Literature Review of Fire Ecology and Effects

United States Air Force Academy
Colorado Springs, Colorado

1997

Report submitted to:
U.S. Air Force Academy, Natural Resources

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Executive Summary

The natural resource planners at the U.S. Air Force Academy (USAFA) Colorado Springs, Colorado contracted the Colorado Natural Heritage Program (CNHP) to conduct a thorough scientific literature review of fire effects and ecology for plants, animals, and natural communities that occur within their purview. The report contains a detailed section adapted from the U.S. Forest Service Fire Effects Information System (FEIS 1997) which discusses fire effects and ecology for 32 dominant native plants within the four major vegetation zones. Discussions of fire history, regimes, case studies, and biological diversity are also included, as well as fire effects and ecology of six noxious plants and eight common animals.

The U.S. Air Force Academy is located within an area of ecological transition between the southern Rocky Mountains to the west and the Great Plains to the east. The influence of these mixed environments allows for a unique diversity of native plant communities. Fire has been, and continues to be, an important process that contributes to the biodiversity of the flora and fauna on USAFA. USAFA has been utilizing prescribed fire as a forest management tool since 1992 to reduce forest fuels, minimize wildfire threat, improve forest and grassland ecosystems, and increase species diversity. There are two globally rare plants (Porter's feathergrass and southern Rocky Mountain cinquefoil), one federally proposed endangered mammal (meadow jumping mouse subspecies preblei), and several state rare plants and animals that are known to occur within the 18,455 acres of USAFA (CNHP 1997).

Suggested prescribed burning guidelines are derived from the literature review for USAFA's priority areas (Jacks Valley, west of Rifle Range, the foothills, Farish Memorial Recreation Area, and Monument Creek). **The general guidelines presented herein are derived from case studies and should not be interpreted as fire prescriptions.** Fire prescriptions are precise statements of a fire's behavior (intensity, duff consumption, rate of spread, frequency). Effective fire management and development of proper fire prescriptions require an understanding of fire processes and heat transfer that explain characteristics of fire behavior as well as an understanding of how fire behavior is coupled to specific fire effects (Johnson and Miyanishi 1995).

An important point to recognize is that a single prescription for a given vegetation type does not exist. The best approach to prescribed burning is to use an experimental design in which a range of fire behavior and fire effects can be obtained, rather than to make a decision based solely on one particular fire regime. Eighteen permanent plots were established at Farish Memorial Recreation Area to provide the groundwork for a long-term study of fire effects and processes in the mixed coniferous forest (Appendix A). The regional effect of fire on vegetation is influenced by a variety of factors, including precipitation patterns before and after a burn, the composition of the vegetation, topography, and the time of the burn. Fire managers need to factor in fuel load and moisture, scale, and topography for each proposed burn. This report is intended to provide a knowledge base to assist in the development of a comprehensive fire management plan for USAFA.

Suggested Burning Guidelines

Prairie grasslands and oak shrubland--Jacks Valley and west of Rifle Range

Prescribed low-intensity fires in grasslands would be most effective if applied in the spring or fall every 2-3 years to maximize the benefit of post-fire recovery. This fire regime will favor the native perennials, maintain or reduce shrub density, and discourage introduced species. Grazing intensity pre- and post-fire needs to be considered (currently there is no livestock grazing on USAFA).

Late spring burning is the best time to use prescribed burning to increase densities of big and little bluestem and other warm-season grasses.

Gambel's oak is seldom killed by fire, however control of Gambel's oak can be accomplished by high-intensity biennial burns. Additional control may require at least three treatments (burning, mechanical removal, or chemical control) in successive years. To create a mosaic of young stands, burning every 30 years in the fall after leaf fall is recommended. Presence of a continuous fuel bed and weather (e.g., high winds) needs to be closely monitored for each burn.

The response of mountain-mahogany to fire may vary seasonally. High- and low-severity fire treatments applied to mountain-mahogany during the dormant season can be more effective in increasing biomass production than those applied during the growing season.

Ponderosa pine and mixed conifers--Foothills and Farish Memorial Recreation Area

Ponderosa pine/bunchgrass savanna can be maintained using low-intensity ground fires at a 3-5 year frequency to reduce fuel and thin tree seedlings. Timing is critical; too early in the fuel cycle and there will be insufficient fuel to carry an effective fire; too late and fires may be intense resulting in high tree mortality. The primary factors to consider for prescribed burning are fuel moisture and production.

Use moderate to high-intensity fires to create patch mosaics that increase species diversity and improve the heterogeneity of the stand.

Maintenance of vigorous aspen forest requires moderate-intensity fire, once every 100-300 years in the late summer or fall. To promote suckering and stand rejuvenation, a low-intensity, 2-5 year fire frequency is required.

Coniferous forests (e.g., Douglas-fir, Colorado blue spruce, white fir, etc.) in the southern Rocky Mountains are characterized by frequent, low- to moderate-intensity surface fires. Where ponderosa pine is a major associate, prescribed fire frequencies for Douglas-fir are

approximately 1 every 10 years. In cooler sites, 10- to 30-year fire frequencies for Douglas-fir forests are more likely. The most important consideration for timing of a prescribed burn in a mixed conifer forest is fuel moisture.

Riparian forests--Monument Creek

Low to moderate-intensity fire every 25 to 50 years will encourage root sprouting in cottonwoods and willows. Fire thins the overstory, allowing more light penetration and exposing the mineral soil. This allows seeds to establish if soil moisture is adequate.

Animals

Prescribe burning in meadow jumping mouse subspecies preblei habitats (e.g., riparian areas) should be performed between October and late April when the jumping mouse is in hibernation. Additional research is needed to determine specifically the fire effects on this federally proposed endangered subspecies.

Quick, hot fires will benefit wildlife species (e.g., deer, elk, and beaver) in the following ways: creation of mosaics/various seral stages, improvement of forage (e.g., aspen suckers, cottonwoods, willows, Rocky Mountain maple, mountain-mahogany, and snowberry sprouts), reduction of shrubs, and creation of corridors.

Prescribed fire could be used to enhance habitat, create seral stage diversity, and increase the prey base for raptors.

