—Cypress-dominated landscape with cypress domes, woodlands, and savannas, Big Cypress National Preserve, Florida. Photo by Ronald Myers.

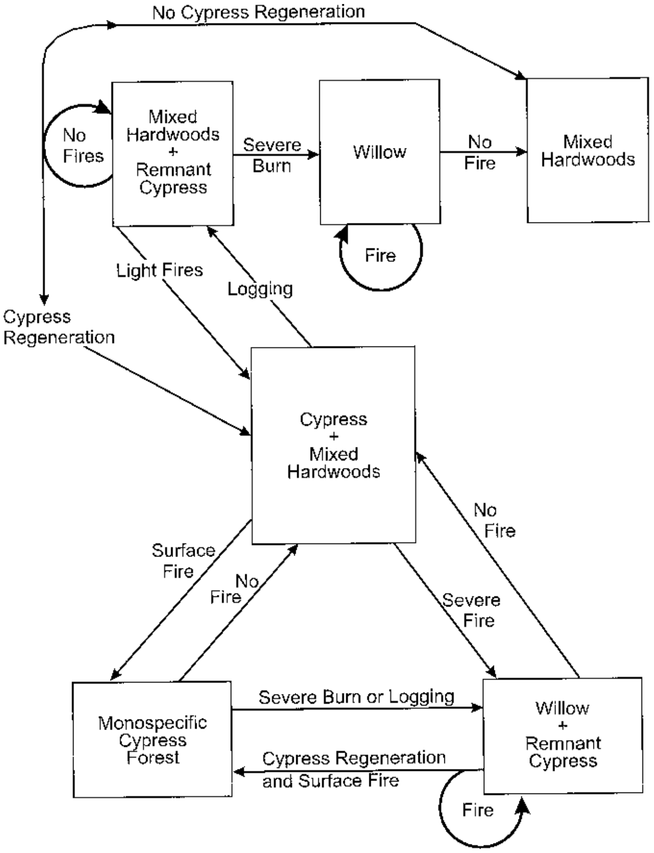

Figure 7-4—Generalized succession scheme for showing the role of different fire regimes in cypress wetlands and related vegetation types (Gunderson 1984).

fires coupled with increased flooding allows the accumulation of organic soils.

During those rare instances when fire penetrates to the interior of domes and strands, the organic matter may be consumed and the fires become lethal to the cypress causing stand-replacement, even though there is no flaming front. Such fires usually recur on the order of centuries. When they occur more frequently, open water or herbaceous marsh vegetation is present rather than forest. Some domes have an open water center giving them an appearance of a doughnut from the air. Similarly, the midline of strands may possess a series of ponds. Rare, severe fire events that consume organic soil probably play a role in the creation of these ponds.

Dwarf cypress forests and savannas occur on shallow sandy or muck soils overlying limestone bedrock. Density of stunted cypress trees varies from a few to nearly 700 trees per acre, with a ground cover of grasses, sedges, and forbs serving as the principal carrier of the surface fires (Wade and others 1980). Dwarf cypress forests form a continuum grading into cypress strands and domes as soils deepen, hydroperiods lengthen, and maximum water depth increases. Productivity and hydroperiod mediate the accumulation and density of the herbaceous fuel bed, which to a large degree controls the recurrence of fires. Return intervals may range from 5 to 20 years in dwarf cypress forests.

Little is known about the fire tolerance of pondcypress, but its bark has good insulation properties relative to other native wetland species, and the trees seem quite tolerant of low-intensity surface fires (Hare 1965). Limited studies have shown that pondcypress is much more likely to survive surface fires than competing hardwoods and shrubs (Ewel and Mitsch 1978), and years of observing wildfires and prescribed burning in the Big Cypress National Preserve attest to pondcypress' ability to withstand fire. Some trees, particularly those in dwarf cypress forests, possess the ability to resprout from their swollen buttresses if top-killed. Observations also point to the importance of fire in checking the encroachment of swamp hardwoods and permitting the establishment of cypress seedlings (Gunderson 1977).

After a long fire-free interval, pondcypress domes and strands develop a hardwood understory. In the continued absence of fire, the pondcypress is replaced by a hardwood-dominated swamp or bay. Once the shrub layer forms a continuous fuel bed under the cypress, conditions for lethal fires exist with the potential to eliminate the cypress overstory. It has been postulated that many of the bayheads (clumps of evergreen hardwood shrubs and trees) in the Everglades represent former cypress domes that were burned out during the droughts of the late 1930s (Wade and others 1980). If such fires include ground fuels, all woody vegetation would be removed; depending on the depth of the burn this would create open water, marsh, or willow, or permit re-establishment of cypress, provided there was a seed source. Without a ground fire, the hardwoods would resprout creating either a deciduous hardwood swamp dominated by red maple and Carolina ash, or a bay dominated by red bay, sweetbay, and loblolly-bay.

Shallower pondcypress wetlands are susceptible to invasion by melaleuca following fire. There are a number of examples where a single high-intensity fire in pondcypress that included scattered mature melaleuca led to the replacement of pondcypress by melaleuca.

Management Considerations—Most prescribed burning in cypress is limited to the savannas and woodlands that have herbaceous fuel beds. This effectively reduces the hazard of fire getting into deeper domes and strands during droughts. Burning during exceedingly dry periods is usually avoided to prevent excessive damage to trees on organic soils and to avoid extended smoke problems. However, preventing or avoiding "muck" fires will eventually lead to the loss of deeper water cypress forests and their habitats as they succeed to hardwood swamp forests.

Cabbage Palmetto Savannas and Forests

Cabbage palmetto-dominated savannas and forests are most prevalent on soils having a calcareous layer (shell, limestone) near the soil surface. The role of fire in creating and maintaining palm-dominated communities is poorly understood, but cabbage palmetto is tolerant of fire (Myers 1990a). Its single terminal bud, even as a seedling, is well protected from fire. The absence of a fire-sensitive surficial cambial layer in monocot stems allows palms to survive considerable charring, even some consumption of the stem. The palms appear able to withstand higher intensity fires than slash pine. Cabbage palmetto is frequently a codominant with live oak in hardwood hammocks. Where cabbage palmetto is the sole dominant, it may indicate that either the pine forest or live oak forest had been eliminated in a severe fire or series of fires, or perhaps that the pines had been harvested. In the absence of fire, oaks invade palm forests, eventually becoming the dominants.

Live Oak Forests

Live oak is generally considered a non-fire type or climax of pine and palmlands in the absence of fire. Live oak forests, however, do burn. Saplings and small trees have the ability to both resprout and to send runners forming clones. Large trees are protected by a relatively thick bark. Live oak litter produces a particularly compact fuel that limits flame length. It rarely burns without the momentum of a heading fire. The litter also holds moisture making it fire resistant except during periods of extreme drought. When it burns, flame lengths are short (<2 feet). Intense fire with some crowning may occur where the oak forest has expanded over a continuous saw palmetto fuel bed such as in long unburned pine flatwoods or dry palmetto prairies (Huffman and Blanchard 1991). Live oak forests are commonly embedded in flatwoods, prairies, and palmlands on sites somewhat protected from the frequent fires that normally spread through the matrix vegetation. For example, oak forests are frequently found on the leeward side of wetlands, ponds, lakes, and wet drainages where they would rarely be exposed to direct heading fires. The importance of these "fire shadows" was first pointed out by Harper (1911). Live oak forests that have gone for extended periods without being exposed to fire may exhibit a higher diversity of broadleaved trees and shrubs including other oaks and red maple. At more southerly and coastal locations in the Florida peninsula, this component takes on a more tropical character. Where oaks have expanded into other vegetation types due to fire exclusion, repeated growing season burning, undertaken before the summer rainy season, is most effective in killing oaks and restoring historic landscape patterns (Huffman and Blanchard 1991).

Mixed Fire Regimes

Major Vegetation Types

Melaleuca forests, woodlands, and savannas are an unusual wooded vegetation type dominated by an introduced Australian tree, melaleuca. It occurs in both Florida and Hawaii. In Florida, melaleuca is an aggressive invader of fire-prone wet prairies, shallow marshes, wet pine flatwoods, and cypress swamps (Meskimen 1962; Myers 1983); in Hawaii, it is a component of upland plantations composed of a variety of other introduced species. Although the Hawaiian melaleuca plantations are subject to anthropogenic ignitions creating significant fire control problems, it is in Florida where melaleuca seems most at home and appears to be gaining control of the fire environment (fig. 7-5). By the mid-1990s nearly 500,000 acres (202,400 ha) were infested with melaleuca, and of that, 40,000 to 50,000 acres were considered "pure" monospecific forests (Ferriter 1999).

Fire Regime Characteristics and Vegetation Response

The fire regime mediating Florida's melaleuca forests varies from one characterized by low-intensity surface fires in savannas, with some torching of individual trees, to high-intensity crowning fires in denser stands. Regardless of the fire intensity, little or no mortality occurs to any of the trees beyond the seedling stage (Myers and Belles 1995). The bole of melaleuca is protected by thick spongy bark. The outer bark layers carry fire into the crown, consuming branches and leaves and releasing seed from serotinous capsules. New branches sprout from the bark-protected larger stems. Fire in stands of melaleuca containing any mature capsule-laden individuals leads to the spread of the melaleuca forest into susceptible habitats nearby, resulting in a shift from a fire regime controlled by surface fuels to one dominated by aerial fuels. This is a fire regime heretofore unknown in the Florida environment and is likely to result in significant changes to wetland habitats, especially the species composition. Once melaleuca gets a foothold in a pine or cypress dominated habitat, the shift from a low-intensity to high-intensity fire regime results in the mortality of the native pine and cypress and subsequent conversion to melaleuca. This regime of varying fire intensity makes it difficult to categorize the melaleuca fire regime in table 7-1. Understory burns occur in melaleuca savannas. In mixed stands of

Figure 7-5—Torching melaleuca during prescribed research burn in a melaleuca-invested wet prairie, Big Cypress National Preserve, Florida. Photo by Holly Belles.

melaleuca and cypress or pine, the fires are lethal to cypress or pine but not to melaleuca. In pure melaleuca forest, high-intensity crowning fires are not lethal to the main stem of the trees. The combination of limited stem mortality and high-intensity fire is unusual in North American ecosystems. Placing melaleuca forest in the mixed fire regime is a compromise between low mortality and high intensity.

Management Considerations—Controlling the spread of melaleuca is a major concern and challenge on public lands in southern Florida. Successful control involves a strategy of first targeting outlying individuals and populations, then treating mature trees individually with herbicide, followed by prescribed burning after released seeds have germinated. Seedlings and saplings less than 3 feet tall are generally killed in these burns. Larger individuals will resprout (Myers and Belles 1995).

Stand-Replacement Fire Regimes

Major Vegetation Types

Vegetation types ranging from hardwood forests to grasslands in subtropical Florida, Hawaii, Puerto Rico, and the Virgin Islands are characterized by stand-replacement fire regimes.

Grassland vegetation types are considered to be stand-replacement regimes because the dominant aboveground vegetation is burned and replaced. When grasslands possess an overstory of trees—that is, a savanna type—the fire regime becomes an understory fire regime if repeated fires are primarily nonlethal to the overstory (see chapter 1). In Florida, the prairie types, known as dry and wet prairies, have ground cover and surface fuels that are identical to that found in pine flatwoods or pondcypress savanna without the overstory trees. The lack of a tree overstory, which determines whether a fire regime type is stand-replacement or understory, may be related to a history of frequent (nearly annual) burning or other aspects of fire and land use. Whether open pine, cypress or prairie, the fuel characteristics, burn conditions, and fire behavior are nearly identical, and the ground cover vegetation responses are the same.

In many tropical environments, including Hawaii, Puerto Rico, and the Virgin Islands, there is an interplay between lethal and nonlethal stand-replacement fire regimes, one fueled by grasses, the other by forest fuels. Fires originating in agricultural fuels, usually non-native range grasses, burn up to and penetrate forest edges killing trees and allowing grasses to encroach at the expense of forest. Once within the forest, these fires create feedbacks in future fire susceptibility, fuel loading, and fire intensity that favor grass fuels—a process, which if left unchecked, has the potential to transform large areas of tropical forest into shrubland, savanna, or grassland (D'Antonio and Vitousek 1992; Koonce and Conzales-Caban 1990).

The stand-replacement fire regime type also characterizes the fires that occasionally occur in mangrove vegetation, along with frequent fires in Florida's salt and freshwater marshlands, and fires that occur in a number of successional stage communities between marshes and swamp forest. These successional communities are dominated by willows, ashes, and bays.

Grasslands and herbaceous wetlands are common fire-maintained vegetation types in both Florida and Hawaii, with notable differences between the two locations. In Hawaii virtually all of the grasslands are dominated by introduced exotics, such as thatching grass, Natal redtop, molasses grass, broomsedge, fountain grass (fig. 7-6), and Columbian bluestem. These fuels have created altered fire regimes to which the

Figure 7-6—The introduced fountain grass retains dead leaves due to its bunchy structure and is highly flammable. It is an aggressive invader that readily replaces native plants especially where it carries fire into less flammable native vegetation. It invades lava beds creating continuous fuels where fuel breaks formerly existed. Photo by Jim Brown.

native Hawaiian flora are not adapted (Smith and Tunison 1992). In some cases, the nonnative grasses form the understory of introduced pine and eucalyptus plantations. In Florida a few introduced grasses are causing problems in pinelands, but Florida's extensive wet and dry prairies are native fire-maintained ecosystems.

In subtropical Florida, most native herbaceous vegetation types are associated with wetlands. They include salt marsh, sawgrass marsh, wet prairie, and miscellaneous broadleaved herbaceous marshes. The vegetation type known as dry prairie or palmetto prairie represents the ground cover vegetation of pine flatwoods or palm savanna without the trees. It is more common in the transition zone between temperate and subtropical vegetation in Florida and is on the periphery of the region discussed in this chapter.

Fire Regime Characteristics and Vegetation Response

Florida Freshwater Marshes and Wet Prairie—Freshwater marsh and wet prairie vegetation types include dense and sparse sawgrass marshes, wet prairie, marl prairie, spikerush flag marsh, beakrush flag marsh, and American white waterlily marsh (Kushlan 1990). Fire in these marsh types affects species composition and may limit or reduce peat accumulation. The sharp demarcation between different marsh types frequently indicates the boundary of a past burn (Kushlan 1990). The fire regime and fire effects in these herbaceous wetlands result from the interplay of hydroperiod and fuels, which together determine whether the fires are lethal or nonlethal to the dominant species, and whether or not fire is a succession-initiating process. In most cases, the aboveground vegetation is consumed, but the fires are usually not lethal to the dominant species that make up the fuel. These species simply resprout from underground buds, tubers, or rhizomes.

Two conditions can create lethal fires and initiate a vegetation change (Herndon and others 1991). One is when fire coincides with severe drought and consumes some, or all, of the organic substrate, destroying root systems and underground regenerative organs of the dominant species. The second occurs when water levels rise faster than vegetative regrowth after a burn and the site remains flooded long enough to cause the death of the vegetation. In either case, vegetation is replaced primarily by species present in the seed bank.

Sawgrass, once covering several million acres, is the dominant marsh vegetation type in the Everglades. Dense sawgrass marsh occurs on organic soils, while sparse sawgrass marsh is found on marl soils. Fuel loadings in dense sawgrass are sufficient to allow fires to burn over standing water, and lightning ignitions are common. These wet season fires go out when they burn into other marsh types. During the dry season, fires may burn through sparse sawgrass stands and wet praires with low fuel loads by smoldering through a dry algal mat called periphyton, composed mostly of filamentous blue-green alge, that forms over the substrate surface when the marsh is flooded. During severe droughts, organic soils may be consumed in both dense sawgrass and in many of the deeper water marsh sites. Sawgrass marshes and wet prairies can burn every 3 to 5 years (Wade and others 1980). In the absence of fire, sawgrass will succeed through a willow stage to hardwood bay vegetation.

Wet prairies are the least flooded of Florida's marsh vegetation types. The dominant species are identical to those in pine flatwoods and cypress savannas. Soils vary from periphyton-derived marls to sands. Dominants include maidencane, cordgrass, beakrush, and hairawn muhly. The fire frequency is 2 to 5 years.

Flag marshes are named after herbaceous broadleaved marsh species that have a flag appearance. They have a long hydroperiod and only burn during severe droughts. These fires may consume organic soils.

Except for the deepest water marshes, all types are susceptible to invasion by melaleuca. Dense sawgrass marshes and wet prairies are particularly susceptible.

Florida Salt Marsh and Mangrove—Although salt marshes occur in both the temperate zone and the tropics, fire mediates the tension zone between mangrove and salt marsh only in the tropics. In subtropical Florida, salt marsh is wedged between mangrove on the seaward side and freshwater marsh on its inland edge. Freezing temperatures probably have some influence on the juxtaposition of mangrove and salt marsh, but fires originating either in the salt marsh or further inland in freshwater marsh, control the inland advance of mangrove. When intense fires are stopped by mangrove, the outer fringe of trees is killed and the marsh expands (Wade and others 1980). Under moderate burn conditions the mangrove acts as a firebreak. Similar fire dynamics probably occur between marshes and mangrove in Cuba and the Bahamas.

Florida's salt marshes are dominated by black rush, gulf cordgrass, sand cordgrass, and inland saltgrass, mixed with a number of species found in fresh water marshes, notably sawgrass and cattail. Because salt marsh is under tidal influence and relatively isolated from human activities, a large proportion of ignitions are lightning caused. Fires supported by high fuel loadings frequently burn over standing water. Fire behavior varies considerably depending on the dominant species contributing to the fuel. Cordgrass fuel loadings are frequently as high as 22 tons/acre (49 t/ha) with a fuel bed depth of 8 feet. Beakrush fuel loadings and fuel bed depths are half that amount. Saltgrass fuel loadings vary considerably, and the fuel bed depth is only 1 to 2 feet (Everglades NP FMP 1991).

Mangroves rarely burn, but they are influenced by fire in seasonal environments such as in Florida. Mangroves are a tropical and subtropical forest type growing in brackish to high salinity coastal sites that have weak wave action. Four species of mangrove are found in Florida: red mangrove, black mangrove, white mangrove, and buttonwood. Each tends to be indicative of different zones of salinity or tidal influence. Lightning may be an important factor in the structure and dynamics of mangroves by creating numerous circular holes of dead and dying trees that may develop into patches of more flammable herbaceous vegetation. Fire is responsible for checking the encroachment of mangrove into salt marsh, and it is not uncommon to find red or white mangrove scattered through long unburned fresh water marshes. They have even been observed in the understory of cypress domes.