Controlled burning should be used in moderation where fire sensitive invertebrate species are present. Effects on butterfly nectar or host plants need to be considered when determining fire intensity and time of year of a prescribed burn.

Noxious Plants

In an Idaho study, burning cheatgrass in the fall or early summer reduced next spring's seed production. However in other studies, frequent fires were found to actually favor cheatgrass by eliminating competing perennial vegetation.

Control of smooth brome can be accomplished by burning in the spring and early summer when it is actively growing. Fuel moisture and fuel accumulation need to be considered for each burn.

Japanese brome can be contained or killed with prescribed fire every 5 years or less. Prescribed burning is most effective when Japanese brome is either in seed (late summer) or during a low precipitation period to maximize the reduction.

Canada thistle can be controlled and sometimes destroyed using consecutive spring burning of low to moderate-intensity fires.

Fire alone is not effective in the removal of spotted knapweed and may cause its increase.

Introduction

Purpose

The U.S. Air Force Academy (USAFA) reintroduced fire on its land in 1992 to reduce forest fuels, improve forage conditions, and prevent catastrophic wildfires. Prior to 1992, fires were not allowed to burn since the establishment of USAFA in 1954. Natural resource planners at USAFA contracted the Colorado Natural Heritage Program (CNHP) to begin monitoring the effects of fire on native vegetation. The focus of the data collection was to document changes in plant communities in burned areas. The information gathered on the ecological processes created and maintained by fire will provide the groundwork in developing a fire management plan for USAFA.

This report provides information on fire effects and ecology for 32 native dominant plants, 4 vegetation types, 6 noxious weeds, and 8 common animals that occur on USAFA. A thorough literature search was conducted using current available information including the U.S. Forest Service Fire Effects Information System (FEIS 1997), scientific journals, and local fire experts. Data from 18 permanent plots established at Farish Memorial Recreation Area are presented in Appendix B.

It is cautioned that natural processes such as fire are inherently variable. Predicating ecosystem responses to disturbance is very difficult due to the wide range of potential responses, and any attempt to anticipate the effects of change requires knowledge of reference conditions (Covich et al. 1994; Baker 1992; Veblen pers. comm.). A fire management plan must include fire behavior (e.g., intensity, duff consumption, rate of spread, frequency), as well as ecological effects. In the past, fire management plans have been based on ideas of the historical "natural" occurrence of fire and not on the fire effects desired (Johnson and Miyanishi 1995). Thus the decision to use fire is frequently based on as little information as was the previous decision to suppress fires, both being based on what appeared at the time to be self-evident reasoning. A past history of fires in an ecosystem is not necessarily justification for (nor does it provide enough understanding of) the use of prescribed fire (Johnson and Miyanishi 1995). There is a tendency to define the desired effects in terms of what was observed and is therefore considered to be natural. Ecosystems are dynamic. Choosing a particular time to mimic is arbitrary and often ignores the sequence of events that led up to that time. Baker (1992) concluded that landscapes that have been altered by settlement and fire suppression cannot be restored using traditional methods of prescribed burning, which will simply produce further alteration. Thomas Veblen, University of Colorado-Boulder, agrees stating that historical fire frequencies cannot be duplicated, due to the anthropogenic influences on fuels and their availability (pers. comm.).

Fire managers need to consider the scale and climatic variables of each potential burn. Brown (1997) states that patterning in fire regime parameters of frequency, spatial extent, severity, and seasonality occurs at multiple scales through time and across space owing to differences in scaling of driving variables. Local vegetation types and topography control to a large extent the occurrence and spread of individual fires by influencing fuel continuities and loadings. Annual

weather patterns control fuel conditions and fire ignitions on seasonal time scales (Brown 1997). Merrill Kaufman, U.S. Forest Service-Rocky Mountain Forest and Range Experiment Station concurred, stating that regional and local climatic conditions primarily control fuel availability (pers. comm.).

The overall recommendation derived from this literature search is to recognize that a single prescription for a given vegetation type does not exist. The best approach to prescribed burning is to use an experimental design in which a range of fire behavior and fire effects can be obtained rather than to make a decision on one particular fire regime (Veblen pers. comm; Brown pers. comm.; Kaufmann pers. comm.; Johnson and Miyanishi 1995). The permanent plots located at Farish Memorial Recreation Area can provide the groundwork for a long-term study of fire effects and processes in the mixed coniferous forest vegetation type.

The four priority areas to burn on USAFA are:

Farish Memorial Recreation Area
Farish Memorial Recreation Area has been targeted by USAFA natural resource planners for fall burning in 1996 and spring burning in 1997. Fire management will be used in this area to maintain grasslands for elk habitat. The Pike National Forest is also interested in coordinating a future large scale burn with USAFA in this area to burn aspen and Douglas-fir stands to improve and maintain aspen forests.

Monument Creek
The riparian vegetation along the creek is dominated by willow (*Salix exigua*) and intermittent stands of narrowleaf cottonwood (*Populus angustifolia*). Some willow stands show considerable accumulation of dead wood. These stands may benefit from burning.

Burning along Monument Creek riparian area would extend into the uplands where ponderosa pine (*Pinus ponderosa*), mixed prairie dominated by introduced grasses and patches of tallgrass prairie with big bluestem (*Andropogon gerardii*) occur. The purpose of burning in these communities is to enhance the native vegetation and control the introduced grasses (e.g., smooth brome (*Bromus inermis*), and cheatgrass (*Bromus tectorum*). USAFA is interested in improving wildlife habitat in the riparian locations.

A population of the meadow jumping mouse subspecies preblei (*Zapus hudsonius preblei*), a federally proposed endangered species, occurs in this area. Colorado Natural Heritage Program located the meadow jumping mouse subspecies preblei in 1995 in this area (Corn et al. 1995). Monument Creek is the best global occurrence for this subspecies of meadow jumping mouse (CNHP 1997). Monument Creek is divided into zones A and B. Zone A encompasses the creek and the willow carr. Zone B provides approximately a 0.5 mile buffer radius from the creek. This area is intended to be managed to protect the mouse from feral cat and coyote predation. CNHP would support an experiment involving prescribed fire and its effect on the mouse in Zone B before an accidental fire occurs (pers. comm. Chris Pague). Presently, little is known about the ecology of this subspecies in Colorado, including effects of fire on its habitat.