Florida Coastal Prairies—Coastal prairies are closely related to salt marsh; they occur in southern Florida, the Bahamas, and Cuba. They are less frequently inundated than salt marshes, but are sometimes flooded with salt or brackish water. Species composition includes saltgrass, seaside tansy, and batis. Coastal prairies are maintained by a combination of fire and hurricanes (wind damage and storm surge). If fire is absent for several decades, coastal prairies develop into buttonwood forests (Craighead 1971). The extent of coastal prairies may be a function of past clearing for charcoal production followed by cattle grazing and associated frequent burning. Coastal prairies on the Zapata Peninsula in Cuba have been contracting at the expense of buttonwood since the area was made a national park and the burning associated with cattle operations was curtailed (Myers 1999). See chapter 4 for additional discussion of freshwater and salt marshes.

Florida Tropical Hardwood Forests—Hardwood forests in southern Florida are usually islands imbedded in a matrix of marsh, prairie, or savanna. Fires burning in tropical hardwood forests (called hammocks in Florida), hardwood swamps, and bays or bayheads likely originate in the more easily ignited matrix fuels. The forest islands usually serve as effective firebreaks with fires burning only at their periphery. Fires have the opportunity to enter these ecosystems during extreme droughts, where they may cause conversion to earlier successional stages or shifts to marsh vegetation. Depending on fuel and weather conditions, vegetation structure, and type of substrate, the fires may be low-intensity surface fires, ground fires that burn out organic soils, or crowning fires moving through dense low shrubs, palmettos, or trees. Many of the tropical hardwoods have the ability to resprout if top-killed, but fires are lethal if the organic substrate is consumed. It is not uncommon for fires burning through grass fuels to go out at night as humidity recovers, but will hold over in hardwood forests, igniting the grass fuels the next day.

Hawaiian Forests and Grasslands—Fire regimes in Hawaiian native forest were probably always stand-replacement. Lava flows are the most common cause of natural ignitions, but resulting fires probably have not been an important evolutionary force shaping the characteristics of the vegetation. Some tree species such as koa and a'ali'i show some tolerance to fire, but these may represent preadaptations from their noninsular evolutionary environment (Smith and Tunison 1992). Most of Hawaii's native vegetation is extremely sensitive to fire. Historically, lightning may have caused some ignitions; today, lightning ignitions may be more common due to the prevalence of exotic grass fuels. Human sources, however, are by far the most frequent cause of fires.

The litter fuels in native Hawaiian forests, which range from dry forest to montane rainforest, are generally not sufficient to carry fire. The woody vegetation itself is not flammable. The presence of flammable grasses is essential for fires to spread. The grass fuels tend to be available throughout the year. Fire may have played a natural role only in the seasonal montane environment where, in places, native grasses and shrubs form a continuous fuel bed (Mueller-Dombois 1981).

Vegetation Dynamics—Although the vegetation, and thus fire regimes, of Hawaii changed dramatically in the later part of this century, some changes actually began over 1,600 years ago when the Polynesians colonized the Islands. The Polynesians encountered vegetation that was not fire prone, but, like indigenous peoples elsewhere, they used fire to manipulate the vegetation to plant crops, to facilitate travel, and to stimulate native grasses used for thatch such as pili grass. Early Euro-American visitors to the Islands reported encountering open savannas and grasslands at lower elevations and observed Polynesian burning practices (Kirch 1982).

Fire frequency and fire size greatly increased in the late 1900s when non-native grasses were introduced and spread (fig. 7-6). The invasion of fire-prone alien grasses coincided with the removal of feral goats from areas like Hawaii Volcanoes National Park. The grasses provided the fuel that carried fire into nonfire-adapted vegetation leading to the death of the native vegetation and its replacement by the grasses. Many fires are confined to mesic and dry forest habitats; however, grasses are also encroaching into some rainforest habitats and subalpine ecosystems. The prevalence of fire in these areas has been increasing (Mueller-Dombosis 1973).

Management Considerations—Managing fires in Hawaii is problematic. Fires of all intensities, timing, and sources are destructive to the Islands' native ecosystems, and most fires should be aggressively suppressed even if they result from natural ignitions. Prescribed fire has potential as a tool to reduce alien grass fuels and create firebreaks to prevent fires from entering sensitive native vegetation. Evidence indicates that prescribed fire may have a limited application in the restoration of a'ali'i shrublands and koa forests or woodlands because fire stimulates resprouting in both of these species. Prescribed fire might be useful in managing habitat for the endangered Hawaiian goose or nene (*Nesochen sandwicensis)* (Smith and Tunison 1992).

Forests of Puerto Rico and the Virgin Islands—Unlike Cuba and Hispaniola, which have native fire-maintained pine and palm forests and savannas and fire-maintained herbaceous marshes and wet prairies, Puerto Rico and the U.S. Virgin Islands do not possess any significant fire-adapted native vegetation types. Native forest types of Puerto Rico and the Virgin Islands include mesic forests on the windward sides that grade into rainforest or montane cloud forest with increasing elevation. The leeward lowlands support dry forest types. Fires occur in the dry forests, but the sparse accumulation of litter fuels only supports low intensity surface fires that generally go out at night as the humidity rises. If the thin-barked trees are top-killed, many have the ability to resprout, probably from an adaptation to drought, not fire. Where disturbances have created grassy openings, higher intensity fires may cause significant damage to the forest cover particularly at the forest edge. Repeated burning favors the grasses at the expense of forest. Similar fire damage occurs on more mesic sites where agricultural fires may escape and encroach into the forest during dry periods. This pattern of burning is more prevalent and a greater problem in other tropical areas where slash and burn (swidden) agricultural practices are common.

Severe fires have occurred in hurricane-damaged tropical dry and seasonal forests on the Yucatan Peninsula in Mexico (Whigham and others 1991). The potential for similar damage and subsequent fires exists in all of the tropical regions covered in this chapter.

Notes

Kevin C. Ryan

Chapter 8: Global Change and Wildland Fire

Global change, the combined effect of human activity on atmospheric and landscape processes (Vitousek 1994), affects all aspects of fire management. Scientists have documented changes in the global carbon cycle due to increases in atmospheric carbon dioxide (CO_2), changes in biogeochemical cycling due to increased nutrient deposition (for example, nitrogen), and changes in land use and cover. These changes are expected to continue for the foreseeable future (IPCC 1996a,b).

Changes in the global atmospheric chemistry are attributed to biomass burning and industrial processes. These alterations in the chemical makeup of the atmosphere are predicted to have a significant impact on biogeochemical processes and Earth's radiation balance, the so called "greenhouse effect." These changes in the chemical composition of the atmosphere and Earth's energy balance can be expected to modify precipitation, temperature, humidity, and vegetation development—all affecting fire management. In addition, historic changes in patterns of land use—roads, subdivisions, timber harvesting, farming, and ranching—have altered vegetation and fuels, affecting the potential ignition, spread, and severity of wildland fires. Continued migration of people into wildlands further complicates prescribed fire management and wildfire suppression.

Because of the complex interactions of all these processes, it is difficult to make definitive estimates about the rate and, in some cases, even the direction of change. However, given current knowledge, the anticipated changes can be expected to increase the pressure on fire management organizations. This chapter examines the complexity of global change and the possible influences on vegetation and fire management.

Changes Over Time

Vegetation and fire regimes have been in a constant state of flux throughout geologic time. Climate has changed throughout the millennia (Bradley 1999). New species have evolved as others became extinct, with climate and herbivory as the dominant influences. In the current geological epoch, the Holocene (0 to 10,000 years before the present), activities of humans have increasingly influenced vegetation and fire. Numerous climate fluctuations during this time include the Medieval Warm Period, AD 900 to 1350, and the Little Ice Age, 1450 to1900. Average temperatures have varied by as much as 5.4 °F (3 °C) over periods as short as a few decades (Bradley 1999; Mann and others 1999). Human societies (Ahlstrom and others 1995; Lipe 1995) and the prevalence of fire have been significantly affected by these changes (Bonnicksen 1999; Clark 1990; Swetnam 1993).

Prior to the arrival of Europeans in North America, native people routinely used fire to drive game animals and manage vegetation near encampments (Barrett and Arno 1982; Bonnicksen 1999; Boyd 1999; Clark and Royall 1995, 1996; Pyne 1982). In some regions Native Americans developed large agrarian communities where vegetation was extensively altered. Although the degree to which fire was used to initiate and maintain agriculture is uncertain, agriculture and harvesting of biomass for energy did lead to substantial change in fire regimes and vegetation in some areas. Across much of the landscape, lightning was the primary source of ignition. And fire spread was only hindered by the availability of fuels, weather, and natural barriers. Forest and rangeland sites developed with characteristic fire regimes and vegetation. Landscapes developed with a characteristic mosaic of stands of varying age, structure, and species composition. Fauna developed life cycle and behavior patterns tuned to these landscape patterns. While early Euro-American settlers may have seen many desirable features in existing patterns of vegetation, these features were not static but represented only a point in time in the development of North American vegetation (Betancourt and others 1990; Bradley 1999; Bonnicksen 1999; Delcourt and Delcourt 1987; Prentice and others 1991; Woolfenden 1996).

Since Euro-American settlement across the continent, fragmentation of the landscape resulting from agriculture, mining, and urbanization (Bahre 1991; Baker 1992; Veblen and Lorenz 1991) has significantly altered the fire potential of many ecosystems. This transformation of the vegetation prevented fires that formerly swept across prairies and steppes into adjacent forests (Gruell 1985). Domestic livestock have reduced the availability of fine fuels for fire spread. Grazing and fire exclusion together have led to the replacement of grasslands by shrublands in some areas (Wright and Bailey 1982). The introduction of exotic species has led to substantial changes in the species composition and fire potential of many ecosystems, particularly in arid and semiarid areas (Billings 1990). Timber harvesting has led to unnatural patterns of vegetation, modified fuel beds, and altered fire severity.

One of the most significant changes in land use in the 1900s was the suppression of wildfires. Fire suppression has led to changes in species composition and vegetation structure, and it has led to a significant buildup of fuels (Arno and Brown 1989) and increased forest health problems (Mutch 1994). The shift from ranching to ranchettes and urban encroachment on wildlands (Riebsame and others 1997) is also leading to a buildup of fuels. The result is that fires, though apparently less frequent than in the 1800s, are now often larger and more severe (Agee 1993; DeBano and others 1998; Sampson 1997) than formerly. When fires do occur they can result in serious threat to life and property. Larger, more severe fires also have greater potential to adversely affect postfire vegetation composition and structure, as well as soils and water, cultural resources, and air quality, as described in other volumes of the Rainbow Series ("Effects of Fire on Soil and Water," "Effects of Fire on Cultural Resources and Archaeology," and "Effects of Fire on Air").

Emissions from industrial processes, burning of fossil fuels, and slash-and-burn agriculture in the tropics have increased the concentrations of greenhouse gasses (GHG) in the atmosphere particularly in recent decades (Tett and others 1999). Chief among these is carbon dioxide (CO_2), but water vapor (H_2O), ozone (O_3), methane, nitrous oxides (NO_x), and various chlorofluorocarbons (CFCs) are also important (IPCC 1996a). Carbon dioxide has risen from approximately 270 ppm in the preindustrial atmosphere to around 365 ppm. While there is not universal agreement that increases in GHG have caused temperatures to rise, the observed 20th century warming reversed a millennial cooling trend. The 1990s were the hottest decade in the millennium, and 1998 was the hottest year (Mann and others 1999). There is growing scientific consensus that we are experiencing a greenhouse warming effect (IPCC 1996a).

Changes in the atmosphere due to GHG are expected to alter global weather patterns and significantly change regional climate. Due to the complexity of general circulation models used to predict climate change, there is much uncertainty as to the magnitude of effects of increased GHG on regional climate. Globally, average annual temperatures are expected to increase on the order of 2 to 8 °F (1.1 to 4.5 °C), depending on location. At this time estimates of regional climate changes are more tentative than estimates of global change, but increases are expected to be greater at high latitudes, in mid-continent regions, and in fall and winter (IPCC 1996a). The growing season may be extended by 1 to 2 months depending on latitude and altitude. Average annual precipitation may increase as much as 20 percent, but little summer rain is expected in much of North America's interior. Maritime climates may be wetter than today, but it is uncertain if increases in precipitation will be adequate to compensate for higher temperatures (Franklin and others 1991). Because continents are expected to warm up more rapidly than oceans, the interiors of the continents are expected to experience major drought by the middle of the next century (IPCC 1996a; Rind and others 1990).

Climate is generally defined as the 30-year average weather for a location. While temperature change (that is, global warming) is the major focus, the atmosphere, hydrosphere (water), cryosphere (ice), and biosphere (flora and fauna) are mechanistically coupled, and their interactions affect the relations between climate and wildland fire and between vegetation and fire regimes (K. Ryan 1991) (fig. 8-1). If the expected global warming occurs, the increase in CO_2 and changes in precipitation will alter growth and competitive interactions of plant communities. This will result in changes in ecosystem structure and species composition.

Climate change, therefore, will directly affect the frequency and severity of weather favorable to fire spread. Changes in plant communities will affect fire regimes indirectly by altering the physical and chemical properties of fuels. Fire is a major source of mortality in many communities, but in turn it creates gaps for new species. As a result, changes in fire regimes will modify the rate at which communities respond to climatic change.

Climate, Weather, and Fire Interactions

Regional fire potential is strongly coupled to regional climate. This connection between weather and fire potential is a fundamental part of all models and training in fire behavior, so clearly any significant climate change will affect the frequency and severity of conditions suitable for the ignition and spread of fires. In addition, each type of fuel has characteristic physical and chemical properties that affect flammability, and these properties vary with climate and weather. On the other hand, predicted temperature increases are not expected to have large direct effects on fire potential (fig. 8-2) but are expected to be correlated with a number of other changes that will affect fire potential (table 8-1). One of these changes is drying rate. Each type of fuel has its own drying rate, and its moisture content varies according to site moisture history. Decreased relative humidity can be expected to result in lower fine fuel moisture (fig. 8-2) and more rapid fire spread, whereas higher humidity can be expected to result in higher fuel moisture and less rapid spread. Therefore, if humidity patterns are altered, the moisture and combustion properties of fuels will change correspondingly.

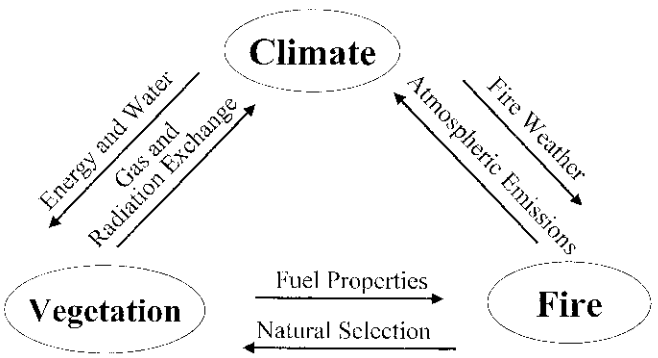

Figure 8-1—Climate, vegetation, and fire are dynamically coupled. Any change in one factor will lead to changes in the other two.

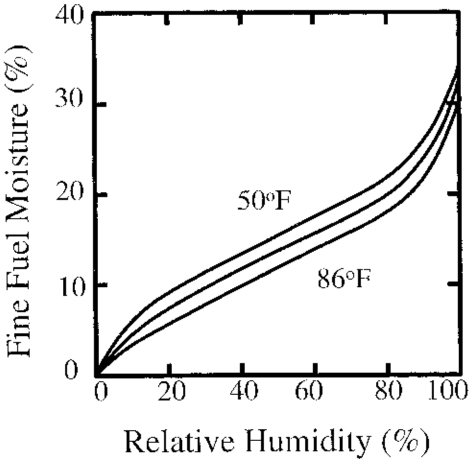

Figure 8-2—Fine fuel equilibrium moisture content as a function of relative humidity and temperature.

Increases in the frequency and severity of drought and an extension of the length of the fire season (Rind and others 1990; Wotton and Flannigan 1993) will result in more severe fires and increased consumption of long time-lag fuels (logs, duff, and organic soils). Wind is also a major factor in determining how fast fires spread. If the frequency of high winds changes, the potential for large fires will also change. Lightning is a major source of ignitions, and increased temperature, precipitation, and evaporation will change thunderstorm patterns. As a result, the frequency of lightning-caused fires is expected to increase some 30 to 70 percent depending on location (Price and Rind 1994). Further, much of the increase is expected to come during periods of moisture deficit.

Based therefore on climate projections, increases are expected in the length of the fire season (Wotton and Flannigan 1993), the frequency of lightning fires (Price and Rind 1994), the frequency of drought (Rind and others 1990), and area burned (Flannigan and Van Wagner 1991; Stocks 1993) in much of Canada and the United States; but some regions are expected to experience a decrease in fire activity (Bergeron and Flannigan 1995). In addition to these direct influences

Table 8-1—Fire interactions with the climate/weather system.

Climate/weather influences on fire	Fire influences on climate/weather
Relative humidity	**Carbon dioxide** (CO_2)
Wind (speed, persistence, extremes)	**Carbon monoxide** (CO)
Drought (frequency, persistence)	**Methane** (CH_4)
Length of fire season	**Water vapor** (H_2O)
Lightning (dry vs. wet)	**Particulates** (Pm 2.5, Pm 10)
Dry cold fronts (frequency)	**Nitrous oxides** (NO_x)
Blocking high pressure (persistence)	**Ammonium** (NH_4)
	Trace hydrocarbons
	Trace gasses (including VOC)

of weather on fire, storms are predicted to be more severe in the altered climate (IPCC 1996a). If so, increased wind damage to forests could greatly increase available fuels.

Complete combustion of biomass results in the production of CO_2 and H_2O, but combustion is rarely complete, and a variety of other chemical species are produced (table 8-1) (Crutzen and Goldammer 1993; Goode and others 1999; Hao and others 1996; Levine 1996; Ward and Hardy 1991). Globally, biomass burning is a major source of several chemical species in the atmosphere. Many of the compounds released by burning are greenhouse gasses. Particulate matter (for example, Pm 2.5, Pm 10) can produce a local short-term cooling effect by reducing solar heating. Particulates can also result in reduced precipitation (Rosenfeld 1999).

Biomass burning contributes to the overall problem society faces in managing greenhouse gasses and providing for clean air. See the volume “Effects of Fire on Air” in the Rainbow series for a state-of-the-knowledge review of the fire management issues associated with fire’s impact on local and regional atmospheric conditions.

Climate and Vegetation Interactions

Climate is considered the principal determinant of vegetation distribution throughout the world (COHMAP Members 1988; Neilson 1995; Woodward 1987). Solar radiation, temperature, humidity, precipitation, and wind all affect the physiological ecology of plants (Bazzaz 1996), thereby affecting their ability to complete life cycles and sustain populations (table 8-2). Vegetation, therefore, is governed by the cumulative history of climate, vegetation, and disturbance processes, and as climate changes, the distribution of the world’s vegetation will change. Moreover, the pattern and severity of disturbance, especially fire, will also change (Overpeck and others 1990, 1991; K. Ryan 1991).

Climate-controlled relationships between vegetation structure and species composition occur similarly along both altitudinal and latitudinal gradients. Changes occur along these gradients such that every 1,640 foot (500 m) increase in altitude is roughly proportional to a 171 mile (275 km) increase in latitude (Hopkins bioclimatic law) (McArthur 1972). Given

Table 8-2—Climate/weather system interactions with vegetation.

Climate/weather influences on vegetation	Vegetation influences on climate/weather
Solar energy	**Albedo**
Temperature	**Evapotranspiration** (H_2O)
Relative humidity	**Photosynthesis** (O_2)
Precipitation (timing, amount)	**Respiration** (CO_2, H_2O)
Atmospheric chemistry	**Methane** (CH_4)
Wind (direction, speed, extremes)	**Convection**
	Advection
	Desertification

time to establish equilibrium following an average annual temperature increase of 6.3 °F (3.5 °C), vegetation zones in the Rocky Mountains can be expected to shift approximately 2,167 feet (630 m) up mountain slopes (fig. 8-3), or 213 miles (350 km) farther north. The rate of vegetation movement associated with the shift in isotherms is several times faster than known species migration rates (Davis 1990; Gates 1990). Massive shifts in biome boundaries should be expected (IPCC 1996a; King and Neilson 1992; Neilson 1993; Overpeck and others 1991). In many cases, species will not be able to migrate and populations will become fragmented (Peters 1990).

Greenhouse changes will affect numerous biochemical processes that will alter ecological relationships (Joyce and Birdsey 2000; Schimel and others 1999). Photosynthesis, respiration, decomposition, and nutrient cycling will all be affected (Agren and others 1991; Bazzaz 1996; Long and Hutchin 1991; Mooney 1991). Responses of these four major physiological functions are interdependent and vary with temperature and atmospheric chemistry. However, they each exhibit different responses to changes in temperature. For example, the temperature response curves for photosynthesis and respiration differ, and an increase in CO_2 will have a major impact on photosynthetic rates. So the current balances within individual plants or communities cannot be projected into the future. Substantial increases in water use efficiency (ratio of the amount of CO_2 assimilated during photosynthesis to the amount of H_2O transpired) may result from increased atmospheric CO_2 (Bazzaz 1996; Houghton and others 1996; Mooney and others 1991; Strain 1987). As a result plant growth may accelerate greatly. However, all living cells respire, and the respiration rate increases with temperature (M. Ryan 1991).

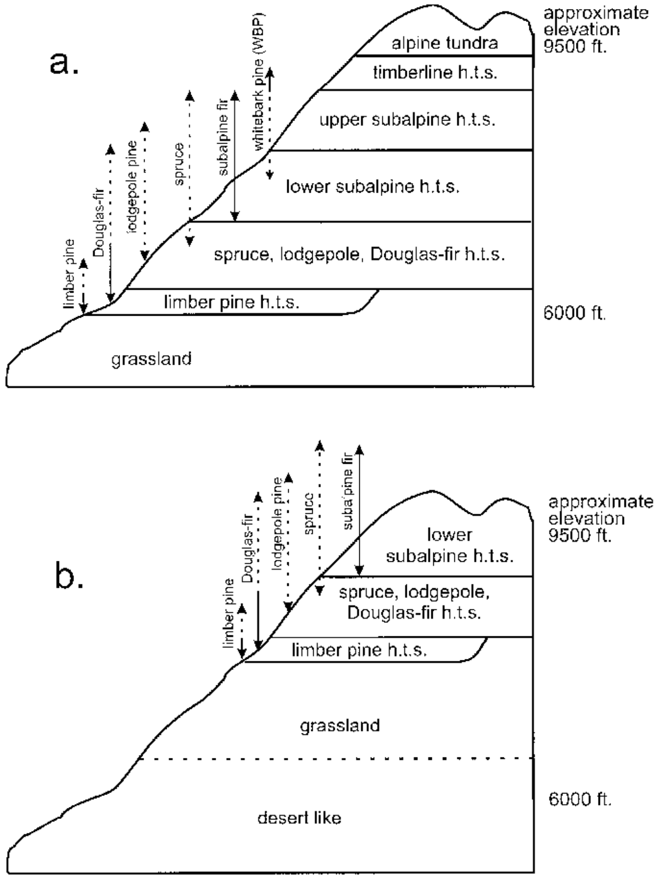

Figure 8-3—Current vegetation zones in the Bitterroot Mountains of Montana and Idaho as a function of elevation (Arno 1980) (A). Projected vegetation zone changes associated with warmer annual temperatures associated with a doubling of CO_2 in the atmosphere (B). This simple one-dimensional projection does not take into account the many dynamic interactions but does illustrate the relative magnitude of possible shifts in vegetation zones.

Because woody plants have more nonphotosynthetically active living tissue (for example, large root systems and sapwood) than herbaceous plants, temperature-influenced changes in forests and woodlands are expected to be relatively large compared to grasslands (IPCC 1996b), and mature forests are more likely to be severely affected than young forests (M. Ryan 1991; Waring and Running 1998). The loss of carbon during respiration will increase with temperature, thereby potentially reducing the effect of increased water use efficiency. If increased CO_2 alters carbon to nitrogen ratios of plants, then decomposition, nutrient cycling, and insect and disease resistance will be altered (Vitousek 1994).

Regional climate dominates the zonation of vegetation, but microclimate, soils, life cycle processes (for example, germination and growth), and ecological interactions such as competition, herbivory, and fire strongly affect the external morphology and physiological ecology of communities within vegetation zones. All of these can be expected to change in response to changes induced by greenhouse gasses.

Studies have not addressed interspecific and intraspecific interactions that affect growth rates, allocation (that is, how a plant's growth is allocated between leaves, roots, fruiting, and so forth), and community relationships (Joyce and Birdsey 2000; Mooney 1991). However, given the complexity of species traits, it is unreasonable to expect current community relationships to remain unchanged in the future (Delcourt and Delcourt 1987; Foster and others 1990; IPCC 1996a,b). For example, temperature, moisture, and photoperiod exert strong controls over phenology and growth.

Species are adapted to a range of seasonal patterns. Significant changes in these seasonal patterns can lead to asynchronous development, which can lead to reproductive failure and growth loss (Grace 1987). Also, height growth and foliage biomass have been shown to increase at elevated CO_2 levels (Kramer and Sionit 1987). If canopy structure changes, the light available for understory plants will be altered, and elevated CO_2 partially compensates for low light (Cure and Acock 1986). Because not all species sustain enhanced growth for long periods of time, the effect on the competitive relationships between overstory and understory species is uncertain. Water use efficiency varies by species, and some species respond to elevated CO_2 by increasing root to shoot ratios. Thus, competition for water and nutrients will change. Species with the C_3 photosynthetic pathway (for example, woody plants and "cool season" grasses) show greater increases in growth at elevated CO_2 than plants with C_4 pathway (for example, "warm season" grasses) (Houghton and others 1996; Smith and others 1987). Cheatgrass, an exotic C_3 grass in the Western United States, is especially responsive to elevated CO_2 (Smith and others 1987). Climate change is expected to favor early successional species assemblages over later ones (Bazzaz 1996).

Climate not only affects regional vegetation, but vegetation in turn affects both regional climate and microclimate (fig. 8-1, table 8-2). The character of surface vegetation affects the amount of solar energy absorbed versus reflected. Evapotranspiration from actively photosynthesizing foliage contributes substantial amounts of water vapor to the atmosphere, potentially affecting local precipitation. Both living and dead vegetation produce CO_2 during respiration and release a variety of other compounds to the atmosphere. Some of these compounds are greenhouse gasses (for example, CH_4), and some contribute to air quality problems such as regional haze and smog (for example, trace hydrocarbons).

Without question, global change has affected interspecific relationships and will continue to do so, likely at an accelerated rate. The effects will likely cascade throughout ecosystems. For example, increased water use efficiency of upland plants can be expected to reduce stream flows (Running and Nemani 1991), thereby affecting aquatic systems.

Interspecific relationships are too complex and poorly understood to be predicted. Given this complexity of the interactions, managers and policy makers are not likely to have significantly improved scientific bases for their actions, and many changes are likely to go undetected until major shifts occur.

Fire and Vegetation Interactions

The species composition of a community is determined by the successive birth and death of the individual community members. How fast vegetation responds to changing climate depends on species life histories, migration rates, and rates with which suitable regeneration gaps are created. Fire has played and will continue to play a significant role in determining vegetation physiognomy, structure, and species composition in the world's temperate and boreal ecosystems (Agee 1993; Crutzen and Goldammer 1993; DeBano and others 1998; Rundel 1982; Wright and Bailey 1982).

Fire is a major cause of plant mortality. For example, fire preferentially kills trees of short stature or thin bark (fig. 8-4). Likewise, fire creates gaps that new individuals colonize. Thus, changes in fire may greatly accelerate vegetation's response to changing climate. Such interaction of climate, vegetation, and fire has influenced the presence and rate of most ecosystem processes in forest and rangeland settings (Heinselman 1981). For example, fire return intervals influence the distribution of life forms and regeneration modes present on a site (Noble and Slayter 1980). The composition and structural integrity of some ecosystems are so strongly influenced by the fire regime that they are considered to be "fire dependent" (DeBano and others 1998; Habeck and Mutch 1973; Turner and Romme 1994; Wright and Bailey 1982). The severity of fire, which depends on the amount and type of biomass present and weather conditions at the time of the fire, exerts a strong influence on plant survivorship and regeneration (fig. 8-5). Therefore, altered fire regimes under a future of global climate change can be expected to accelerate vegetation changes on the landscape (King and Neilson 1992; Overpeck and others 1991;

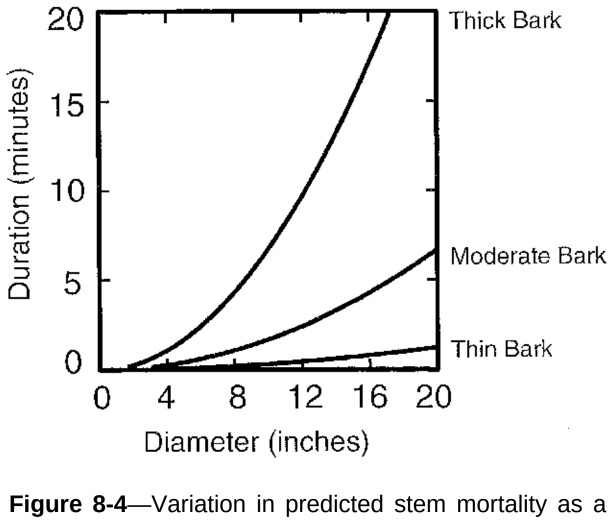

Figure 8-4—Variation in predicted stem mortality as a function of bark thickness and fire duration (adapted from Peterson and Ryan 1986). Area above a bark thickness curve implies cambium death, area below implies survival.

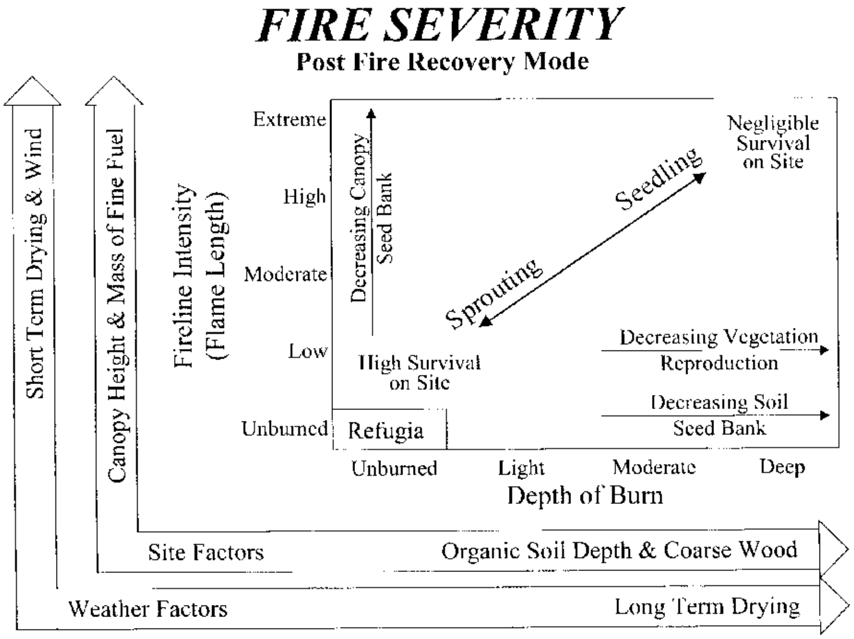

Figure 8-5—Postfire vegetation recovery varies with fire severity (adapted from Ryan and Noste 1985). In this concept of fire severity, the y-axis represents heat pulse above the fire and the x-axis represents the heat pulse down into the soil.

Romme and Turner 1991; K. Ryan 1991; Weber and Flannigan 1997).

Fire exerts selective pressure both at the individual plant and community level (Noble and Slatyer 1980; Rowe 1983). Short fire cycles favor species that endure fire by juvenile sprouting or evade fire by storing seed in the soil, invading from offsite, and having short life cycles. Intermediate fire cycles favor species that resist fires when mature or evade fire by storing seed in the canopy, but sprouting and invasion by offsite colonizers also occur. Long fire cycles favor species that typically avoid fire. Such species exhibit low resistance to fire injury and regenerate predominantly by seed. If the fire return interval is reduced to a period less than the time to sexual maturity, then a species will no longer be able to complete its life cycle on the site and could be lost from the site. The ensuing rate of reseeding will depend largely on the size of the area burned and the mobility of the seed.

The quantity, chemistry, and size distribution of fuels will change as species, growth patterns, and decomposition change (table 8-3). For example, high temperatures, drought, and nutrient shortages may lead to stress-induced mortality (King and Neilson 1992; Waring 1987) and early leaf senescence, thereby accelerating fuel accumulation. Also, an increase in carbon to nitrogen ratios will reduce decomposition (Agren and others 1991) and modify the role of fire as an agent of decomposition and nutrient cycling (Gosz 1981; Rundel 1982).

Table 8-3—Fire interactions with vegetation.

Vegetation influences on fire	Fire influences on Vegetation
Biomass ("loading" mass/area)	**Survival** (resistance to fire injury)
Bulk Density (mass/volume)	**Regeneration** (seeding vs. sprouting)
Size Distribution (surface area/volume)	**Injury** (stress and loss of vigor)
Chemistry (volatiles vs. nonvolatiles)	**Competition** (light, water, nutrients)
Live vs. dead ratio	**Community dynamics**
Shading/exposure	**Structural composition**
Strata (surface vs. overstory)	
Continuity (horizontal and vertical)	

Uncertainty of Interactions: Can We Predict the Future?

Interactions between climate, vegetation, and fire are complex and uncertain; thus, expectations for fire management are general and tentative. We can hypothesize how change in one factor will change another, but in actuality several climatic forcing factors will change simultaneously and initiate many internal adjustments within individual plants and communities. The relative abundance of species may shift because some are less adapted to the climate-altered site. Some species may regenerate but will be unable to successfully complete their life cycle given new climate and fire regimes. For example, redstem ceanothus, a valuable wildlife forage species in the Northern Rocky Mountains, and similar species that rely heavily on seed stored in the soil, sometimes for centuries, could be eliminated from sites by regeneration failure resulting from new climatic extremes, particularly early season drought and severe fire. If changes in climate and fire regimes lead to extensive species losses on a site, then migration of species from offsite will be accelerated. Species with wide ecological amplitude should be favored over those with narrow, specific habitat requirements. Regeneration strategies best suited to unstable conditions should also be favored. The additional environmental stresses and the increased frequency and severity of disturbance will likely favor the expansion of exotic and invasive species (Baskin 1998; Hogenbirk and Wein 1991).

Given changes in climate, soils, nutrients, and fire, many endemic populations will not be able to compete and successfully complete their life cycles on their current sites. They will become locally rare or extinct unless they are able to colonize new areas. Some species, particularly those that predominantly reproduce vegetatively or from seeds stored in the soil, are not highly mobile. While they may regenerate prolifically following site disturbance, they are less likely to take advantage of climatic-induced disturbance off site. These species should be slow to migrate to new areas that are within their ecological amplitude.

Given the altitudinal shift in life zones, numerous alpine species will become locally rare or endangered because there is no higher zone into which they can migrate (Franklin and others 1991; Peters 1990; Romme and Turner 1991). Similarly, subalpine species such as whitebark pine will be lost from all but the higher mountain ranges. Poor soil development will retard the migration of subalpine species into the former alpine zone, but montane species should migrate freely to higher elevations. If high temperatures and moisture stress severely limit productivity, they could threaten the continued existence of low elevation forests (IPCC 1996b). The advance of dry woodland and steppe species into these forests may be slowed by their lack of shade tolerance, but they should invade readily on sites disturbed by fire. Increased temperature and drought can be expected to increase the decomposition of peat soils and increase their susceptibility to fire (Hungerford and others 1995). The current boundaries between temperate and boreal forests, and between boreal forests and tundra, are expected to shift northward but not necessarily at the same rate.

In general, climatic change may be expected to result in improved habitat conditions at the cooler-wetter limits of a species' range and poorer conditions at its warmer-drier limits. However, many communities exist as "habitat islands" isolated by ridges or valleys or surrounded by cultivation and urban areas (Peters 1990; Peters and Darling 1985). These form effective barriers against species migrations. The rate of climatic change may be much more rapid than species' ability to migrate (Davis 1990; Gates 1990).

Understanding the potential impacts of climate change on vegetation and fire will require a level of integration not previously attempted in ecosystem studies (Mooney and others 1991). Several authors (Agren and others 1991; Franklin and others 1991; Keane and others 1997; Neilson 1993; Overpeck and others 1991) have attempted to understand the complex interactions by using process-based computer models to simulate long-term ecosystem changes in response to changes in climate. Keane and others (1997) provided the most comprehensive treatment of fire and climatic interactions on biogeochemical cycling. They used the Fire-BGC and FARSITE (Finney 1998) models to simulate changes in stand structure, species composition, and water and gas exchange over a 250-year time span in Glacier National Park, Montana. For model comparisons they simulated four fire management scenarios: (1) current existing climate and complete fire exclusion, (2) current existing climate and recent historical fire frequencies, (3) future expected climate and complete fire exclusion, and (4) future expected climate and expected future fire frequencies. Their results indicate that, because fire tends to maintain younger forests and younger forests have lower respiration, the Glacier National Park landscape respires less carbon to the atmosphere with periodic fires, even after taking fire emissions into account (table 8-4). Smoke emissions nearly doubled in the future climate/fire scenario (4), but these fluxes were small relative to those from autotrophic and heterotrophic respiration in unburned forests (scenario 3). Future climate was predicted to result in more frequent and severe fires.