Ponderosa pine/grasslands in Jacks Valley
A thinned ponderosa pine (*Pinus ponderosa*) forest approximately 120 acres in size located northwest of the Jacks Valley training area is targeted for burning in 1997. The purpose of reintroducing fire to this area is to reduce fuels and restore the ecological function of fire which has been historically documented throughout the Front Range (Laven et al. 1980; Veblen and Lorenz 1991; Goldblum and Veblen 1992; Veblen and Kitzberger 1995).

Oak Scrub West of Rifle Range
Prescribed burns located in this area would be coordinated with the Pike National Forest and would be on a large scale, approaching 400 acres or more. The purpose of burning oak scrub is to improve bighorn sheep habitat, create corridors, and increase patch dynamics.

Fire, Landscape Pattern, and Biological Diversity

Fire is a major force in structuring pattern and species diversity in the landscape (Heinselman 1981). There is growing appreciation that disturbances like fire are processes that alter the structure and pattern of the landscape, and ultimately, species composition. Not only do disturbances create patch mosaics, future disturbance behavior is largely determined by the spatial distribution of these same patches (Pyne et al. 1996).

By inducing both spatial and temporal landscape heterogeneity, fire creates a variety of regeneration environments suitable for species colonization. Increased heterogeneity consequently leads to increased species diversity (Pyne et al. 1996; Kaufmann pers. comm.). In the past, ecologists have assumed environments to be homogeneous. Today, ecologists realize that ecosystems are so frequently disturbed that environmental homogeneity is ephemeral, if not theoretical. Landscapes truly are collections of ecosystems recovering from the most recent disturbances (Pyne et al. 1996).

Connell (1978) presented his intermediate disturbance hypothesis to explain the high species diversity found in tropical rainforests and coral reefs. He reasoned that when disturbances are relatively infrequent or small, strong competitors (resident species) eliminate weaker competitors. When disturbances are intermediate in frequency and intensity, the resultant environmental heterogeneity provides opportunities for both resident and colonizing species to persist, thereby maximizing biological diversity. The intermediate disturbance hypothesis has been invoked to explain the observed higher biological diversity associated with fire-prone ecosystems.

In a study of grassland communities in the Serengeti National Park, Joy (1992) states that species diversity increased after burning. Grime (1979) hypothesized that it is more likely that species will coexist in heterogeneous environments where niche diversification will occur. Denslow (1985) pointed out that disturbances that severely alter the composition of a community may

increase diversity because they increase the variability of the environment, thus precluding exclusive use of resources by one or more species.

Biodiversity has been variously defined over the years. According to Cooperrider (1991), "Biological diversity refers to the variety and variability among living organisms and ecological complexes in which they occur. Diversity can be defined as the number of different items and their relative frequency." Sandlund et al. (1992) define biodiversity as "the structural and functional variety of life forms at genetic, population, species, community and ecosystems levels".

The Colorado Natural Heritage Program's mission is to facilitate the protection of Colorado's natural biodiversity. CNHP is the state's primary comprehensive biological diversity data center. There are two globally rare plants (Porter's feathergrass and southern Rocky Mountain cinquefoil), one federally listed threatened mammal (meadow jumping mouse ssp. preblei), and several state rare plants and animals occur within the 18,455 acres of USAFA lands (CNHP 1997).

USAFA is located within an area of ecological transition between the mountains to the west and plains to the east. The influence of these mixed environments allows for a unique diversity and combination of native plant communities. Four vegetation zones are recognized on USAFA: prairie grassland, oak shrubland, mixed coniferous forests, and riparian areas (Ripley 1994; ESCO 1992). Each vegetation zone occurs broadly in various areas of North America, but on USAFA, they occur in close proximity to one another, creating a relatively uncommon mosaic of plant species.

Ripley (1994) and Murphy (ESCO Associates, Inc. 1992) recognize the following vegetation zones for USAFA (for detailed descriptions of dominant plants for each vegetation zone see Appendix A):

Grasslands
Dominated by mountain muhly (*Muhlenbergia montana*), yellow sedge (*Carex pensylvanica* ssp. *heliophila*), needle-and-thread grass (*Stipa comata*), little bluestem (*Schizachyrium scoparium*), big bluestem (*Andropogon gerardii*), blue grama (*Bouteloua gracilis*), prairie sand reed (*Calamovilfa longifolia*), Parry oatgrass (*Danthonia parryi*), hairy grama grass (*Bouteloua hirsuta*), purple pinegrass (*Calamagrostis purpurascens*), Japanese brome (*Bromus japonicus*), smooth brome (*Bromus inermis*) and cheatgrass (*Bromus tectorum*).

Oak shrubland
Dominated by Gambel's oak (*Quercus gambelii*), piñon pine (*Pinus edulis*), one-seeded juniper (*Juniperus monosperma*), mountain-mahogany (*Cercocarpus montanus*), and Rocky Mountain maple (*Acer glabrum*).
Mixed coniferous forest
Dominated by Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), ponderosa pine (*Pinus ponderosa*), limber pine (*Pinus flexilis*), common juniper (*Juniperus communis* ssp. *alpinus*), Colorado blue spruce (*Picea pungens*), quaking aspen (*Populus tremuloides*), white fir (*Abies*

concolor), bush oceanspray (*Holodiscus dumosus*), and common snowberry (*Symphoricarpos albus*).

Riparian
There are two distinct types within the riparian vegetation zone: tree/shrub and grass/forb. Riparian tree/shrub type is dominated by narrowleaf cottonwood (*Populus angustifolia*), plains cottonwood (*Populus deltoides* ssp. *monilifera*), coyote willow (*Salix exigua*), crack willow (*Salix fragilis*), yellow willow (*Salix lutea*), and peachleaf willow (*Salix amygdaloides*). Riparian grass/forb type is dominated by mannagrass (*Glyceria grandis*), redtop (*Agrostis scabra*), spikerush (*Eleocharis palustris*), sedge (*Carex microptera*), Nebraska sedge (*Carex nebrascensis*), and rush (*Juncus balticus*).