These results are for one ecosystem, and results are likely different for other ecosystems, especially where fire has not played such a strong historic role in vegetation development. However, the prediction of reduced atmospheric flux of greenhouse gasses

Table 8-4—Annual carbon flux (thousand tons C/year) on the McDonald and St. Mary drainages, Glacier National Park landscape averaged across the 250 year simulation period. Table adapted from Keane and others (1997).

Carbon sources[a]	No fires, current climate (1)	Historical fires, current climate (2)	No fires, future climate (3)	Future fires, future climate (4)
Heterotrophic respiration (HR)	820	768	942	810
Autotrophic respiration (AR)	1,168	1,087	1,466	1,128
Total respiration (TR=HR+AR)	1,989	1,855	2,409	1,938
Total fire emissions	0	15	0	24
Total carbon emissions	1,989	1,871	2,409	1,962

[a] Carbon, expressed in units of 1,000 tons/year, can be converted to Gg/year if multiplied by 0.9072. Gg is a gigagram (10^9 grams).

associated with periodic fire illustrates that, because of the complex interactions among ecosystem functions, ecosystem responses may be counterintuitive.

Considerable uncertainty still exists as to how far and how fast climate will change. The autecology of many species is poorly known so it is not possible to make quantitative determinations of how they will respond. Because future climate and vegetation are uncertain, it is not possible to quantify changes in fire potential. Considerable research is needed before we can confidently predict the magnitude of climate change, its effects on vegetation and fire, and feedbacks to the climate system. Given the complexity of the problem, it is unreasonable to expect significantly better information in the near future. Given the magnitude of potential implications to fire management, long-range planning should recognize the need for greater resources in fire management (K. Ryan 1991; Stocks 1993).

Global change is a fundamental fact that natural resource managers must face. The direction and magnitude of climate change over the next few generations are uncertain, particularly at the regional level. But the continued changes in land use are likely to affect fire management regardless of the degree of climate change. Given that weather patterns and atmospheric chemistry are likely to change, and given the introduction of exotic species, management activities based on the goal of restoring the historic range of variation may not succeed (Millar 1997). Active manipulation of wildlands and their disturbance regimes may be necessary to try to maintain the continued presence of numerous species (Peters 1990; Sampson 1997).

Notes

James K. Brown

Chapter 9: Ecological Principles, Shifting Fire Regimes and Management Considerations

This chapter presents a broader, more fundamental view of the ecological principles and shifting fire regimes described in the previous chapters that have important implications for ecosystem management. Also included are strategies and approaches for managing fire in an ecosystem management context and sources of technical knowledge that can assist in this process. Research needs are also described. The ecological fundamentals that underlie the effects of fire on flora and fuels can be described under four broad principles:

1. Fire will occur with irregular pattern depending on climate.
2. Diversity of species and vegetation pattern depends on fire diversity.
3. Fire initiates and influences ecological processes such as regeneration, growth and mortality, decomposition, nutrient fluxes, hydrology, and wildlife activity.
4. Humans exert a commanding influence on ecosystems by igniting and suppressing fire.

Ecological Principles

Fire Recurrence

Fire as a disturbance process on wildlands has occurred as long as vegetation has been present on earth. The history of fire can be traced through charcoal fragments back to the Paleozoic Era, several hundred million years ago (Agee 1993). Lightning that can start fires occurs at a mind boggling rate. Approximately 8 million strikes per day occur globally (Pyne 1982). Human ignitions were common historically and continue to be common today. Wildland fires will continue to happen; the important questions about fire occurrence are when, where, and of what severity?

The frequency of historical fire varied widely across North America depending on climate. Fire return intervals typically ranged from 2 to 5 years in ecosystems supporting abundant cured or dead fine fuels such as the Southern pines, Southwestern ponderosa pine, and oak savanna. They ranged from 5 to 35 years for dry site conifers, shrublands including California chaparral, and most grasslands; 35 to 200 years for mesic site Western and Northern conifers; 200 to 500 years for some Eastern hardwoods and wetter site conifers; and 500 to 1,000 years for extremely cold or wet ecosystems such as alpine tundra and Northwestern coastal spruce-hemlock forests.

Our knowledge of fire frequency is largely based on tree ring analyses and postfire stand ages, which only allow a glimpse of fire history over the past several hundred years—a rather short climatic period. Nonetheless, it provides a basis for understanding the recurrence of fire that can be useful in planning. Keep in mind that climate could indeed change and in turn influence the occurrence of fire and the nature of vegetation response.

Historically, fires have occurred at irregular intervals, largely determined by climate. Dendroclimatological studies in western Canada (Johnson and Larsen 1991) and the United States (Swetnam 1993) have shown that climatic cycles within cycles sometimes influence fire frequency. For example, in giant sequoia forests, precipitation was the most important influence on fire occurrence over periods of years such as the recurrent episodes of the climatic phenomena El Nino and La Nina (Swetnam and Betancourt 1990). However, temperature was the most important influence on fire frequency over periods of decades to centuries. In both cases fuel moisture content was probably the important fuel property most influenced by climatic trends in precipitation and temperature. A study of presettlement fire frequency regimes of the United States (Frost 1998) suggests that patterns of fire recurrence, termed "fire periodicity," can be considered as regular or irregular. For fire regimes having high fire frequencies (average fire-return intervals of 0 to 10 years), individual fire occurrences were considered nonrandom because they clustered around a mean fire frequency. For fire frequencies greater than 10 years, individual fires occurred irregularly or in a random pattern.

Biodiversity

Biodiversity is broadly defined as the variety of life and associated ecological processes that occur in an area. This variety is sometimes broken down into genetic, species, and ecosystem components (Salwasser 1990). In dealing with vegetation, it is convenient to think of the spectrum of components as being plant, community, and landscape. The landscape can be viewed as a mosaic of patches, which are plant communities typically described as vegetation types, successional stages, stands, and age classes.

Fire regime types influence biodiversity in various ways (Duchesne 1994). In forest ecosystems, understory fire regimes have the greatest influence on biodiversity within plant communities because the understory vegetation is more affected by fire than the overstory. Stand-replacement fire regimes substantially influence biodiversity across the landscape by affecting the size, shape, and distribution of patches. Mixed fire regimes probably have the most influence on biodiversity within plant communities, but also affect patch characteristics or between community diversity. In grassland ecosystems, fire frequency and seasonal timing largely determine biodiversity.

Biodiversity can be increased by fire in many ecosystems and reduced by eliminating fire (Keane and others, in press). Variability of fire regimes in time and space creates the most diverse complexes of species. Thus, landscapes having fires with high variability in timing, intensity, pattern, and frequency tend to have the greatest diversity in ecosystem components (Swanson and others 1990). The phrase "pyrodiversity promotes biodiversity" coined by Martin and Sapsis (1992) aptly summarizes this concept. However, biodiversity can be reduced when fires occur much more frequently than happened under the historical fire regime. An understanding of the underlying relationships provides a basis for managing fire to meet conservation of biodiversity goals.

Plant Response to Fire

Chapter 2 explained the many adaptive traits that allow plant species to survive fire. In fact, many species depend on fire to continue their existence. Traits such as thick bark, fire resistant foliage, and adventitious buds allow plants to survive low to moderate intensity fires of relatively short duration. Traits such as fire stimulated germination, belowground sprouting parts, and serotinous cones allow plants to reproduce following high severity fires. For any particular plant to survive and persist, its adaptive traits must be compatible with characteristics of the fire and the timing of its occurrence. Fires can vary in intensity, duration, severity, seasonal timing, and frequency. Other factors, especially weather and animal impacts, can greatly affect whether a species can reproduce and continue its existence following fire. Grazing by ungulates can influence postfire successional patterns and flammability of future fires (Smith 2000).

Fire severity and intensity have a large influence on composition and structure of the initial plant community following fire. Fire intensity mostly influences survival of aboveground vegetation. Fire severity accounts for both upward and downward heat fluxes; thus, it is a better indicator of initial postfire flora and other fire effects. For example, when moisture contents of the forest floor fuels are high, a surface fire may burn at high intensity yet not damage sprouting tissues in the duff layer and mineral soil. Conversely, under low forest floor moisture contents, a surface fire may burn at low to moderate intensity yet consume the forest floor and damage many sprouting tissues. As a general rule, burned areas tend to return to the same flora that was there before fire (Christensen 1985; Lyon and Stickney 1976). However, fires of high severity create opportunities for new plants to establish from offsite seed. Large, high severity burns can be slow to recover depending on available seed sources. Fires of low severity are followed by a strong sprouting response except where annuals are the dominant vegetation.

The timing of fire including both seasonality and frequency is crucial to managing for conservation of biodiversity. This aspect of fire management can be easily overlooked because of emphasis on controlling fire and meeting air quality constraints. Seasonal timing of fire is important because it largely determines fire severity and related mortality. It particularly affects reproduction of herbaceous plants and shrubs. For example, in some ecosystems spring and summer fire may produce abundant postfire flowering while late summer and fall fires may produce little. Perennials in Texas survive spring fire, but annuals are harmed if fire occurs before seed is produced (Chandler and others 1983). Evidence suggests that to maintain long-term (decades) diversity in a tall grass ecosystem, fire should be applied at different times of the year to achieve successful seedling establishment and productivity for a variety of plants (Bragg 1991).

Fire frequency is a particularly important consideration in short fire return-interval regime types because a period of several years to perhaps a decade can be critical for survival of some species. Frequent fire regimes that allow control of shrubs are critical to maintaining grassland ecosystems (Wright and Bailey 1982). Many rare and threatened species have declined with reduction of fire frequency (see Greenlee 1997). Some fire dependent species in the Southeastern United States seem to require a 1 to 3 year fire return-interval (Frost 1995). In contrast, local species extinctions can occur with fires that occur too frequently, although it is generally accepted that locally rare plants have greater chances of surviving on landscapes having diverse vegetation communities and structure created by diverse disturbance histories (Gill and Bradstock 1995). A problem today is that plants adapted to short fire return-intervals can be harmed by fires burning with high intensity and severity in accumulated fuels that resulted from long fire-free periods (Sheppard and Farnsworth 1997).

Community and Landscape Responses to Fire

Species diversity within a vegetation community such as a stand or a patch depends on the collection of species in the community, their adaptive traits, the timing of fire, and the nature of fire as it moves through the community. The spatial arrangement of fuels and individual plants can be important to survival, particularly where fuels are unevenly distributed. Variable fire weather can also influence survival. Concentrations of live or dead fuels can generate much greater fire intensities and severities on relatively small sites. This could enhance or reduce diversity depending on the community. For example, in a Douglas-fir forest, localized fuel concentrations may result in fire-created gaps or holes in the canopy. This would create structural diversity and stimulate understory vegetation, a typical response to fire in a mixed fire regime (fig. 9-1). However, in a ponderosa pine forest, excessive mortality to highly valued old growth trees could be a consequence.

Ecosystems and plant communities are considered to be fire dependent when their continued existence depends on recurrent fire. Where fires occur regularly and frequently, such as in African savannas, open pine communities, and Mediterranean shrublands, they may remain stable for millennia (Chandler and others 1983). Repeated fires in fire-dependent communities maintain a dynamic process that creates diversity across the landscape, but if fire is excluded, biodiversity would probably diminish (Chang 1996). It has been argued that fire-dependent communities have evolved flammable characteristics that help ensure repeated fires and the cycle of renewal (Mutch 1970). However, the evolutionary argument remains unsettled (Chang 1996, Christensen 1993b).

Stand-replacement and to some extent mixed regime fires create patches on the landscape of differing dominant vegetation and stand structures (fig. 9-2). Patches can vary greatly in size and shape depending on the biophysical features of the landscape and fire behavior. Winds of variable speed and direction can cause fire behavior to create a variety of fire shapes. Terrain and landforms, rather than other fire influences, primarily determine patch dynamics in heavily dissected landscapes (Keane and others, in press). For example, fires in the nonmountainous boreal forests were typically large (often well over 10,000 acres) but medium to large (100 to 10,000 acres) in conifer forests of western mountains (Heinselman 1981). Even in large fires in mountainous terrain, fire severity can vary considerably within the burn, leaving a patchy distribution of fire effects (Turner and Romme 1994). Generally, on landscapes characterized by large stand-replacement fires, the pattern is naturally coarse grained. On landscapes supporting smaller stand-replacement fires, the pattern is finer grained. On landscapes having understory fire regimes, occasional trees are killed, creating gaps. This leaves a fine grained pattern in the overstory such that the notion of patches is not as helpful for describing landscape

Figure 9-1—A mixed severity fire burned through this Douglas-fir stand in Yellowstone National Park killing about half of the trees leaving gaps and large openings in the canopy.

Figure 9-2—Stand-replacement fire sustained during low wind speeds by burning in heavy accumulations of dead surface fuels, Yellowstone National Park.

diversity. In these fire regimes, considerable structural diversity can exist within communities.

As time since last fire increases, succession advances all stands to similar communities gradually reducing structural diversity (Keane and others, in press). Extending fire-free periods also increases the likelihood of larger fires, hence larger patches and less patch diversity (Bonnickson and Stone 1982; Heinselman 1981; Swetnam 1993). In whitebark pine forests, Murray (1996) found that lack of fire created high elevation landscapes with high mean patch size and low diversity. Romme (1982) found that fire control policies tended to reduce landscape richness and patchiness and increase evenness in Yellowstone National Park, although in some situations, exclusion of fire actually increased landscape diversity. Knowledge of fire regimes can help managers choose alternative land practices involving fire that favor landscape diversity compatible with natural ecosystems.

Ecological Processes

Fire is an ecological process that triggers an amazing network of other processes and associated conditions. To explain this network, it can be helpful to categorize fire effects into first and second orders. First order effects are the immediate actions of fire and include plant mortality, consumption of organic material, creation of smoke, and changes to the physical-chemical environment. Second order effects are many and depend on the nature of first order effects and the postfire environment, especially soil, weather, and animal activity. For example, here is a partial list of second order effects:

1. Change in microclimate
2. Increase in range of soil temperatures
3. Change in soil nutrients and microbial activity
4. Regeneration of vegetation
5. Succession and new vegetation patterns
6. Change in plant growth rates and competitive interactions
7. Altered wildlife habitat and activity of invertebrates and vertebrates
8. Changed water storage capacity and pattern of runoff

Plant mortality, regeneration, and growth are fire effects of obvious importance to land managers because they determine the characteristics of flora and fuel that are readily observable as succession proceeds. Less apparent but nonetheless important, especially to the pattern of fuel change, is the decomposition process that involves fire, insects, and pathogens in varying roles.

Successional Pathways

The classical concept of succession was based on the perception that plant communities evolved over time toward a final climax state that remained stable indefinitely. However, modern ecologists have rejected this concept and now view succession as a dynamic process that can move in alternative directions under the influence of periodic disturbance and never reach a stable end point (Christensen 1988). A useful method of portraying succession utilizes the multiple pathways approach (Connell and Slayter 1977; Kessell and Fischer 1981) where successional classes or stages are linked along pathways converging to one or several somewhat stable late-successional community types. Successional classes are described by vegetation type and structural stage. The number of succession classes, pathways, and time steps between classes can vary depending upon knowledge and the application. This approach allows fire of varying severity and other disturbances such as grazing and silvicultural cuttings to be incorporated in the conceptualization of successional processes.

Time is a key element in understanding succession (Wright and Heinselman 1973) and explaining it to others. Some plant communities such as mesic and wet site grasslands regain their former composition and structure within only 1 or 2 years after disturbance (fig. 9-3). For other ecosystems, some compositional change may continue to occur long into the future. Forest and shrubland communities vary greatly in the time necessary for recovery to a mature condition. In understory fire regimes, vegetation usually recovers rapidly. Structural changes are small or fine-grained and may not be readily apparent. In stand-replacement fire regimes, a young forest condition may appear within 20 or so years. But it could take several times longer in large severe burns where tree seed sources are limited.

Decomposition

Fire, insects, and pathogens are responsible for the decomposition of dead organic matter and the recycling of nutrients (Olson 1963; Stoszek 1988). Fire directly recycles the carbon of living and dead vegetation. The relative importance of fire and biological decomposition depends on site and climate (Harvey 1994). In cold or dry environments biological decay is limited, which allows accumulation of plant debris. Fire plays a major role in recycling organic matter in these environments. Without fire in these ecosystems, nutrients are tied up in dead woody vegetation. In forests, tree density and understories thicken causing increased competition and moisture stress. In turn,

Figure 9-3—One year after a prescribed fire in a mountain big sagebrush community, this mesic site recovered to domination by perennial grasses and forbs, Caribou National Forest, Idaho.

this increases the likelihood of mortality from insects and diseases leading to increased dead fuels, higher intensity fires, and possibly volatilization of more nutrients. In grassland ecosystems where both fire and grazing are excluded, thatch or dead herbaceous litter accumulates, which depresses herbage yields and the number of plant species (Wright and Bailey 1982). Fire can help control encroaching shrubs and trees; increase herbage yield, utilization of coarse grasses, and availability of forage; and improve habitat for some wildlife species.

Fire both creates and consumes fuel. It increases available fuel by killing shrubs and trees, which leads to falldown of dead material into the surface fuel complex. Moisture contents of dead fuels average much lower than live fuels, which also increases fuel availability. Insects and diseases perform similar roles. They both kill vegetation, which creates available fuel, and decompose organic matter. Fire in some circumstances enhances the opportunity for insect and disease attack. For example, bark beetles may overwhelm fire-injured conifers, and wood rotting organisms may invade fire-scarred deciduous trees. A complex interaction that is not well understood exists between insects and disease organisms, fire, and the environment. However, we do know that fire, insects, and pathogens evolved together as vital components of ecosystems.

Fuel Accumulation

Fuel accumulation is a term often used loosely to indicate an increasing potential for fire to start, spread, and intensify as the time since the last fire increases. Generally, in ecosystems where annual biomass increment exceeds decay, total vegetative biomass increases steadily with time because photosynthesis is an ongoing process. Fuels accumulate but not necessarily in a steady fashion (Brown 1985a). On forested sites much of the annual biomass increment is tied up in live tree boles where it is unavailable for combustion. In grasslands and forests having short fire intervals, fuels increase regularly over time as biomass increases. However, in medium to long fire interval conifer forests, available fuel, and fire potential may decrease as a postfire stand develops, then increase as the stand becomes old and overmature (Brown and See 1981).