Significant Natural Communities, Plants, Vertebrates, and Invertebrates

The Nature Conservancy (1996) and CNHP (1996) list the following plants, vertebrates, and invertebrates for USAFA. (See Tables 6 and 7 for a complete description of natural heritage imperilment ranks and federal and state codes).

Table 1. Rare or imperiled plants occurring on U.S. Air Force Academy.

Common name	Scientific name	Global rank	State rank	Fed status	State status	Fed sens
carrrion flower	*Smilax lasioneura*	G5	S3S4			
dwarf wild indigo	*Amorpha nana*	G5	S2S3			
New Mexico woodsia	*Woodsia neomexicana*	G4?	S2			
Porter feathergrass	*Ptilagrostis mongholica* ssp. *porteri*	G2T2	S2	(C2)		FS
prairie violet	*Viola pedatifida*	G5	S2			
southern Rocky Mountain cinquefoil	*Potentilla ambigens*	G3	S1S2			

Table 2. Rare or imperiled birds occurring on U.S. Air Force Academy.

Common name	Scientific name	Global rank	State rank	Fed status	State status	Fed sens
Cedar Waxwing	*Bombycilla cedrorum*	G5	S3B,S5N			
Evening Grosbeak	*Coccothraustes vespertinus*	G5	S2S3B,S5N			
Gray Catbird	*Dumetella carolinensis*	G5	S3S4B,SZN			
Ovenbird	*Seiurus aurocapillus*	G5	S2B			
Prairie Falcon	*Falco mexicanus*	G4G5	S3S4B,S4N			
Red-Eyed Vireo	*Vireo olivaceus*	G5	S3B,SZN			

Table 3. Rare or imperiled amphibian occurring on U.S. Air Force Academy.

Common name	Scientific name	Global rank	State rank	Fed status	State status	Fed sens
northern leopard frog	*Rana pipiens*	G5	S3		SC	FS

Table 4. Rare or imperiled mammal occurring on U.S. Air Force Academy.

Common name	Scientific name	Global rank	State rank	Fed status	State status	Fed sens
meadow jumping mouse ssp. preblei	*Zapus hudsonius preblei*	G5T2	S2	LT	SC	FS

Table 5. Rare or imperiled invertebrates occurring on U.S. Air Force Academy.

Common name	Scientific name	Global rank	State rank	Fed status	State status	Fed sens
Moss's elfin	*Callophrys mossii schryveri*	G4T3	S2S3			
sedge darner	*Aeshna juncea*	G5	S3?			
Snow's skipper	*Paratrytone snowi*	G4	S3			

Table 6. Rare or imperiled natural community occurring on U.S. Air Force Academy.

Common name	Scientific name	Global rank	State rank	Fed status	State status	Fed sens
Montane grassland	*Danthonia parryi*	G2?	S2?			

Table 7. Colorado Natural Heritage Program Ranks.

Global rarity ranks are similar, but refer to a species' rarity throughout it range. State and Global ranks are denoted, respectively, with an "S" or a "G" followed by a character. Note that GA and GN are not used and GX means extinct. These ranks should not be interpreted as legal designations.

Rarity Ranks (applied to an element only)

G/S1 Critically imperiled; usually 5 or fewer occurrences in the state; or may be a few remaining individuals; often especially vulnerable to extirpation.

G/S2 Imperiled; usually between 5 and 20 occurrences; or with many individuals in fewer occurrences; often susceptible to becoming endangered.

G/S3 Vulnerable; usually between 20 and 100 occurrences; may have fewer occurrences, but with a large number of individuals in some populations; may be susceptible to large-scale disturbances.

G/S4 Common; usually > 100 occurrences, but may be fewer with many large populations; may be restricted to only a portion of the state; usually not susceptible to immediate threats.

G/S5 Very common; demonstrably secure under present conditions.

G/SU Status uncertain; often because of low search effort or cryptic nature of the element.

T Trinomial; specifies the rank of that species and sub species.

S#B Refers to the breeding season imperilment of elements that are not permanent residents.

S#N Refers to the non-breeding season imperilment of elements that are not permanent residents. Where no consistent location can be discerned for migrants or non-breeding populations, a rank of ZN is used

SZ Migrant whose occurrences are too irregular, transitory, and/or dispersed to be reliable identified, mapped, and protected.

Notes: When a question mark follows a numerical rank (e.g., S2?), it indicates uncertainty about the accuracy of this rank. When two numbers appear in a state or global rank (e.g., S2S3), the actual rank of the elements falls between the two numbers. When a ‘Q’ follows a rank, it indicates uncertainty about the taxonomic status of the element.

Table 8. Federal and State Agency Designations.

Federal Status:

U.S. Fish and Wildlife Service (58 Federal Register 51147, 1993)

LE	Endangered; taxa formally listed as endangered.
LT	Threatened; taxa formally listed as threatened.
P	Proposed E or T; taxa formally proposed for listing as endangered or threatened.
(C1)	FORMERLY: Notice of Review, Category 1: taxa for which substantial biological information exists on file to support proposing to list as endangered or threatened.
(C2)	FORMERLY: Notice of Review, Category 2: taxa for which current information indictes that proposing to list as endangered or threatened is possible, but appropriate or substantial biological information is not on file to support an immediate rulemaking.

U.S. Forest Service (Forest Service Manual 2670.5) (noted by the Forest Service as "S")

FS:	Sensitive: those plant and animal species identified by the Regional Forester for which population viability is a concern as evidenced by:
a.	Significant current or predicted downward trends in population numbers or density.
b.	Significant current or predicted downward trends in habitat capability that would reduce a species' existing distribution.

State Status:
Colorado Division of Wildlife

E	Endangered
T	Threatened
SC	Special Concern

Fire History, Fire Regimes, and Case Studies for Major Vegetation Zones

Prairie Grassland

Fire History

Historically, fires occurred frequently in the tallgrass prairie and were essential in maintaining these grasslands (Daubenmire 1968). Across the Great Plains, lightning-caused fires may have occurred as frequently as every 1 to 6 years (Kucera 1981). Because trees are not present in most prairies, there are no reliable historical records on fire frequency. Wright et al. (1978) believed that the natural fire frequency in prairie grasslands is probably 5 to 10 years. Fires may create monotypes in grasslands by stimulating reproduction of dominate plants and eliminating other species. In other cases, fires may permit invasion by annuals, short-lived perennials, weeds, or aggressive exotics (Lotan et al. 1981).

Gleason (1922) proposed that the adaptation that protects grasses from drought is their ability to die down to underground roots, exposing only dead tops above ground. He noted that the same adaptation that protects grassland plants from drought also affords protection from fire. Grassland fires tend to move rapidly, although soil surface temperatures can vary from 181-1256 degrees F(83-680 degrees C) (Wright 1974b; Rice and Parenti 1978). However, soil is a good

insulator, thus there is little penetration of heat more than a 0.4 inches (1 cm) below the soil surface (Anderson 1982).

Fire can act as a stabilizing or destabilizing factor in vegetation, depending upon the quality and quantity of fuel availability and species composition. Golley and Golley (1972) indicate that grasses produce more biomass than can be decomposed; this excess herbage production is probably a response to grazing. They also noted that productivity of grassland systems declines if this excess biomass is not removed by grazing or periodic fires. Anderson and Brown (1983; 1986) examined the role of fire in maintaining the mosaic of prairie, savannah, and forest occurring on sand deposits along the Illinois River in central Illinois. On this site, fire acted as a factor to maintain sand prairies, savannahs, and open forests, but destabilized closed oak forest. These differential responses to fire are related to the species composition in these varied vegetation zones and to the availability of fuels.

The regional effect of fire on vegetation is influenced by a variety of factors, including precipitation patterns before and after a burn, the composition of the vegetation, topography, and the time of the burn (Daubenmire 1968; Vogl 1974; Bragg 1982b). Drought can interact with fire to influence vegetation in several ways. Drought patterns can determine the amount of fuel available to carry fires, influence the post-burn response of vegetation to burning, and determine when grassland fires are possible. Grasslands will burn any time they are dry including mid-summer, when they support green biomass (Anderson 1972). Also, historical records provide accounts of fires set by Native Americans or lightning during the growing season (Bragg 1982b).

Collins and Wallace (1990) state that there is no single prescription that describes the historical role of fire in grasslands. For example, while the arid shortgrass prairies and semi-desert shrub grasslands were historically subjected to periodic fire (Humphrey 1949; 1958; Cable 1967; Wright 1980), the role of fire in preventing invasion of woody species into these grasslands was complemented by the activities of browsing animals. The recent expansion of trees and shrubs into some grasslands may in part be the result of overgrazing by domestic cattle and the associated reduced competitiveness of the grasses, as well as fire suppression (Humphrey 1958; Wright 1980; Archer et al. 1988). The specific response of vegetation to a fire will vary as a function of species composition, season of the burn, fluctuating climatic cycles, and the complementary actions of other organisms, including grazing and browsing animals (Collins and Wallace 1990).

Fire Regime

In general, grassland fires are of low-intensity because the flames pass quickly, and the soil temperature 1 inch (2.54 cm) below the surface rises very little (Kucera 1981). Plants burned during the spring, when dormant, quickly send up vigorous new growth because of stored carbohydrate reserves in below ground roots and stems. After above ground foliage is consumed by fire, new growth is initiated from rhizomes. The well-developed rhizomes are generally 1 to 2 inches (2.5-5 cm) below the soil surface (Albertson 1937; Weaver 1958). If burned during the summer when plants are actively growing, plants normally survive by initiating new growth from rhizomes; however, regrowth may be slower and less vigorous than in plants burned when dormant (Ewing and Engle 1988).

Fire affects prairie grasslands in two ways: through site modification, especially by altering soil pH and temperature, and through the elimination of invading plants, including selected grasses and forbs and most shrubs and trees. The timing of the fire markedly influences the effects of fire. If burned during a drought when soil moisture is low, or during midsummer when plants are beginning seed production, post-fire recovery is delayed. Consequently, anthropogenic fires are prescribed in the spring and fall (Pyne et al. 1996).

Generally, big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), and mountain muhly (M*uhlenbergia montana*) increase significantly in number and yield following a spring fire; prairie junegrass (*Koeleria macrantha*), sand dropseed (*Sporobolus cryptandrus*), blue grama (*Bouteloua gracilis*), and hairy grama (*Bouteloua hirsuta*) show lesser increase; and buffalograss (*Buchloe dactyloides*) is unaffected (Pyne et al. 1996). Annuals (e.g., cheatgrass (*Bromus tectorum*) tend to proliferate briefly after a burn. Frequent fires actually favor cheatgrass by eliminating competing perennial vegetation. Its seeds survive in the unburned organic material on a site. Rapid growth and vigorous reproduction ensure cheatgrass dominance in the post-burn stand. But even on routinely burned sites, perennials remain fundamental constituents of the plant community and their vigor is maintained as a result of the consumption of the accumulated litter by the fire (Pyne et al. 1996). Introduced species (e.g., cool season grasses) initiate growth earlier in the spring than native species and continue growing later in the fall are selectively removed by properly timed spring and fall burns. For native perennials, fire typically enhances seed production, germination, and seedling establishment (Pyne et al. 1996).

Forb and shrub densities and cover tend to decrease following fires, though the species composition of the forbs remains constant and shrubs are rarely eliminated from the tallgrass prairie. A few forbs, however, do increase as fire scarifies the seedcoat and thereby breaks dormancy: prairie sunflower (*Helianthus petiolaris*), dotted gayfeather (*Liatris punctata*), Missouri goldenrod (*Solidago missouriensis*), false boneset (*Kunhia eupatorioides*), and silky prairie clover (*Dalea villosa*), for example; as do a few shrubs, such as smoothleaf sumac (*Rhus glabra*), lead plant (*Amorpha canescens*), and western snowberry (*Symphoricarpos occidentalis*). In the absence of fire, virtually all shrubs show dramatic growth, and eventually trees may become established. Partly because of repeated anthropogenic disturbances, the data necessary to correlate pre-settlement fire history to ecosystem dynamics are lacking. In broad terms, fire favors grasses and forbs over shrubs and trees, and frequent fire strongly favors native perennials. Where fire has been removed, the distribution of the tallgrass prairie is markedly reduced (Pyne et al. 1996).