Fuel accumulation and associated fire potential depend on fuel quantity as well as other important fuel properties such as compactness and continuity (vertical and horizontal). To be useful for estimating fire behavior, fuel quantity must be expressed by size classes for live and dead components. In a given vegetation type, fuel quantity, size distribution, dead-to-live ratio, and continuity are the important properties that change as succession progresses. Generally, fuel quantities accumulate to greater levels on the more productive sites in grassland, shrubland, and forest ecosystems (Brown and See 1981; Wright and Bailey 1982). In forest ecosystems much of the dead fuel exists as coarse woody debris, which includes pieces larger than 3 inches in diameter and sometimes larger than 1 inch diameter (Harmon and others 1986). The more productive sites grow larger trees, which eventually become coarse woody debris. An important consideration in management of temperate ecosystems is that coarse woody debris be recognized for the many roles it plays. It contributes to biodiversity by being part of the life cycle of macroinvertebrates, soil mites, insects, reptiles, amphibians, birds, and mammals (McMinn and Crossley 1996). It is a source of nutrients, habitat for terrestrial and aquatic life, and fuel for wildfire (Harmon and others 1986). As a fuel its most significant feature is that it becomes rotten wood, which prolongs burnout and allows fire to persist on site for long periods. Historically, large fires occurred because fire remained smoldering in rotten wood and duff for extended periods until low fuel moistures combined with high wind speeds to support intense, fast spreading fires.

Flammability increases as dead-to-live ratios increase. As fuels accumulate through growth and mortality of plants, flammability thresholds may be reached that allow fires to increase greatly in intensity. Surface fires become crown fires in conifer forests, and shrub communities burn intensely as a single fuel complex.

Fuel continuity is important because it partly controls where a fire can go and how fast it travels. In grasslands and open shrublands, heavily grazed areas and areas of low productivity form discontinuous fuels that limit spread of fire, which can be a critical obstacle to use of prescribed fire. In forests, existence of ladder fuels from understory vegetation allows surface fires to reach into the crown canopy. If the canopy is mostly closed, crown fire can readily develop under adequate wind speeds. Open canopies do not support crown fires. Increased fuel continuity can account for changes in fire severity from understory to mixed and from mixed to stand-replacement. Many options are available to land managers for altering fuel continuity through manipulation of vegetation.

Effects of fire on fuel arise basically two ways: first, reducing fuel through consumption, and second, increasing fuel by killing vegetation. Both processes affect several properties of fuel and fire potential. Initially dead surface fuel loadings are reduced, also lowering the dead-to-live ratio. If substantial amounts of shrubs, small conifers, and limbs and foliage of larger conifers are killed by fire but not consumed, they will contribute to surface fuels in the years ahead as they accumulate on the ground. Fire greatly influences fuel continuity by creating vertical and horizontal gaps within and between surface fuels and crown fuels.

Accumulation in Forests—Live and dead fuels, as well as small and large diameter fuels, can follow different patterns of accumulation. Typically, live herbaceous and shrub fuels increase following fire during early stages of stand development. Then as tree canopies close, live herbaceous and shrub fuel quantities tend to decrease on mesic sites (Habeck 1976; Lyon and Stickney 1976). However, a decrease in biomass may not occur where understories contain shade tolerant species. Fine dead fuels from foliage, bark flakes, twigs, and cured herbaceous vegetation become incorporated in the forest floor. Once crown canopies close, the amount of litter fuel remains fairly constant as newly fallen litter is offset by older litter moving into the duff layer. Duff quantities continue to increase for some time until equilibrium with decay is reached. This period varies widely from approximately 5 years in Southeastern United States (McNab and others 1978) to well over a hundred years in some boreal ecosystems.

Dead branches and tree boles accumulate on the ground in response to natural mortality and factors causing downfall (Brown 1975). Mortality factors such as fire, insects, disease, canopy suppression, and wind and snow damage impact stands in a rather haphazard manner. Thus, accumulation of downed dead fuel often occurs in an irregular pattern that is correlated poorly with stand age (Brown and See 1981).

Conifer crown fuels increase regularly; however, likelihood of crown fire may increase then decrease as the lower canopy level grows further above surface fuels. Eventually, crown fire potential increases again when surface fuels increase and understory conifers become ladder fuels. Shade tolerant species tend to have more foliar biomass than intolerant species due to their longer needle retention and higher crown densities (Brown 1978; Keane and others 1999). Because of their shade tolerance they can fill in crown canopy gaps and develop into understory ladder fuels.

Fuels critical to fire spread differ considerably between short and long fire interval fire regime types (Brown 1985a). In short fire interval forests, fine fuels such as grass, live shrubs, and needles create flammable understory fuels even in forests with vastly different decomposition rates such as in longleaf pine and ponderosa pine. The substantial quantity of fine fuels coupled with long periods of suitable burning conditions largely account for the understory fire regime. In long fire interval forests the forest floor and accumulated coarse woody debris are critical fuels. They burn with considerable heat release over a relatively long duration resulting in extensive mortality to overstory trees. They ignite other surface and aerial fuels and serve as excellent receptors of spotting embers that often allow fire to move in a leap frog fashion. Fire intervals and environments differ considerably between long fire interval types such as cedar-hemlock forests on warm moist sites and subalpine and boreal forests on cold, dry sites. Nevertheless, in both cases accumulated forest floor and downed woody fuels support stand-replacement fire particularly during extended dry periods (Romme and Despain 1989).

Accumulation in Shrublands and Grasslands—On many grasslands, grazing eliminates most of the annual production so fuel accumulation is inconsequential. In the absence of grazing, fuel quantities depend primarily on annual production, which varies substantially by site potential and annual precipitation (Wright and Bailey 1982). Fuel loading may increase for several years after a fire as some slow responding grassland communities recover. Frequently, however, productivity is increased within 1 or 2 years following fire (Wright and Bailey 1982). Herbaceous litter accumulates in some grassland ecosystems but only marginally in others. Ratios of accumulated litter-to-current production typically range from 0.25 to 0.50 (Reinhardt and others 1997).

In shrub and shrub/grass ecosystems young communities generally have a low dead-to-live ratio. Flammability depends largely on grass and sedge fuels. As shrubs become senescent or undergo mortality, dead stemwood accumulates, which significantly increases potential flammability. Dead fuel quantities tend to increase with time since last fire or with age of plant community as suggested for chaparral, however, not in a uniform nor readily predictable fashion (Paysen and Cohen 1996). Besides age, other factors such as drought, winter kill, insects, and disease can cause periodic dieback that creates substantial dead fuel quantities. As cover and height of shrubs such as sagebrush increase, fire intensity and rate of spread potential increase markedly (Brown 1982).

Human Influences

People are part of ecosystems and certainly have exerted a major, far reaching influence on fire across the landscape. Indian burning was common throughout the United States and Canada. Pyne (1982) quotes Henry Lewis as saying, “To simply note that all Indians used fire to modify their environments is no more an ecological generalization than to note that all farmers used plows.” The extent of Indian burning varied considerably, however, depending on locale and population movements (Boyd 1999; Pyne 1982). Indian burning greatly extended grasslands especially in the Eastern and Midwestern United States. Most of the coastal plain from Massachusetts to Florida to Texas was savanna. Western valleys and foothills were maintained as grasslands and open forests (Gruell 1985).

Considerable debate exists about the relative importance of Native Americans and lightning in maintaining historical fire regimes (Barrett and Arno 1982; Frost 1998; Keane and others 1999). The relative importance of Native American fires was probably greater in topographically complex areas where fire compartments were smaller and where lighting ignitions were infrequent (Frost 1998). Also debated is whether anthropogenic burning should be considered part of the native or natural fire regime (Arno 1985; Kilgore 1985). Fires set by Indians were often of different seasonality, frequency, and landscape pattern than those started by lightning (Frost 1998; Kay 1995). Indian and lightning-caused fire existed for thousands of years, a short evolutionary period but a long time for plant communities to adjust to fire disturbance. This long period of fire on the landscape argues strongly for accepting both sources of ignition in considerations of Euro-American presettlement fire history used to guide management of ecosystems.

Efforts to suppress fires were modest at first relying on wet blankets and buckets around dwellings and campsites (Pyne 1982). Modern suppression capabilities relying on sophisticated communications, rapid attack, specialized equipment, and many fire fighters are a far cry from the early 1900s. Fire protection has succeeded in reducing the extent of fire and increasing fire intervals. Chandler and others (1983) suggested that as protection succeeds, fire intervals become greater and flammability increases. Then, more protection is needed to keep burned acreage down. A given protection effort and annual burned area will eventually reach equilibrium. Since the 1980s, the costs of protection and greater understanding of the role of fire have led to more hazard reduction and ecosystem maintenance rather than just protection.

For the past 100 years or so, human use of fire—earlier termed controlled burning and now prescribed fire and wildland fire use—has met with considerable controversy politically and within land management organizations. “Light burning” (understory fire) was once widely applied in the southern pines and ponderosa pine type especially in California. However, the perceived threat to effective organized fire control largely curtailed the program on publicly owned lands (Pyne 1982). Some benefits of controlled burning were still recognized, especially hazard reduction and preparation of seed beds for regeneration. In the West justification for prescribed fire was fuel reduction, namely slash burning. This single purpose use of prescribed fire resulted in short-term successes but long-term failure to optimize societal objectives for forests (Agee 1993).

More recently, the concept of ecosystem management has led to a much wider understanding of the ecological role of fire and its importance in the functioning of ecosystems. Concerns over air quality, control of fire, and costs, however, remain as major constraints on the application of prescribed fire and wildland fire use. The responsibility to see that fire is properly managed as a component of the ecosystem is now greater than ever because land managers have the power to delay and exclude fire as well as an understanding of fire's important ecological role.

Shifting Fire Regimes

Chapters 3 through 7 clearly show that fire regimes have shifted from what they were historically across most of the United States and southern Canada. In a comprehensive assessment of burning in the contiguous United States, Leenhouts (1998) estimated that approximately 10 times more area must be burned than at present to restore historical fire regimes to nonurban and nonagricultural lands. The greatest departure from historical fire regimes is in the Rocky Mountains where only a small fraction of the pre-1900 annual average fire acreage is being burned today (Barrett and others 1997). Kilgore and Heinselman (1990) estimated that the greatest detrimental effects of fire exclusion were in short interval fire regimes of the Rocky Mountains. In contrast, in long fire regimes, the effects of fire protection have not had a significant influence. In the Canadian and Alaskan boreal forest limited protection due to remoteness has maintained fire regimes essentially as they were historically.

Extensive grazing by domestic stock that reduces fuels, and fragmentation by agriculture and human developments, have also contributed to shifting fire regimes. Lengthened fire return intervals have resulted in changes of minor to major consequence to vegetation and fuels by increasing wildfire severity and decreasing species and structural diversity. A comparison of historical and current fire regimes in the Interior Columbia River Basin of about 200 million acres showed that fires have become more severe on 24 percent of the area (Morgan and others 1998) (see fig. 5-1 in chapter 5 of this volume). Fire severity was unchanged on 61 percent of the area. Fires were less frequent on 57 percent of the area, unchanged on 33 percent, and more frequent on 10 percent of the land area. Fire protection, reduced fine fuels from grazing, decreased fuel continuity from human development, and in some cases exotic plants are the most probable causes (Chang 1996; Keane and others 1999). Further analyses of changes in fire regimes and condition classes of vegetation are currently under way for the United States (Hardy 1999).

Forests and Woodlands

Changes in forest composition and structure due to shifting fire regimes have been widely documented. Generally, shade-intolerant species are being replaced with shade-tolerant species. Stand densities are increasing with development of multiple layer canopies. Outbreaks of insects and occurrence of root diseases appear to be worsening (Stewart 1988). The greatest impacts have occurred in the understory fire regime types typified by ponderosa pine and longleaf pine ecosystems (fig. 9-4). Although these two ecosystems experience widely different climates, they share the same end results of fire exclusion made worse in some locations by selective harvesting of old growth trees. Where fire regimes have shifted, growth and vigor of trees is reduced, insect and disease mortality is increased, and understory fuel loadings and continuity increased so that wildfires tend to be of high intensity, killing most or all of the overstory pine. Diversity of understory herbs and shrubs is decreased. The loss or depletion of the pyrophytic herb layer is considered to be one of the unrecognized ecological catastrophes of landscape history (Frost 1998). The extent of the problem is greater in ponderosa pine where relatively little prescribed fire has been applied. Although prescribed fire is widely applied in the South it has largely been used only for rough (accumulated understory fuels) reduction during the dormant season. Thus, lack of seasonal fire diversity in the southern pine types has limited plant diversity.

In mixed fire regime types such as coastal and inland Douglas-fir, whitebark pine, red pine, and pinyon-juniper, the results of fire exclusion have created the same problems as found in understory fire regimes. Mixed fire regimes are experiencing considerably less nonlethal understory fire than in the past (Brown and others 1994). The mixed fire regime is shifting toward a stand-replacement fire regime that favors more shade tolerant species and less landscape diversity.

In stand-replacement fire regimes, fire intervals have generally lengthened; however, the effects of this vary widely depending largely on presettlement fire return intervals and accessibility for fire suppression efforts. For example, in the lodgepole pine/subalpine fir type, which dominates the Selway-Bitterroot Wilderness, presettlement stand-replacement fire was 1.5 times more prevalent than during the recent period (Brown and others 1994). The presettlement fire return-interval was approximately 100 years. In the same type in Yellowstone National Park, characterized by a fire return-interval of about 300 years, the area burned probably has not differed between presettlement and recent periods (Romme and Despain 1989).

The age distribution of marginally commercial and noncommercial forests such as those in wilderness areas and parks is shifting to an abundance of older stands (Brown and Arno 1991). Succession is increasing the shade tolerant component of stands, making a major species shift likely if fire continues to be

Figure 9-4—A stand-replacement fire supported by accumulated dead surface fuels and live ladder fuels from dense understory trees occurred in this understory fire regime type killing the old growth ponderosa pine, Yosemite National Park.

excluded. In the case of western aspen more than half of the type has been lost (Bartos 1998), much of it due to successional replacement by conifers (Bartos and others 1983). Fire protection policies have resulted in the fire cycle in aspen shifting from about 100 years to 11,000 years; thus, if this degree of fire exclusion continues, the loss in biodiversity will be considerable. In jack pine forests the more shade tolerant balsam fir is gradually assuming dominance aided by natural deterioration and harvesting of jack pine.

Fuel accumulation patterns vary widely in coniferous stand-replacement fire regime types. Mature forests may support abundant or relatively little available fuel. However, as fire intervals are allowed to increase and stands become over mature, downed dead woody fuels and live ladder fuels from shade tolerant understory conifers can be expected to dramatically increase. The result will still be stand-replacement fire but at higher intensities, which will tend to propagate larger fires in spite of suppression efforts. This trend could lead to fewer but larger fires burning during severe fire weather years, causing less diversity in patch size and age (Keane and others 1999).

Grasslands and Shrublands

Grassland fire regimes have shifted dramatically from the presettlement period. Many ecologists consider the reduced frequency and extent of fires on rangelands due to fire protection to be among the most pervasive influences in the United States by non-Native Americans (Pieper 1994). The shift to woody plant domination has been substantial during the past hundred years. Grazing and possibly climate changes have acted with reduced fire to give a competitive advantage to woody plant species. Some woody plants such as honey mesquite become resistant to fire, develop fuel discontinuities, and reduce spread of fire. In time, recovery following fire favors shrubs over perennials (Archer 1994). This can alter the composition of ecosystems to the point that a return to the grassland type becomes nearly impossible or impractical (Brown 1995).

Historically, fires were more frequent in Eastern than in Western grasslands. High productivity of biomass was maintained in the tallgrass prairie by frequently occurring fire that recycled accumulated thatch. A diverse composition was probably favored by variable frequency and seasonality of fires (Abrams and Gibson 1991; Bragg 1991). Western grasslands appear to have generally experienced fire less frequently (Gruell 1985; Wright and Bailey 1982) but still frequently enough to hold back invasion of woody plants.

Fire regimes have shifted to too much fire in the drier portions of the sagebrush-steppe ecosystem that occupies over 100 million acres in Western United States. Fire frequency has increased in many areas due to invasion of cheatgrass and medusahead, introduced annuals that cure early and remain flammable during a long fire season. Increased fire frequency exerts strong selective pressure against many native plants (Keane and others 1999). A contrasting situation exists for the more mesic mountain big sagebrush type where decreased fire frequency and encroachment by conifers is causing a reduction in herbaceous and shrub vegetation (fig. 9-5).

Managing Fire

Fire is an integral component of ecosystems that can affect all aspects of ecosystem management. Fire regimes have shifted as a result of human influences and may continue to shift with clearly detrimental results in some ecosystems. Land managers need to know how to plan and carry out fire management strategies that successfully incorporate the ecological role of fire. Constraints on managing prescribed fire and smoke make it difficult to achieve resource goals, while protection against wildland fires allows development of undesirable ecological consequences (Brown and Arno 1991). Overcoming this predicament requires that land managers and the public alike recognize the role of fire in the functioning of ecosystems and in meeting varied resource objectives.

Strategies and Approaches

Vegetation and fire management objectives should be derived from broader ecosystem management goals to achieve desirable fire effects. Determining objectives, and the strategies and approaches for achieving them, can be simple to complex depending on land ownership and direction provided by the owners. For example, a small woodlot owner may simply want to reduce fire hazard, in which case fuel reduction objectives can be clearly stated and, if appropriate, a prescribed fire conducted to reduce the unwanted fuel. Where the direction is ecosystem management, a goal recently adopted on many Federal and some State lands (Salwasser 1994), a more elaborate process may be required to determine objectives and strategies.

Figure 9-5—Without disturbance, this sagebrush/grass community being encroached by Douglas-fir will eventually become a closed canopy forest with sparse understory vegetation, Deerlodge National Forest, Montana.

To steer this process, a guiding principle or goal for ecosystem management is to provide for conservation of biodiversity and sustainability of ecosystem composition, structure, and processes (Kaufmann and others 1994). This involves molding a management plan based on an understanding of ecosystem processes. An element missing or minimally considered from many past planning efforts was the landscape of varying scales. For this a perspective is needed that involves consideration of ecological processes across a hierarchy of land units (Hann and others 1993).

The setting of goals and objectives starts out broadly with a goal specifying the future condition of the ecosystem or a particular tract of land. This desired future condition is a vision for the future and not an objective for management action (Kaufmann and others 1994). An assessment of the ecosystem, resource potentials, and needs of people is a prerequisite for setting the desired future condition. From this, more specific objectives can be derived for managing fire. They should be specified in terms that can be monitored. Different approaches may be appropriate for doing an assessment and setting the desired future condition and the ensuing management objectives.

Consider the planning task by three types of land use zones (Arno and Brown 1989):

- **Zone I – wilderness and natural areas** objectives call for allowing fire to play its natural role to the greatest extent possible. Fire objectives may vary depending on whether it is a wilderness or natural area intended to preserve a particular condition or process.
- **Zone II – general forest and range management**, where the need to provide resource values means a wide range of vegetation and fire objectives will be appropriate.
- **Zone III – residential wildlands**, where the natural role of fire will be constrained considerably and fuel management is the primary objective.

Two occasionally troublesome facets of setting goals and objectives in Zones I and II that rely on knowledge about the ecological role of fire involve the "historical range of variability" and the goal orientation of "process versus structure."

Historical Range of Variability

The historical range of variability (also called natural range of variability) in ecosystem components can be used to help set desired future conditions and fire management objectives. It can serve as a basis for designing disturbance prescriptions at varying spatial scales and help establish reference points for evaluating ecosystem management (Morgan and others 1994). Reference points to past functioning of ecosystems can be interpreted from various sources such as historical records, palynology, natural areas, archival literature and photographs, GIS data layers, and predictive models (Kaufmann and others 1994; Morgan and others 1994). Historical fire regimes of forest ecosystems are often characterized by determining age distribution and areal extent of seral classes across a large landscape and dating fire scars to determine fire return intervals. These techniques provide a snapshot of ecosystem conditions that covers the past 100 to 400 years. Pollen analysis can extend this period but with less precision about disturbance events (Swanson and others 1993). Estimation of historic fire frequencies in grasslands and shrublands is more problematical because of a lack of fire scars and easily determined age classes. It relies largely on historical accounts of human activities.

To what extent should knowledge of the historical range in variability be relied upon to help establish goals and objectives? This depends largely on soundness of the ecological knowledge and other ecosystem issues such as human needs and threatened and endangered species (Myers 1997). A strong argument can be made that knowledge of historical fire should be used as a guide for understanding landscape patterns, conditions, and dynamics, but not necessarily for creating historical landscapes. Knowledge of historical variability provides a basis for bringing the range of existing conditions in a landscape within the historical range (Swanson and others 1993).

A scientifically based rationale underlies the use of historical variability as a guide for managing biodiversity. Native species evolved and adapted to natural disturbance events over at least the past 10,000 years. Numerous ecological studies emphasize the close dependence of species on disturbance regimes (Swanson and others 1993). Genetic diversity (Frankel and Soule 1981) as well as landscape diversity are maintained through disturbance regimes. Where fire regimes have shifted markedly, species and landscape diversity have declined.

Concerns and limitations to using historical variability as a guide to managing ecosystems (Morgan and others 1994; Swanson and others 1993) are:

1. Difficulty interpreting past variability due to insufficient data.
2. Degree to which past and future environmental conditions may fall outside the established range of historical conditions. For example, the possibility of future climate change due to global warming is a significant concern.
3. Extent to which the range of ecosystem conditions desired by society differs from historical variability.

The natural range of variability can be determined and applied with reasonable confidence in high frequency fire regimes of forests. In understory fire regimes, considerable data on fire frequency often can be obtained by consulting published accounts or conducting studies of fire intervals on fire scarred trees. Variability of fire-return intervals can be quantified and compared with recent fire history to determine whether a significant departure has occurred (Brown 1993). In long interval stand-replacement fire regimes of some forests and tundra, estimates of the historical range of variability are more difficult to establish with certainty because of the limited number of disturbance events that can be studied. Perhaps the best technique for measuring fire regime characteristics in this situation utilizes satellite and GIS technologies to map vegetation pattern (Morgan and others 1994), an approach requiring considerable resources.

A question that often arises in interpreting fire history especially concerning wilderness and other natural areas is how Indian ignitions should be treated (see Lotan and others 1985). The prevailing thought seems to be that because Indian burning occurred over a long period, ecosystems were adjusted to fire effects from human and lightning ignitions combined and this reflects historical fire regimes. Disturbance history can only be readily and reliably measured for the past 200 to 400 years. Variability in climate, vegetation composition, and disturbance patterns has been substantially greater over the past several thousand years than over just the last 400 years. But land managers need a consistent basis on which to plan, and using measurable fire history is a practical approach. The concept of the historical range of variability can be valuable in understanding and illustrating the dynamic nature of ecosystems and in evaluating current ecosystem health.

Process Versus Structure Goals

Process and structural goal setting approaches are important to management of Zone I lands. These concepts originated with establishment of wildernesses and natural areas where the goal was to manage for naturalness. The proper role of fire in wilderness and natural areas has been characterized in terms of process-oriented and structure-oriented goals (Agee and Huff 1986). Expressed simply, do we want a natural fire regime (process) or rather the vegetation that a natural regime would have created (structure) (Van Wagner 1985)? The answer to this may always involve some degree of debate because of philosophical differences over the concept of natural (Kilgore 1985). In practice, both approaches or a mixture of the two may be appropriate depending on circumstances. Practical aspects such as costs, fire safety considerations, and size and boundaries of the ecosystem will often determine the most appropriate approach.

A strictly process-oriented goal is probably only appropriate in large wilderness areas. The process goal approach modified by practical considerations will usually be necessary.

In understory fire regimes where surface fuels have accumulated to the point that high intensity fire is likely, a structure-oriented goal is the best approach to ultimately achieve natural conditions. After fuels have been reduced using a prescription for low severity fire to avoid killing the overstory, a process goal of allowing natural ignitions can be followed if it will maintain the understory fire regime (Bonnickson and Stone 1985). Structural goals will continue to find application in understory fire regime types to restore and even maintain the natural role of fire. The structural goal approach is probably the best for management of threatened and endangered species. It may also be more efficient and esthetically pleasing (Agee and Huff 1986).

Mixed fire regime types in wilderness areas present variable, complex landscape patterns that can make structural goals difficult to achieve. Fire frequencies in the mixed type typically range from 35 to 100 years. In some localities fire has been absent long enough that fuels and stand structures appear to be falling outside the range of historical variability (Arno and others 2000). In such cases, where accumulated surface fuels and naturally occurring ignitions would favor stand-replacement fire, structural goals aimed at retaining a portion of the overstory may be appropriate to restore the mixed fire regime. If excessive fuels have not accumulated, process goals seem to be the most reasonable.

Another consideration in wilderness areas, regardless of whether structural or process goals are chosen, is when and where to use prescribed fire to meet wilderness objectives. In the contiguous United States 75 percent of Congressionally classified Wilderness areas, which occupy half of the classified wilderness land area, are too small to maintain natural fire regimes by relying strictly on natural ignitions (Brown 1993). Constraints such as concern over escape fire, lack of lightning-caused fires, conflicting wilderness goals, and air quality regulations will require prescribed fire to restore fire and mimic natural processes. Decisions to use prescribed fire must be ecologically based, but also with the realization that exacting solutions to mimicking natural fire processes are probably not feasible. Neither the determination of fire history nor applications of prescribed fire are precise undertakings.

For residential and commercially zoned lands (Zones I and II), structural goals are the most appropriate. Clearly definable and measurable end points are being sought. For example, specific conditions such as tree species and size, stand age distribution, patch size, stimulation of shrubs, increased forage production, and reduced fuel quantities may be desirable objectives.

Landscape Assessment

Managing biodiversity and for sustainability of ecosystem components and processes requires a landscape perspective. Small ecosystems are found within larger ecosystems, individuals occur within communities, and short-term processes are nested within longer term processes (Kaufmann and others 1994). The various scales fit into a hierarchical structure that determines patterns of diversity for an area (Bourgeron and Jensen 1993). A major challenge to setting vegetation and fire objectives in the context of ecosystem management is evaluating and interpreting the ecological significance of multiple scales. Vegetation scales range from individual plants, communities, seral stages, potential vegetation types, to the biome level.

Species and individual plant communities are dealt with using a fine filter approach. Traditionally, assessments of fire effects and other environmental impacts have been done on a project basis using fine and mid scale evaluations. The coarse scale aspects of ecosystems have been largely neglected. The coarse filter approach, which deals with higher scale levels such as aggregations of communities, can operate with relatively little information, yet be an efficient way to meet biodiversity goals (Bourgeron and Jensen 1993; Hunter 1990; Kaufmann and others 1994). A single ecosystem can be too small to hold viable populations of all its species, especially large predators. Thus, the coarse filter approach is best used on assemblages of ecosystems such as watersheds and mountain ranges. Both approaches are necessary to evaluate all facets of an ecosystem and meet the goals of ecosystem management (Hann and others 1993a).

Assessment of landscape and ecosystem properties can be undertaken with varying degrees of sophistication and effort. Some of these planning efforts, which are evolving through trial and error, are mentioned as examples. During the past decade agencies such as the U.S. Forest Service and Bureau of Land Management have undertaken landscape analyses on extremely large areas such as the 200 million acre Upper Columbia River Basin (Keane and others 1996) and smaller areas such as the Pike and San Isabel National Forests and Cimarron and Comanche National Grasslands in Colorado (U.S. Forest Service 1997) and the 130,000 acre Elkhorn Mountains and 46,000 acre North Flint Creek Range in Montana (O'Hara and others 1993). Details of these landscape evaluations varied but they followed three general steps (Hann and others 1993b):

1. Characterize the general composition, structure, and processes of the ecosystems and landscapes within the designated analysis area.
2. Analyze data to assess changes in structure and composition and relate the changes to previous management treatments.
3. Examine the ecosystem processes important for the area and their effects on ecosystem and landscape composition, structure, and rate of change.

Succession Modeling—Simulation of succession provides a means of predicting the long-term interaction of processes such as fire, insects, disease, and cutting of vegetation on landscapes of varying scale. Simulation can be helpful to managers and the public by helping them understand how ecosystems function and for evaluating different management alternatives. The wider availability of powerful computer capabilities has led to an increase in succession modeling efforts particularly for landscape applications. Manager-oriented computer models that simulate successional processes across large landscapes are faced with a tradeoff between realistic portrayal of ecological processes and utility of the model. Some models are too complicated to use without special training or assistance. Nonetheless, managers are increasingly using succession models in their planning while models are continually evolving and computer capabilities growing.

In choosing a model for a particular application, it is important for the temporal and spatial scales of the model to match the intended use (Reinhardt and others, in press). Models that operate over a period of decades are useful for scheduling treatments. For example, the Fire and Fuels Extension to Forest Vegetation Simulator (Beukema and others 1997) simulates fuel quantities, tree characteristics, and tree mortality in the event of a fire for single stands. Managers can use the model to help schedule thinnings and fuel treatments when potential fire behavior and fire effects on an area are deemed unacceptable (Reinhardt and others, in press).

Models that simulate fire effects over centuries are useful for providing targets for managers, for estimating the historic range of conditions, for evaluating implications of climate change, and for understanding possible long-term consequences of management actions. For example, CRBSUM was used to simulate landscape changes for different management scenarios in the Columbia River Basin (Keane and others 1996). Some of the current models that have been applied to assist land managers are summarized in appendix B.

Restoration of Fire

Restoration of fire is needed to varying extents in most ecosystems of North America to meet the holistic goals of ecosystem management. The need for restoration is most evident in high fire frequency regimes such as understory fire regime types and some grasslands and shrublands where fire has been excluded for several times longer than the average fire return interval. Although considerable knowledge supports the need for restoration of fire into wildland ecosystems, constraints and obstacles confront land managers (Brown and Arno 1991; Mutch 1994). Limited funding, air quality restrictions, concerns over escape fire, and inadequate public support can pose difficulties. Some breakthroughs in managing emissions and obtaining support have provided more latitude for prescribed fire programs (Mutch and Cook 1996).

Successful restoration involves clearly stated objectives, plans based on scientific knowledge of fire's role in the ecosystem, and adaptive learning from prescribed fire efforts. Adaptive learning is important because prescribed burning usually improves with experience. Prescription conditions and firing techniques may need to be modified to achieve objectives such as a given level of fuel reduction or to meet constraints such as holding overstory mortality to certain limits. Fire may not spread adequately under an initial prescription, thus requiring lower fuel moisture contents or higher wind speeds to be successful.

Restoration of fire can be undertaken on an entire ecosystem or on an individual plant community basis. Ideally, restoration of individual plant communities would be based on ecological considerations of the broader ecosystem of which they are a part. The extent of ecosystem assessment that is appropriate for planning restoration will depend largely on land ownership and direction given to management. For large land ownerships, restoration of entire ecosystems or large landscape areas is the soundest approach to manage landscape pattern and meet biodiversity goals. It also allows for effective placement of fuel treatments designed to disrupt fuel continuity and reduce threat of large fire occurrences. The steps undertaken by Keane and Arno (1996) to restore fire in the whitebark pine ecosystem may be useful in other situations including grasslands and shrublands. They recommend first, an inventory of landscape and stand characteristics at multiple scales; then, writing descriptions of the important processes of the landscape and stands. Landscapes and stands can then be prioritized for restoration treatment and selected based on inventory, description, priority, and feasibility. Treatments should be designed for each selected stand or landscape based on inventory and description information and implemented as efficiently as possible. Finally, treatments should be monitored to evaluate restoration success.

Restoration of fire in grasslands, shrub steppe, and savannas requires careful consideration of seasonal timing and frequency to assure that prescribed fires will spread at appropriate severities. Once woody plants have encroached to a point of dominating a site, it becomes difficult to get fire to spread with sufficient heat to kill aboveground stems such as oak in savannas (Huffman and Blanchard 1991) and juniper in sagebrush/grass communities. Perhaps the greatest obstacle to success lies with areas that have successionally lost the native mix of species and lack sufficient grass fuel to carry fire. Seeding of native species following fire may be necessary to restore a resemblance of former plant composition. Where conifers invade grasslands such as pinyon-juniper and inland Douglas-fir (Gruell and others 1986), successful spread of surface fire may require fuel enhancement work such as cutting numerous trees to create adequate surface fuels. Otherwise, crown fire may be required, which will necessitate a more flammable, narrow fire prescription that can limit burning opportunities.

Prescribed Fire and Silviculture

Prescribed fire and silviculture can go hand in hand for restoration of forest stands and ecosystems. Some consider prescribed fire to be a silvicultural technique even though it goes far beyond the usual goals of silviculture that are oriented to producing tree products and desirable forest stand structures. One debatable point is the extent to which it is desirable to have management mimic the kinds of stands and landscape structures that typified presettlement fire regimes. However, an understanding of similarities between characteristics of fire regime types and silvicultural stand structures can be helpful for integrating fire with silviculture to restore fire as a process and meet ecosystem management goals. The following description of stand structure and silvicultural practices based on a discussion by Weatherspoon (1996) applies to individual stands. Stands can be treated differently to manage landscape-level vegetation.

Even-Aged Stands—These stands originated naturally mostly from high-severity, stand-replacement fires that killed most of the trees. Silvicultural methods that produce even-aged stands include clear-cutting, seed tree, and shelterwood cutting. Shelterwood or seed trees are typically removed after regeneration is secured. Pile burning or broadcast burning is commonly used to reduce fuels and prepare sites for regeneration. Leaving snags, large downed woody material, and untreated patches in larger treatment units is important for meeting biodiversity goals.

Two-Storied Stands—These stands were associated with moderate to high severity fire typical of the mixed fire regime type. Retention shelterwood (also called irregular shelterwood or shelterwood without removal) is the silvicultural method for treating the stand. Prescribed underburning can often be practiced to manage fuels and create within-stand diversity. Once created, the stand would never be devoid of large trees because each regeneration cutting would be accompanied by retention of some overstory trees. Snags could be readily created.

Uneven-Aged Stands with Even-Aged or Even-Sized Groups—These were associated with low to moderate severity fires associated with the understory fire regime type and perhaps to some extent with the low severity end of the mixed fire regime type. Silviculturally this stand structure is mimicked with the group selection cutting method. Skillful prescribed underburning is required to apply the proper severity for maintaining this structure. Jackpot burning and two-stage burning under different prescription conditions may be appropriate.

Uneven-Aged Stands with Fine Tree Mosaic—These stands are characterized by three or more sizes and ages of all tree species distributed rather uniformly throughout the stand. This stand type is thought to have developed primarily with shade-tolerant conifers over long periods following stand-replacement fire. It is incompatible with frequent fires. The individual tree selection method is used to maintain this structure. This stand structure could be considered to represent open stands of ponderosa pine and longleaf pine. Ecologically, however, they fit better with the previous category of even-aged groups.

Understory Fire Regime Type

Restoration of the understory fire regime type requires application of frequent, low intensity fire, which has been excluded for excessive periods of time. Restoration approaches can vary considerably depending on stand and fuel conditions. The objective generally is to create more open stand structures consistent with historical disturbance regimes. A wide range of stand densities can be appropriate depending on site potential and silvicultural objectives. Various even-aged and uneven-aged stand structures can be utilized. Favoring the long needle pine component through regeneration and retention of old growth trees is frequently a high priority need. Often the major problem to overcome is excessive understory fuel accumulations particularly live ladder fuels, and buildup of duff around the base of desirable leave trees. Another consideration is burning to encourage the historical understory vegetation diversity. This requires burning during the growing season, which is a departure from the traditional application of prescribed fire during the spring, fall, or winter dormant seasons.

Conducting the first prescribed fire after a prolonged period of no fire must be done cautiously to avoid flare-ups in sapling thickets or rough that might kill desirable trees. For ponderosa pine, thinning of dense understories and piling and burning slash before conducting a prescribed underburn may be necessary to reduce flammability and remove competitor species that might survive most prescribed fires (Fiedler and others 1996). However, too much caution where the understory consists of thick patches of fir will result in inadequate fire. Some fuel augmentation by cutting small fir can help carry the fire with adequate intensity to kill the fir. A series of prescribed fires aimed at gradually reducing the accumulated live and dead fuels may be necessary to return stands to where maintenance underburning is easily manageable (Sackett and others 1996). The best approach to restoration must be determined on a case by base basis, but it will usually require a combination of mechanical treatments and prescribed fire repeated over a period of years.

Mixed and Stand-Replacement Regimes

The mixed fire regime includes a wide range of stand structures and landscape patterns that result from highly variable fire severities. Individual fires may be of either nonlethal understory or stand-replacement severity, or a combination of both severities. Thus, managers have considerable latitude in designing prescribed fire and silvicultural activities (fig. 9-6). Although little guidance based on past restoration efforts exists, the best way to determine restoration objectives is on a large landscape basis because of the wide latitude in individual stand structures. The challenge is to provide a diversity of stand structures with retention of snags and some coarse woody debris in forest ecosystems and unburned patches in grasslands and shrublands. In wilderness and natural area management where fires have not been previously allowed, avoiding excessive stand-replacement due to accumulated fuels may be important.

Stand-replacement fire severities can be created from either severe surface fire or crown fire. Wildfires over prolonged burning periods can leave large proportions of both severities as observed in lodgepole pine (Brown and others 1994). High severity surface fires may be more readily prescribed and achieved than crown fires due to the higher risk and fewer burning opportunities for prescribed crown fires. Ecological effects of severe surface fire and crown fire differ. Crown fire consumes foliage that otherwise would fall and protect the soil. It can kill seeds in cones, redistribute nutrients in ash, and provide more chance for regeneration by offsite colonizers. Where silvicultural objectives are being pursued, an important consideration is avoidance of excessive fragmentation caused by intensive small-scale cutting and prescribed fire activities. Provision for snags and coarse woody debris is also important.