In most managed sites fire is prescribed on a 2- to 3-year cycle, though a 5- to 10-year cycle is probably adequate to maintain the ecological integrity of these prairies (Pyne et al. 1996). Large fires often occur during drought years that follow 2 or 3 years of excellent plant growth (Lotan et al. 1981). No attempt at prairie restoration has succeeded without the application of prescribed fire every 1-3 years, timed to favor native perennials over exotic, cool-season grasses. As tallgrass prairies are converted to farms, wood lots, and towns, wildfire is a concern only in those tiny remnants preserved as ecological relicts. Almost exclusively, fire management needs to use prescribed fire for the restoration and maintenance of such preserves (Wright and Bailey 1982).

Case Studies (adapted from A. Banar 1996)
Literature pertaining directly to prairie grasslands in Colorado and the southern Rocky Mountains is sparse. Most research using fire and its effects on grasslands has been done in the central grassland regions of the United States and Canada. Antos et al. (1983) examines the effects of wildfire in a low-elevation grassland. Fescue (*Festuca scabrella, F. idahoensis*) and blue bunch wheatgrass (*Agropyron spicatum*) were the dominate species within the grassland. Anos et al. (1983) indicate that phenology and fire frequency were important factors to consider for prescribed burns.

Clarke et al. (1943) studied the effects of various practices on maintenance and management of range pasture. The researchers include a brief description of results from burning grassland pastures in southern Alberta and southwestern Saskatchewan, Canada. The grassland community consisted of wheatgrass (*Agropyron smithii*), junegrass (*Koeleria macrantha*), needle-and-thread grasses (*Stipa spartea, S. comata*), rough fescue (*Festuca scabrella*), blue grama (*Bouteloua gracilis*), and bluegrass (*Poa secunda*). Researchers found that on ungrazed burned sites, spring burning decreased plant yield by 50% the year of the burn, but that yield returned to pre-burn levels within two years. Grazed burned sites took three to five years to reach pre-burn productivity levels under moderate grazing, longer than the ungrazed sites.

Oak Shrubland

Fire History
Gambel's oak is the dominant overstory species on about 9.3 million acres of rangeland in the southwestern United States (Kuchler 1964). Over 90% of this area lies within the states of Colorado, Utah, and Arizona (Harper et al. 1985). Oak shrubland fires are commonly characterized by frequent, high-intensity fire. Oak shrubland is distinguished by having high annual net productivity and is favorable for fire start every year (Sando 1978). Because of the arid regions in which this type occurs, severe fire weather is not uncommon. The existence of a continuous fuel bed and high winds are important factors in the fire history of oak shrublands. The dominant shrub along the southern Front Range of Colorado, including the USAFA foothills, is Gambel's oak. Gambel's oak is extremely fire tolerant. It is likely that only extremely severe fires with maximum fuel consumption would produce enough heat to kill the buried rhizomes (Harrington 1985).

Mountain-mahogany co-dominates with Gambel's oak on USAFA lands. In open, dry habitats where mountain-mahogany is likely to occur, fires in pre-settlement times were of low severity because of fuel discontinuity. Today, many formerly open stands are dominated by conifers and decadent shrubs which provide greater fuel loads. When fires occur, they are likely to be more severe (Bradley et al. 1991).

Fire Regime
Biswell (1989) recommended burning oak woodlands every 30 years to create a mosaic of young stands. Gambel's oak is seldom killed by fire. Gambel's oak generally sprouts vigorously from stem bases or from underground lignotubers and rhizomes following fire (Brown 1958; Clary and Tiedemann 1986; Neuenschwander n.d.; Reynolds et al. 1970; Vallentine and Schwendiman 1973). Fire promotes root sprouting and the formation of buds on rhizomes (McKell 1950; Reynolds et al. 1970). In a Colorado study, Gambel's oak increased 100 to 150 percent in density and 10 to 40 percent in frequency following a single burn (Harrington 1985). In some locations, particularly in the southern part of its range, Gambel's oak can readily reoccupy a site through seed protected in buried caches of rodents (Neuenschwander n.d.).

Gambel's oak often grows in very dense thickets on steep slopes. Heavy leaf litter accumulation, results in microclimates unfavorable for most native herbaceous species. Fire may have promoted the growth of these thickets since fire tends to burn rapidly and covers larger areas on steep slopes (Brown 1958). Gambel's oak can also serve as a ladder fuel and can contribute to crowning in ponderosa pine forest fires. Repeated summer burning may significantly inhibit growth or eventually even kill Gambel's oak. Harrington (1985) reported that biennial burns may be the best option for fire application in many areas. Kufeld (1983) recommended that "prescribed burning rather than spraying or chaining be used to manage Gambel's oak rangelands for elk, deer, and cattle".

Some researchers have had success in minimizing oak sprouting by planting highly competitive grasses after fire (Harper et al. 1985; Plummer et al. 1970). Species planted successfully include: smooth brome (*Bromus inermis*), meadow brome (*B. erectus*), mountain brome (*B. carinatus*), intermediate wheatgrass (*Thinopyrum intermedium*), crested wheatgrass (*A. cristatum*), orchardgrass (*Dactylis glomerata*), and tall oatgrass (*Arrhenatherum elatus*). However, the majority of these grasses are considered noxious plants and should not be used if native plant diversity is a management objective.

The response of mountain-mahogany to fire may vary seasonally. High- and low-severity fire treatments applied to mountain-mahogany during the dormant season in north-central Colorado were more effective in increasing biomass production than those applied during the growing season (Young and Bailey 1975).

Case Studies (adapted from J.M. Castillo 1996)
The extent and density of Gambel's oak in west-central Colorado has been influenced more by fire than by any other factor (Brown 1958). In a Colorado study, Brown (1958) found that fire stimulates suckering of Gambel's oak resulting in a thickening and merging of stands into continuous thickets. In a Utah burn, sprouts grew rapidly the first two years following the burn, however after 18 years the stand had recovered to only 75% of its original cover. Furthermore, the number of shoots following the burn was very high, however after 18 years they equaled pre-burn densities (Wright 1972). Brown (1958) further found that oak tends to thin-out and retreat in the absence of fire.