Grazing and Exotic Plants

Introduced exotic species and grazing are two major problems that can seriously interfere with efforts to restore fire as an ecosystem process. Well-intentioned

Figure 9-6—Aspen is being successionally replaced by fir, Bridger-Teton National Forest, Wyoming. Restoration will require a stand-replacement disturbance, which could be facilitated by cutting some of the conifers.

prescribed fire, and silvicultural and rangeland enhancement activities, can fail drastically unless grazing and exotic plants are anticipated and managed properly.

Grazing—Excessive grazing can be the biggest hindrance to successful use of prescribed fire where grass vegetation is a major component, particularly in western grasslands and shrub/grass vegetation types (Wright and Bailey 1982). It is more of a problem for bunchgrasses than rhizomatous grasses (Mack and Thompson 1982). Overgrazing in the absence of fire as well as following fire can reduce plant diversity. Grazing too soon following fire can eliminate or greatly reduce desirable vegetation. In grassland areas woody plants are competitively favored, which could defeat the purpose of burning to halt woody plant encroachment.

Depending on site potential and grazing pressure, grazing should be deferred 1 to 2 years following fire in ecosystems such as sagebrush/grass and semidesert shrub (Wright and Bailey 1982). In forests such as the aspen type, intensive grazing of sprouting plants by livestock and wild ungulates, especially elk, following prescribed fires can greatly retard plant recovery. Small prescribed burns are particularly vulnerable to overutilization because of concentrated grazing (Bartos and others 1991).

Grazing prior to a prescribed burn can easily reduce fine fuels to a point where fire will not spread successfully nor have sufficient heat to ignite or kill woody plants. At least 600 lb/acre of herbaceous fuel is needed for successful prescribed fire in grassland and grass/shrub vegetation (Wright and Bailey 1982).

Exotic Plants—Fire can create favorable sites for nonindigenous plant species to become established and flourish. If exotic plants already grow in or near areas that are candidates for prescribed fire, a potential problem exists. Aggressive exotic species can competitively exclude native vegetation. Severe fires that expose large areas of mineral soil are most apt to be invaded by exotic plants; if exotics are already established, their dominance may be accelerated. Lower severity burns are more resistant to proliferation of exotics because many native species sprout and quickly occupy the site.

Cheatgrass, a nonindigenous annual that dominates millions of acres, is an extreme example of a species favored by fire. Its invasion of the sagebrush-steppe vegetation type has led to increased frequency of wildfire due to abundant, early curing fine fuels. The result is permanent conversion to annual grassland and disruption of the historic fire regime (Whisenant 1990). Another problem with nonindigenous plants can occur from seeding nonnative grasses such as annual ryegrass on severely burned sites as part of wildfire rehabilitation efforts. This practice, which is intended to stabilize soils, can delay reestablishment of native species and possibly alter long-term community composition (Conard and others 1991).

A far different problem is caused by exotics such as Chinese tallow, which has invaded coastal marshes of the Southeast. Its invasion causes a shift from grass-dominated communities to a sparse forb-dominated community that is much less flammable and acts as a fire break. Consequently, once Chinese tallow gains dominance on a site, prescribed fire cannot be effectively used to control the exotic and encroaching woody plants. Thus, the grass-dominated marsh communities are reduced.

Fire Prescriptions

Ecosystem management has brought new challenges to the application of prescribed fire primarily due to the increased scale and complexity of some prescribed burning (Zimmerman and Bunnell 1998). Traditionally, prescribed fire was applied on small, relatively homogeneous units of a single land ownership. Prescribed fire will continue to be important for small-scale operations. But to meet some ecosystem goals, prescribed fire needs to be applied over extensive areas that contain a variety of vegetation communities and fuel conditions.

In designing fire prescriptions, a strong, clear connection is needed between ecosystem goals, resource objectives, and fire objectives. This helps assure that prescribed fire will accomplish the desired effects. It can also help in choosing proper technical aids for determining the prescription and in assuring fires are cost effective and safely conducted. Designing prescriptions through a visible, logical process can also demonstrate professional competence and promote credibility of those in charge of the prescribed fire activities.

Defining fire objectives boils down to specifying first order fire effects that describe what the burning should immediately accomplish (Brown 1985b). Treatment objectives need to specify: (1) how much of what kind of organic matter should be consumed, (2) what vegetation should be killed, and (3) what the size of burned and unburned patches should be. Constraints on achieving the treatment objectives must also be considered. These can be thought of as the fire effects that should be avoided. Controlling fire, managing smoke, and avoiding overstory mortality are the common constraints. Specifying objectives and constraints is a matter of declaring what the fire should accomplish and avoid. Both are fire objectives of sorts, so why regard them differently? One reason is that it helps in demonstrating an awareness of beneficial and undesirable aspects of fire and in explaining the prescribed fire plans to others.

Depending on resource objectives, the fire objectives may call for a wide or narrow prescription window. For example, the resource objective to restore fire as a process in a nonlethal understory fire regime type may only require that prescribed fire be able to spread with minimal mortality to the overstory, an objective that could be accomplished with a wide prescription window. The specific resource objective of attaining natural regeneration while retaining some large downed woody material may call for a fire objective that specifies exposure of 20 to 30 percent mineral soil without consuming more than half of the large downed woody material. This would require a narrow prescription window.

Occasionally, conflicts may arise between fire objectives and constraints. A common example is between the objective to reduce fuels by burning at low fuel moistures and the constraint to control smoke production. Conflict can arise between different objectives; for example, to expose a high percentage of mineral soil and to leave large downed woody material for other ecosystem benefits. When conflicts arise, compromise may prevent the fire from achieving the resource objectives. It is important to recognize those situations so a potentially unsuccessful prescribed fire can be avoided.

Many technical aids are available to assist in preparing fire prescriptions. Most involve prediction of information such as weather probabilities, fuel loadings, fuel consumption, fire behavior, tree mortality, and plant response. Two technical aids—both with user guides that can help in writing and explaining prescribed fire objectives and designing fire prescriptions—are relevant for applications across the United States and much of Canada. They are the Fire Effects Information System-FEIS (Fischer and others 1996) and the First Order Fire Effects Model-FOFEM (Reinhardt and others 1997).

FEIS—This is an easy to use, computerized knowledge management system that stores and retrieves current information as text organized in an encyclopedic fashion. FEIS provides fire effects and related biological, ecological, and management information in three major categories: plant species, wildlife species, and plant communities. The plant species category includes for each species, information on taxonomy, distribution and occurrence, value and use, botanical and ecological characteristics, fire ecology, fire effects, and references. A citation retrieval system can be searched independently by author and keyword. Although the system was originally developed to meet prescribed fire needs, it is now recognized as a valuable aid for obtaining information about species ecology for any application. It can be accessed through a U.S. Forest Service Web site:

http://www.fs.fed.us/database/feis

FOFEM—This system was developed to predict the direct consequences of fire, that is, first order fire effects. FOFEM computes duff and woody fuel consumption, mineral soil exposure, fire-caused tree mortality, and smoke production for many forest and rangeland ecosystems. An update is scheduled to add soil heating effects. FOFEM contains a fire effects calculator to predict effects of fire from the burning conditions and a prescribed fire planner to compute the burn conditions necessary to achieve a desired effect. Users may enter their own fuel data or use default values derived from fuel models provided for natural and activity fuels by many forest cover types. The model is implemented in a computer program available for use on a PC or Forest Service computer. To obtain a current version of the FOFEM software, contact the authors at the Intermountain Fire Sciences Laboratory, (406) 329-4800, or PO Box 8089, Missoula, MT 59807.

Research Needs

The goals of maintaining sustainability of all ecosystem components and processes and conserving biodiversity present new challenges to land management organizations. Knowledge of how ecosystems function and what they provide is essential to making informed environmental decisions. The following broadly stated research needs indicate the knowledge required for managing fire effects on flora and fuel that will contribute to maintaining sustainable ecosystems.

Characteristics of Fire Regimes

- What is the historical range of variability in fire regime characteristics especially fire frequency, seasonality, and severity for fire dependent ecosystems? This should be answered for multiple spatial scales because of the hierarchical structure of ecosystems.
- What are the limits to ecosystem patterns and processes that signal ecosystems are beyond the boundaries of the historical range of variability?
- To what extent has climate influenced fire regime characteristics in the past? How might anticipated climate change alter fire regime characteristics in the future?

Effects of Fire on Ecosystem Processes and Biodiversity

- What are the long-term effects of fire of varying frequencies and severities on nutrient dynamics and vegetation?
- How does fire of varying frequency, seasonality, and severity influence individual plant species and plant community development? The emphasis for research should be on rare species and other vegetation components where knowledge is lacking.
- What interactions between insects and diseases and fire characterized historical fire regimes, and how has this affected landscape patterns? How do these interactions change when ecosystems exceed the natural range of variability and when various management activities are applied?
- What is the interaction of different ecosystem scales on ecosystem processes and biodiversity? To what extent can coarse scale analysis account for ecosystem processes and biodiversity?
- What are the long-term effects of largely excluding fire from ecosystems that evolved under fire regimes?

Restoration of Ecosystems

- What approaches and methods involving wildland fire use, prescribed fire, silviculture, and grazing can be used to restore ecosystems to a semblance of the historical range of vegetation composition and structure while meeting the resource needs of society?
- What fuel management activities can provide an acceptable level of fire hazard and remain compatible with ecosystem goals, especially needs for coarse woody debris?
- How can nonindigenous plant species be managed in combination with prescribed fire and resource utilization activities to maintain biodiversity?

Development of Ecosystem Evaluation Methodologies

- Continue with development of simulation models and ecosystem evaluation techniques that can help in understanding and managing ecosystem dynamics. Succession and landscape models are needed that account for interaction of fire, vegetation, fuels, and climate.
- Fire effects models at small spatial and temporal scales are needed for rigorous fire effects hypothesis testing and as building blocks for models with larger temporal and spatial scales.
- Determine organizational approaches that allow complex ecosystem models requiring specialized skills and high speed computer facilities to be accessible to all land management organizations and units.

Notes

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Appendices

Appendix A: Common and Scientific Names of Plant Species

Common names mentioned in the text and scientific names follow the nomenclature of the U.S. Department of Agriculture (1999) PLANTS database except for some wet grassland species, which follow the Integrated Taxonomic Information System (http://www.itis.usda.gov/). For some species a second common name is shown because it is commonly used.

Common name	Scientific name
Trees and Shrubs	
a'ali'i	*Dodonaea viscosa*
Alaska-cedar	*Chamaecyparis nootkatensis*
alder	*Alnus* spp.
alligator juniper	*Juniperus deppeana*
American beech, beech	*Fagus grandifolia*
American elm, white elm	*Ulmus americana*
American mountain-ash	*Sorbus americana*
antelope bitterbrush	*Purshia tridentata*
Arizona pine	*Pinus ponderosa* var. *arizonica*
Arizona white oak	*Quercus arizonica*
Ashe juniper	*Juniperus ashei*
aspen, trembling aspen, quaking aspen	*Populus tremuloides*
Atlantic white-cedar	*Chamaecyparis thyoides*
baccharis	*Baccharis* spp.
baldcypress	*Taxodium distichum*
balsam fir	*Abies balsamea*
balsam poplar	*Populus balsamifera*
basin big sagebrush	*Artemisia tridentata* ssp. *tridentata*
basswood	*Tilia americana*
batis	*Batis maritima*
bayberry	*Morella* spp.
beaked hazel	*Corylus cornuta*
bear oak	*Quercus ilicifolia*
Bebb willow	*Salix bebbiana*
big sagebrush	*Artemisia tridentata*
bigleaf maple	*Acer macrophyllum*
bigleaf sumpweed	*Iva frutescens*
bitter cherry	*Prunus emarginata*
bitternut hickory	*Carya cordiformis*
black ash	*Fraxinus nigra*
black cherry	*Prunus serotina*
black cottonwood	*Populus trichocarpa*
black greasewood	*Sarcobatus vermiculatus*
black mangrove	*Avicennia germinans*
black oak	*Quercus velutina*
black spruce	*Picea mariana*
black walnut	*Juglans nigra*
blackbrush	*Coleogyne ramosissima*
blackgum, black tupelo	*Nyssa sylvatica*
blackjack oak	*Quercus marilandica*

Common name	**Scientific name**
blue huckleberry	*Vaccinium membranaceum*
blue oak	*Quercus douglasii*
blue spruce	*Picea pungens*
bluejack oak	*Quercus incana*
bog blueberry	*Vaccinium uliginosum*
bog labrador tea	*Ledum groenlandicum*
boxelder, Manitoba maple	*Acer negundo*
Brazilian pepper	*Schinus terebinthifolus*
buckeye	*Aesculus*
buckwheat tree	*Cliftonia monophylla*
bur oak	*Quercus macrocarpa*
bursage spp.	*Ambrosia* spp.
butternut	*Juglans cinerea*
buttonwood, button mangrove	*Conocarpus erectus*
cabbage palmetto	*Sabal palmetto*
California black oak	*Quercus kelloggii*
California red fir	*Abies magnifica*
California sagebrush	*Artemisia californica*
canyon live oak	*Quercus chrysolepis*
Caribbean pine, Honduras pine	*Pinus caribaea*
Carolina ash, pop-ash	*Fraxinus caroliniana*
cenzia, purple sage	*Leucophyllum frutescens*
chamise	*Adenostoma fasciculatum*
Chapman oak	*Quercus chapmanii*
chestnut oak	*Quercus prinus*
Chinese tallow	*Sapium sebiferum*
chokecherry	*Prunus virginiana*
coast Douglas-fir	*Pseudotsuga menziesii* var. *menziesii*
coast live oak	*Quercus agrifolia*
coastalplain staggerbush	*Lyonia fruticosa*
cocoplum	*Chrysobalanus icaco*
common persimmon	*Diospyros virginiana*
creeping barberry	*Mahonia repens*
creosotebush	*Larrea tridentata*
cypress	*Taxodium* spp.
cyrilla, swamp cyrilla	*Cyrilla racemiflora*
dahoon	*Ilex cassine*
digger pine, California foothills pine	*Pinus sabiniana*
Douglas-fir	*Pseudotsuga menziesii*
dwarf chinkapin oak	*Quercus prinoides*
dwarf huckleberry	*Gaylussacia dumosa*
eastern baccharis, groundsel-tree	*Baccharis halimifolia*
eastern cottonwood	*Populus deltoides*
eastern hemlock	*Tsuga canadensis*
eastern redcedar	*Juniperus virginiana*
eastern white pine	*Pinus strobus*
Engelmann spruce	*Picea engelmannii*
eucalyptus	*Eucalyptus* spp.
fetterbush	*Lyonia lucida*
flowering dogwood	*Cornus florida*
forage kochia	*Kochia prostrata*
fourwing saltbush	*Atriplex canescens*
Fraser fir	*Abies fraseri*
gallberry, inkberry	*Ilex glabra*
Gambel oak	*Quercus gambelii*

Common name	Scientific name
giant sequoia	*Sequoiadendron giganteum*
gooseberry, currant	*Ribes* spp.
grand fir	*Abies grandis*
gray birch	*Betula populifolia*
green ash	*Fraxinus pennsylvanica*
greenbriar	*Smilax glauca, Smilax* spp.
ground blueberry	*Vaccinium myrsinites*
hickory	*Carya* spp.
hoaryleaf ceanothus	*Ceanothus crassifolius*
honey mesquite	*Prosopis glandulosa*
horsebrush	*Tetradymia* spp.
incense-cedar	*Calocedrus decurrens*
interior live oak	*Quercus wislizenii*
interior ponderosa pine	*Pinus ponderosa* var. *scopulorum*
jack pine	*Pinus banksiana*
Jeffrey pine	*Pinus jeffreyi*
Joshua tree	*Yucca brevifolia*
juniper	*Juniperus* spp.
koa	*Acacia koa*
large gallberry	*Ilex coriacea*
laurel oak	*Quercus laurifolia*
leatherleaf	*Chamaedaphne calyculata*
live oak	*Quercus virginiana*
loblolly pine	*Pinus taeda*
loblolly-bay	*Gordonia lasianthus*
lodgepole pine	*Pinus contorta*
longleaf pine	*Pinus palustris*
lyonia	*Lyonia* spp.
manzanita	*Arctostaphylos* spp.
melaleuca	*Melaleuca quinquenervia*
mesquite	*Prosopis* spp.
mockernut hickory	*Carya tomentosa*
Mormon tea	*Ephedra torreyana*
mountain alder	*Alnus incana*
mountain big sagebrush	*Artemisia tridentata* ssp. *vaseyana*
mountain hemlock	*Tsuga mertensiana*
mountain-laurel	*Kalmia latifolia*
myrsine	*Myrsine quianensis*
myrtle oak	*Quercus myrtlifolia*
myrtle, wax myrtle, southern bayberry	*Morella cerifera*
noble fir	*Abies procera*
northern pin oak	*Quercus ellipsoidalis*
northern red oak	*Quercus rubra*
northern white-cedar, e. white-cedar	*Thuja occidentalis*
oneseed juniper	*Juniperus monosperma*
oak	*Quercus* spp.
Oregon white oak	*Quercus garryana*
Pacific madrone	*Arbutus menziesii*
Pacific ponderosa pine	*Pinus ponderosa* var. *ponderosa*
Pacific silver fir	*Abies amabilis*
paloverde spp.	*Cercidium* spp.
paper birch	*Betula papyrifera*
persimmon	*Diospyros* spp.
piedmont staggerbush	*Lyonia mariana*
pignut hickory	*Carya glabra*

Common name	**Scientific name**
pin cherry, fire cherry	*Prunus pensylvanica*
pin oak	*Quercus palustris*
pinyon pine	See singleleaf pinyon, true pinyon
pitch pine	*Pinus rigida*
pond cypress	*Taxodium ascendens*
pond pine	*Pinus serotina*
ponderosa pine	*Pinus ponderosa*
poplar	*Populus* spp.
post oak	*Quercus stellata*
rabbitbrush	*Chrysothamnus* spp.
raspberry, blackberry	*Rubus* spp.
red alder	*Alnus rubra*
red bay	*Persea borbonia*
red elderberry	*Sambucus racemosa* ssp. *pubens*
red mangrove	*Rhizophora mangle*
red maple	*Acer rubrum*
red pine	*Pinus resinosa*
red raspberry	*Rubus idaeus*
red spruce	*Picea rubens*
redberry juniper	*Juniperus erythrocarpa*
redstem ceanothus	*Ceanothus sanguineus*
redwood	*Sequoia sempervirens*
rhododendron	*Rhododendron* spp.
Rocky Mountain Douglas-fir	*Pseudotsuga menziesii* var. *glauca*
Rocky Mountain juniper	*Juniperus scopulorum*
Rocky Mountain lodgepole pine	*Pinus contorta* var. *latifolia*
Rocky Mountain maple	*Acer glabrum*
rosemary	*Ceratiola ericoides*
rusty staggerbush	*Lyonia ferruginea*
sagebrush	*Artemisia* spp.
salmonberry	*Rubus spectabilis*
sand live oak	*Quercus virginiana* var. *maritima*
sand pine	*Pinus clausa*
sand post oak	*Quercus stellata* var. *margaretta*
sand shinnery oak	*Quercus havardii*
Saskatoon serviceberry	*Amelanchier alnifolia*
saw palmetto	*Serenoa repens*
scarlet oak	*Quercus coccinea*
scrub oak	*Quercus dumosa*
scrub palmetto	*Sabal etonia*
shadscale	*Atriplex confertifolia*
sheep-laurel	*Kalmia angustifolia*
shore pine	*Pinus contorta* var. *contorta*
shortleaf pine	*Pinus echinata*
silver maple	*Acer saccharinum*
silver sagebrush	*Artemisia cana*
silverbell	*Halesia Ellis*
singleleaf pinyon	*Pinus monophylla*
Sitka spruce	*Picea sitchensis*
slash pine	*Pinus elliottii*
snakeweed	*Gutierrezia* spp.
snowbrush ceanothus	*Ceanothus velutinus*
sourwood	*Oxydendrum arboreum*
southern magnolia	*Magnolia grandiflora*

Common name	Scientific name
southern red oak, cherrybark oak	*Quercus falcata*
speckled alder	*Alnus rugosa*
spiny hopsage	*Grayia spinosa*
spruce pine	*Pinus glabra*
subalpine fir	*Abies lasiocarpa*
sugar maple	*Acer saccharum*
sugar pine	*Pinus lambertiana*
sugarberry	*Celtis laevigata*
swamp bay	*Persea palustris*
swamp chestnut oak	*Quercus michauxii*
swamp tupelo	*Nyssa biflora*
sweetbay	*Magnolia virginiana*
sweetgum	*Liquidambar styraciflua*
sweetpepperbush, poor man's soap	*Clethra alnifolia*
sycamore	*Platanus occidentalis*
Table Mountain pine	*Pinus pungens*
tamarack	*Larix laricina*
tanoak	*Lithocarpus densiflora*
tarbush	*Flourensia cernua*
Texas persimmon	*Diospyros texana*
thimbleberry	*Rubus parviflorus*
thinleaf alder	*Alnus incana* ssp. *tenuifolia*
threetip sagebrush	*Artemisia tripartita*
toyon	*Heteromeles arbutifolia*
trefoil	*Lotus* spp.
true pinyon, Colorado pinyon	*Pinus edulis*
turkey oak	*Quercus laevis*
Utah juniper	*Juniperus osteosperma*
varnish leaf	*Dodonea virginiana*
Virginia pine	*Pinus virginiana*
water oak	*Quercus nigra*
water tupelo	*Nyssa aquatica*
western hemlock	*Tsuga heterophylla*
western juniper	*Juniperus occidentalis*
western larch	*Larix occidentalis*
western redcedar	*Thuja plicata*
western white pine	*Pinus monticola*
white ash	*Fraxinus americana*
white bully	*Sideroxylon salicifolium*
white bursage	*Ambrosia dumosa*
white fir	*Abies concolor*
white mangrove	*Laguncularia racemosa*
white oak	*Quercus alba*
white sage	*Salvia apiana*
white spirea	*Spiraea betulifolia*
white spruce	*Picea glauca*
whitebark pine	*Pinus albicaulis*
willow	*Salix* spp.
willow oak	*Quercus phellos*
winterfat	*Krascheninnikovia lanata*
Wyoming big sagebrush	*Artemisia tridentata* ssp. *wyomingensis*
yaupon	*Ilex vomitoria*
yellow birch	*Betula alleghaniensis*

Common name	Scientific name
yellow buckeye	*Aesculus octandra*
yellow paloverde	*Cercidium microphyllum*
yellow-poplar	*Liriodendron tulipifera*
Grasses and Forbs	
alligatorweed	*Alternanthera philoxeroides*
American white waterlily	*Nymphaea odorata*
annual fleabane	*Erigeron annuus*
annual ryegrass, Italian ryegrass	*Lolium perenne* spp. *multiflorum*
arrowhead	*Sagittaria* spp.
arrowleaf balsamroot	*Balsamorhiza sagittata*
basin wildrye	*Leymus cinereus*
beakrush, Tracy's beaksedge	*Rhynchospora tracyi*
big bluestem	*Andropogon gerardii* var. *gerardii*
black grama	*Bouteloua eriopoda*
blackeyed Susan	*Rudbeckia hirta*
blue grama	*Bouteloua gracilis*
bluebunch wheatgrass	*Pseudoroegneria spicata*
blue-eyed grass	*Sisyrinchium campestre*
bluestem spp.	*Schizachyrium* spp.
bottlebrush squirreltail	*Elymus elymoides*
broadleaf arnica	*Arnica latifolia*
brome	*Bromus* spp.
broomsedge, broomsedge bluestem	*Andropogon virginicus*
buffalograss	*Buchloe dactyloides*
bulltongue arrowhead	*Sagittaria lancifolia*
Burma reed	*Neyrundia reynaudiana*
bush muhly	*Muhlenbergia porteri*
cane, switch cane	*Arundinaria gigantea*
cattail	*Typha* spp.
chairmaker's bullrush	*Schoenoplectus americanus*
chalky bluestem	*Andropogon capillipes*
cheatgrass	*Bromus tectorum*
cogongrass	*Imperata cylindrica*
Columbian bluestem, bush beardgrass	*Schizachyrium condensatum*
common camas	*Camassia quamash*
cordgrass	*Spartina* spp.
curly-mesquite	*Hilaria belangeri*
Curtis' dropseed	*Sporobolus curtissii*
cutgrass	*Zizaniopsis* spp.
deathcamas	*Zigadenus venenosus*
dropseed, sacaton	*Sporobolus* spp.
fescue	*Festuca* spp.
fireweed	*Epilobium angustifolium*
fountain grass	*Pennisetum sataceum*
gayfeather	*Liatris* spp.
glacier lily	*Erythronium grandiflorum*
golden brodiaea	*Brodiaea ixioides*
green arrow arum	*Peltandra virginica*
gulf cordgrass	*Spartina spartinae*
hairawn muhly	*Muhlenbergia capillaris*
heartleaf arnica	*Arnica cordifolia*

Common name	**Scientific name**
hydrocotyle, marshpennywort	*Hydrocotyle* spp.
Idaho fescue	*Festuca idahoensis*
Indian paintbrush	*Castilleja* spp.
Indian ricegrass	*Oryzopsis hymenoides*
Indiangrass	*Sorghastrum nutans*
inland saltgrass	*Distichlis spicata*
Lehmann lovegrass	*Eragrostis lehmanniana*
little bluestem	*Andropogon scoparius*
little bluestem, creeping bluestem	*Schizachyrium scoparium*
lupine	*Lupinus* spp.
maidencane	*Panicum hemitomon*
milkvetch	*Astragalus* spp.
molasses grass	*Melinis minutiflora*
muhly spp.	*Muhlenbergia* spp.
natal redtop, rose natalgrass	*Melinis repens*
needlegrass rush, black rush	*Juncus roemerianus*
panicum	*Panicum* spp.
pasque flower	*Anemone pratens*
pickerelweed	*Pontederia cordata*
pickleweed	*Salicornia* spp.
pili grass	*Heteropogon contortus*
pinegrass	*Calamagrostis rubescens*
pineland threeawn, wiregrass	*Aristida stricta*
pitcherplant	*Sarracenia purpurea*
pond lily	*Nuphar* spp.
prairie cordgrass	*Spartina pectinata*
prairie violet	*Viola pedatifida*
red brome	*Bromus rubens*
reed	*Phragmites* spp.
rough fescue	*Festuca scabrella*
rush	*Juncus* spp.
saltgrass	*Distichlis* spp.
saltmeadow cordgrass	*Spartina patens*
saltmeadow rush	*Juncus gerardii*
sand cordgrass	*Spartina bakeri*
sand dropseed	*Sporobolus cryptandrus*
Sandberg bluegrass	*Poa secunda*
sawgrass	*Cladium* spp.
seaside tansy	*Borrichia* spp.
sheathed cottonsedge	*Eriophorum vaginatum*
showy aster	*Aster conspicuus*
showy partridgepea	*Cassia fasciculata*
sideoats grama	*Bouteloua curtipendula*
sky blue aster	*Aster azureus*
slender bluestem	*Schizachyrium tenerum*
smooth cordgrass	*Spartina alterniflora*
spikerush	*Eleocharis* spp.
star-flowered Solomon's seal	*Smilacina stellata*
strawberry	*Fragaria* spp.
switchgrass	*Panicum virgatum*
thatching grass	*Hyparrhenia rufa*
Thurber's needlegrass	*Achnatherum thurberiana*

Common name	Scientific name
tobosa	*Pleuraphis mutica*
trumpet pitcherplant, yellow pitcherplant	*Sarracenia flava*
twinflower	*Linnaea borealis*
waterlily	*Nymphaea* spp.
western wheatgrass	*Pascopyrum smithii*
western yarrow	*Achillea millefolium*
wheatgrass	*Agropyron* spp.
wild columbine	*Aquilegia canadensis*
wild hollyhock	*Iliamna rivularis*
wild sarsaparilla	*Aralia nudicaulis*
wildrice	*Zizania* spp.
wiregrass	*Aristida* spp.
Mosses, Ferns, Cactus, and Lichens	
bracken fern	*Pteridium aquilinum*
cholla, prickly pear	*Opuntia* spp.
cup lichen	*Cladonia* spp.
dicranum	*Dicranum* spp.
knight’s plume moss	*Ptilium crista-castrensis*
mountain fern moss	*Hylocomium splendens*
Old world or small-leaf climbing fern	*Lygodium microphyllum*
prickly pear cactus	*Opuntia humifusa*
saguaro	*Carnegia gigantea*
Schreber’s moss	*Pleurozium schreberi*
sphagnum	*Sphagnum* spp.

Appendix B: Succession Simulation Models

FIRESUM–FIRE SUcession Model is a stand level, individual tree ecosystem process model developed for western conifers, especially ponderosa pine and whitebark pine, to simulate effects of different fire regimes on tree composition, stand structure, and fuel loadings (Keane and others 1990, 1989).

FVS–Forest Vegetation Simulation is a stand level mensurational model (Wykoff and others 1982).

FFE–Fire and Fuels Extension to the FVS is a stand level model for simulating surface fuel loadings, tree characteristics, expected fire behavior, and expected tree mortality (Beukema and others 1999). This model is particularly useful for growth, mortality, falldown, and decay of conifer trees.

Fire-BGC–The Fire Biogeochemical Mechanistic process model can be used to investigate stand- and landscape-level changes in ecosystem processes and characteristics in fire-dominated environments (Keane and others 1996). It is useful for evaluating effects of climate change.

SIMPPLLE–Simulating Patterns and Processes at Landscape Scales is a rule-based model designed for Northern Rocky Mountain forest types (Chew 1997). It starts at a coarse scale and adds fine scale only as needed to produce acceptable performance.

VDDT—The Vegetation Dynamics Development Tool is a nonspatial, deterministic model where successional pathways connecting successional stages are used to explore community dynamics (Beukuma and Kurz 1995). VDDT only simulates one vegetation type at a time but is useful and efficient for simulating disturbance and succession on mid-scale to fine-scale landscapes. This model can be readily used by managers and is undergoing development for national applications.

CRBSUM–The Columbia River Basin Succession Model was used to simulate landscape changes for the Interior Columbia Basin Ecosystem Management Project (Keane and others 1996). It incorporates disturbance as a stochastic process and models succession for individual landscape pixels. CRBSUM 2 was created from CRBSUM to improve the simulation of fire processes over time and space (Keane and Long 1998).

LANDSUM–the Landscape Succession Model was derived from CRBSUM to operate on a polygon level rather than pixels. This allows it to be used at finer scales of resolution (Keane 1999).

FIREPAT–Fire Pattern Succession Model attempts to more realistically model fire by simulating fire ignition and size to compute number of pixels disturbed by fire (Keane and Long 1998). It operates at a coarse scale and models succession similar to CRBSUM.

INTELAND–Integrated Terrestrial Landscape Model simulates natural processes in boreal forest ecosystems (Gauthier and others 1998). It was designed as a GIS-based model to help define natural system baselines for disturbance regimes, vegetation dynamics, wildlife species composition, and landscape diversity.

Appendix C: Glossary

The terminology here was derived from the following references: fuels and fire behavior from Agee (1993), Brown and others (1982), and Ryan and Noste (1985); fire occurrence from Agee (1993), Johnson (1992), and Romme (1980); and plant reproduction from Allaby (1992), Helms (1998), and Sutton and Tinus (1983).

Fuels

Fuel comprises living and dead vegetation that can be ignited. It is often classified as dead or alive and as natural or activity fuels. Fuel components refer to such items as downed dead woody material by various size classes, litter, duff, herbaceous vegetation, live foliage, live shrub stems and so forth.

Kinds of Fuel

dead fuels: Fallen dead vegetation such as downed woody material, litter, duff, and organic soils and dead upright vegetation such as cured grasses, forbs, and dead attached shrub stemwood.

live fuels: Living plants. Especially important components include tree crowns, shrubs, grasses, forbs, and ferns.

natural fuels: These result from plant growth and death, loss of foliage, branch breakage, and tree blowdown.

activity fuels: These fuels result from human activity such as logging, thinning, chaining, and herbicide use. It usually refers to residues from cutting operations.

down, dead woody fuels: Dead twigs, branches, stems, and boles of trees and shrubs that have fallen and lie on or near the ground (Brown and others 1982). Wood includes sound and rotten components.

litter: The top layer of the forest floor (01 soil horizon), including 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 dead fine fuel. In grasslands, litter is the accumulated dead herbaceous material usually referred to as thatch.

duff: Partially decomposed organic matter lying beneath the litter layer and above the mineral soil. It includes the fermentation and humus layers of the the forest floor (02 soil horizon).

organic soils: The deep layers of organic matter that frequently develop in poorly drained areas such as bogs, swamps, and marshes.

Fuel Properties

loading: The 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 0.25, 0.25 to 1, and 1 to 3 inches and for large material greater than 3 inches.

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.

total fuel: The amount of biomass that potentially could burn.

available fuel: The amount of biomass that will burn under a given set of conditions. Moisture content and fuel size are the primary determinants of availability. Arrangement and compactness of fuel may also determine availability.

fuel moisture content: This is expressed as a percent or fraction of oven dry weight of fuel. It is the most important fuel property controlling flammability. In living plants it is physiologically bound. Its daily fluctuations vary considerably by species but are usually above 80 to 100 percent. As plants mature, moisture content decreases. When herbaceous plants cure, their moisture content responds as dead fuel moisture content, which fluctuates according to changes in temperature, humidity, and precipitation.

Fire Behavior

type of fire: Refers to the fuels that are primarily supporting the fire namely surface fires, ground fires, and crown fires.

surface fires: These fires burn in litter and other live and dead fuels at or near the surface of the ground mostly by flaming combustion.

ground fires: These burn in the organic material below the litter layer mostly by smoldering combustion. Fires in duff, peat, dead moss and lichens, and punky wood are typically ground fires.

crown fires: These burn in the crowns of trees and shrubs usually ignited by a surface fire. They are common in coniferous forests and chaparral type shrublands.

fireline intensity: Also called Byram's intensity, this is 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. It is mathematically related to fireline intensity and tree crown scorch height.

total heat release: The heat released by combustion during burnout of all fuels in BTU per square foot or kilocalories per square meter.

fire duration: The length of time that combustion occurs at a given point. It relates closely to downward heating and fire effects below the fuel surface as well as heating of tree boles above the surface.

ground char: A qualitative measure of a fire's heat pulse downward into the soil. It is determined by visually judging the extent of fuel consumption, charring, and changes in soil texture (Ryan and Noste 1985). It is also referred to as **burn severity** and **depth of burn**, which is a quantitative expression of depth of forest floor consumed by fire. It is largely determined by fire duration and characteristics of the soil including the forest floor.

fire severity: A qualitative measure of the immediate effects of fire on the ecosystem. It relates to the extent of mortality and survival of plant and animal life both aboveground and belowground and to loss of organic matter. It is determined by heat released aboveground and belowground. Ryan and Noste (1985) describe a method for rating fire severity based on flame length and depth of burn.

Fire Occurrence

The definitions here were based on a review of fire history terminology at a fire history workshop (Romme 1980) and phraseology by Agee (1993).

fire cycle: A fire return interval calculated using a negative exponential distribution, applied using current age-class structure on the landscape. It is the average stand age of a forest characterized using the negative exponential distribution.

fire frequency: A general term referring to the recurrence of fire in a given area over time. It is sometimes stated as number of fires per unit time in a designated area. It is also used to refer to the probability of an element burning per unit time (Johnson 1992).

fire rotation: The length of time necessary for an area equal in size to the study area to burn and is equal to the **fire cycle**.

mean fire-return interval: The arithmetic average of all fire intervals in a given area over a given time period.

Plant Reproduction

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.

bulb: An underground storage organ that bears roots on its lower surface and fleshy leaves above. It provides a means of reproduction in perennials.

burl: A mass of woody tissue or wartlike structure fomed on stem, branch, or root; has numerous buds, which rarely develop further.

caudex: A largely underground woody stem base of an otherwise herbaceous perennial forb that produces leaves and flowering stems.

corm: An underground storage organ bearing adventitious roots and scale leaves. It may function as an organ of vegetative reproduction in perennials.

epicormic branch: A shoot arising spontaneously from an adventitious or dormant bud on the stem or branch of a woody plant often following exposure to increased light levels or fire.

lignotuber: A woody organ of food storage and regeneration forming a swelling of stem material, more or less at ground level, that originates from the axils of cotyledons or early seedling leaves.

rhizome: A creeping stem, not a root, growing beneath the surface consisting of a series of nodes with roots commonly produced from the nodes and producing buds in the leaf axils.

root collar: Loosely, the point along the main stem-root axis at which the primary vascular anatomy changes from that of a stem to that of a root, usually applied to trees. Transition point between stem and root. It may be clearly or vaguely apparent.

root crown: A mass of woody tissue from which roots and stems originate, usually applied to shrubs and herbaceous plants; can be considered as the point at which root and stem meet.

Index

RMRS
ROCKY MOUNTAIN RESEARCH STATION

The Rocky Mountain Research Station develops scientific information and technology to improve management, protection, and use of the forests and rangelands. Research is designed to meet the needs of National Forest managers, Federal and State agencies, public and private organizations, academic institutions, industry, and individuals.

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