A detailed fire scar study in a mahogany chaparral community (mixed with shrub live oak) surrounding a ponderosa pine stand in Arizona was found to have burned on an average

frequency of about 25 years (Dieterich and Hibbert 1988). Gruell et al. (1986) found the average interval to range between 5 and 40 years in the Intermountain west.

Mixed Coniferous Forests

Fire History of Ponderosa Pine

Chronicles from 19th century explorers, scientists, and soldiers described the ponderosa pine forest quite differently than what is seen today. Ponderosa pine forests were open and park-like with abundant grass and forb cover dominating the understory (Biswell et al. 1973). Climate and fire are the factors that had the greatest impact (Harrington and Sackett 1992). Early explorers observed both lightning and Native American ignited fires (Cooper 1960), but fire scar records from the ponderosa pine zone document fire history more precisely. These records authenticate the long-term recurrence of extremely short fire intervals. Based on fire scars from a number of ponderosa pine sites in the southwest, Swetnam (1990) reported a mean fire interval of 2 to 10 years between 1700 and 1900. After the turn of the century, fire intervals increased due to fire suppression efforts and removal of fire-spreading fine fuels by increased grazing. Subsequent large fires appeared to be linked to climatic fluctuations caused largely by El-Nino-Southern Oscillation events (Swetnam 1990).

Interpretation of fire scar data from ponderosa pine stands in the Roosevelt National Forest, Colorado, indicate that prior to 1840, stands had a mean fire interval of 66 years (Laven et al. 1980). Goldblum and Veblen (1992) conducted a study near Boulder, Colorado and found the mean fire interval for pre-settlement averaged 22.0 years. Rowdabaugh's (1978) research in Rocky Mountain National Park indicates a mean fire interval for pre-white settlement to be 38.9 years (Table 8). Veblen articulated in a recent telephone interview (April 3, 1997) that the mean fire interval for Great Plains/ponderosa pine ecotone in Boulder County, Colorado should be every 7 years, at a minimum. However, Merrill Kaufman of the U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, cautions that there is no standard mean fire interval that can be universally applied due to climatic and anthropogenic changes (pers. comm.).

In Wintersteen Park (Laven et al. 1980) and Four Mile Canyon (Goldblum and Veblen 1992) studies, fire frequency increased substantially from the pre-settlement to the settlement periods. But contrary to expectation, mean fire intervals in both areas are not longer during the modern fire suppression era than during the pre-white-settlement era. This may imply that modern fire suppression efforts have not substantially altered the natural fire regime of the study areas. However, this implication is probably equivocal for two reasons. Goldblum and Veblen (1992) state that intense burning during the settlement era probably removed most remnant trees so that the frequency of pre-settlement fires may be underestimated. Secondly, owing to the relatively small numbers of fires, slight changes in the years selected for delimitation of the fire suppression era result in large differences in the computed mean fire interval.

There is a substantial difference in overall mean fire intervals among the three studies conducted in the Colorado montane zone. This could, in part, be explained by the different sizes of the study areas, as smaller units (within a study area) will tend to have longer fire return intervals (Arno and Peterson 1983; Dieterich 1983). The small size (50 ha) of the study area of Laven et al. (1980), compared to 600 ha for the Goldblum and Veblen (1992) may confound direct

comparison of the mean fire intervals. The lower mean fire intervals for Four Mile Canyon may actually reflect more frequent anthropogenic fire. Fourmile Canyon was highly impacted by mining and railroads during the settlement era. In contrast, the study area in Roosevelt National Forest was less impacted by humans (Goldblum and Veblen 1992). In an ongoing research project Merril Kaufmann has collected fire scar data for 145 ponderosa pines at Cheesman Lake, southwest of Denver, Colorado. The Cheesman Lake sites historically experienced a series of large, moderate to high-intensity fires, with historical mean fire intervals between 20 to 120 years (pers. comm.).

Table 9. Fire History for ponderosa pine/Douglas-fir forests in the Colorado Front Range.

	Mean Fire Interval (years)
Wintersteen Park, Roosevelt National Forest, CO (Laven et al. 1980)	
Entire chronology (1708-1973)	45.8
By period	
Pre-white settlement (pre-1840) n=6	66.0
Settlement era (1840-1905) n=12	17.8
Suppression era (post 1905) n=3	27.3
Rocky Mountain National Park, CO (Rowdabaugh 1978)	
Entire chronology (n.a.)	37.9
By period	
Pre-white settlement (pre 1840)	38.9
Four Mile Canyon, Boulder, CO (Goldblum and Veblen 1992)	
Entire chronology (1721-1988) n=72	15.2
By period	
Pre-white settlement (pre 1840)	22.0
Settlement era (1840-1905)	7.4
Suppression era (post 1905)	10.3
Cheesman Lake (Kaufmann pers. comm.)	
Entire chronology (1197-1963) n=145	20 to 120

(adapted from Goldblum and Veblen 1992)

A historical perspective is crucial for understanding how forested ecosystems have responded to forcing factors of climate and fire variability (Brown 1997). A principle component of the emerging paradigm of ecosystem management for U.S. Forest Service lands is the need for historical reference conditions to understand how ecosystem patterns and processes have evolved through time (Kaufmann et al. 1994). Peter Brown (pers. comm.) is currently conducting research along the Front Range to determine regional scale patterns of fire or in other words, what drives fire across the landscape, in particular the climate. Data from this study will not only be central to defining reference conditions for fire regimes of lower elevation forests in the Front Range, but in understanding how local fire and stand histories compare to larger regional patterns of fire regimes and stand establishment. Fire history data also provide site-specific information, such as past fire frequency, timing, and seasonality, that is necessary baseline information for development or refinement of prescribed fire plans and other management programs (Brown 1997). Specifically, managers need to compare precipitation amounts with tree recruitment to reconstruct the climate and therefore determine the historical fire regime of a

site. Prescribed fires are needed to reduce the risk of catastrophic crown fires in ponderosa pine forests that are today often outside the range of historic variability in structure or density (e.g., Covington and Moore 1994).

Fire Regime of Ponderosa Pines

Since the beginning of this century, fires have been excluded almost completely, resulting in new age groups of trees and considerable amounts of dead material in stands that otherwise might have had the classic, park-like appearance with numerous grassy openings characteristic of pre-settlement times. Covington and Sackett (1986) identified multiple management problems associated with reduced fire frequencies in ponderosa pine stands including: overstocked sapling patches, reduced growth, stagnated nutrient cycles, increased disease, insect infestations, parasites, decreased seedling establishment, increased fuel loading, and increased severity and destructive potential of wildfires. Fire suppression has resulted in much higher numbers of trees per area and loss of most openings. And when fires occur, they often are of such intensity that the entire plant community is replaced (Kaufmann et al. 1992). Many ponderosa pine stands typical of this regime experience low- to moderate-intensity surface fires that result in mosaics of small patches of fire-killed trees interspersed with broad areas of seed-producing survivors (Heinselman 1981). Fire creates a favorable seedbed for ponderosa pine by exposing bare mineral soil and removing competing vegetation. However, post-burn establishment is successful only when a good seed crop coincides with above average rainfall (Fowells 1965). For perhaps five years, ponderosa pine seedlings must compete vigorously with grasses and are quite vulnerable to fire. Thereafter, the trees develop a thick bark, shed lower branches, and deposit a layer of needles that can suppress the growth of the grasses. These changes make the trees less susceptible to fire and alter the fuel complex and the subsequent fire behavior characteristics (Pyne et al. 1996).

The intent of fire management in ponderosa pine forests is to substitute prescribed fire for wildfire wherever possible. Ponderosa pine depends on frequent surface fires to maintain stand health and stability (Biswell et al. 1973; Cooper 1960). Consequently, ponderosa pine communities have evolved flammable properties to encourage recurrent, low-intensity burning (Mutch 1970). The resinous needles provide an abundant, annual source of highly flammable fuel, with yearly accumulations in dense stands exceeding 3,500 pounds per acre (3,120 kg/ha) (Biswell 1973). Despite such characteristics, fire frequencies for ponderosa pine under natural fire regimes vary greatly according to site conditions and geographical area.

Timing is critical in the mature ponderosa pine forest; too early in the fuel cycle and there will be insufficient fuel to carry an effective fire; too late and fire intensity may result in high tree mortality. In an arid region where good seed years are unpredictable and regeneration is difficult, fires with high tree mortality are not desirable. Yet in high fire danger areas, some fuel management is essential, and prescribed fire on a cycle of 5-10 years is an effective solution (Pyne et al. 1996; Peet 1988). Brown (1997) states that patterning in ponderosa pine fire regime parameters of frequency, spatial extent, severity, and seasonality occurs at multiple scales through time and across space owing to differences in scaling of driving variables. Local vegetation types and topography control to a large extent the occurrence and spread of individual fires by influencing fuel continuities and loadings. Annual weather patterns control fuel conditions and fire ignitions on seasonal time scales (Brown 1997). Merrill Kaufman (pers.

comm.) concurred stating that regional and local climatic conditions primarily control fuel availability.

Case Studies of Ponderosa Pine (adapted from S. Joy 1996a)
Weaver (1951) reported production of grasses and herbaceous plants increased in ponderosa pine forests after fire, the result of reduced competition and litter, and increased nutrient availability. The growth of species beneficial to the soil, such as algae, fungi and bacteria, was also encouraged. Following light surface fires, seedlings become established readily in exposed mineral soil (Pearson 1950). Frequent surface fires result in some seedling mortality; however, once trees reach the sapling stage, they have a high probability of maturing (Cooper 1960).

Where fire suppression has occurred, response to fire will vary with environmental conditions, fuel build-up, and timing of the fire. In general, the longer the post-fire interval, the greater the potential for stand-replacing fire due to high fuel accumulations (Biswell et al. 1973). In Colorado, tree mortality resulting from a controlled burn was found to be most strongly correlated with high crown damage and small tree size (Wyant 1981). In response to fire, surviving trees showed increased bud size and fascicle length compared to unburned areas, enhancing tree growth for at least two growing seasons after the burn (Wyant 1981). Ponderosa pine was able to recover from crown damage at greater levels than were similarly sized Douglas-fir trees (Wyant 1981).

Fire History of Mixed Coniferous Forests
Natural fire frequency varies widely in the Douglas-fir zone. Dry Douglas-fir habitat types in the northern Rocky Mountains experienced low- to moderate-intensity ground fires at less than 30 year intervals (Arno 1980; Pfister et al. 1977). These frequent ground fires maintained relatively open stands of Douglas-fir or, more frequently, seral stands of ponderosa pine since pine saplings are more fire resistant than Douglas-fir saplings (Arno 1980; Fischer and Bradley 1987; Loope and Gruell 1973). Fire suppression has resulted in long fire-free periods which have allowed Douglas-fir regeneration to become well-established. In some areas, dense thickets have formed, which provide a continuous fuel ladder to the crown of overstory trees. Thus, fire suppression has increased the potential for severe, stand-destroying wildfires. Where ponderosa pine is a major associate, fire frequencies of about 10 years (Arno 1976) are common. However, in the cooler sites, 10 to 30 year frequencies are more likely. It should be recognized that fires originating in the lower and drier sites frequently burn into these more mesic areas. Fire damage in this zone is likely a function of the length of time since the last fire; the longer the period, the greater the probabilities of conflagration (Lotan et al. 1981).

Conflagrations appear to be the general rule in the spruce-fir type (Brown 1975). Although no extensive data are available, severe fires most likely occur every 100 to 500 years (Brown 1975). However, recent evidence indicates that low ground fires do occur at 30- to 40-year intervals in some areas (Arno 1976; Gabriel 1976). Perhaps the most common reason for the conflagrations is the combination of a hot, dry season and an extensive fuel accumulation either due to insects, (Amman 1975) and disease (Hawksworth 1975), or due to slash from logging. These fuels are often ignited by fires that originate in the lower drier forests. Even severe fires do not destroy all trees in the spruce-fir forests but usually leave "islands" that can serve as a seed reservoir for restocking the area. True firs, Douglas-fir, and spruce are generally good seed producers and

disperse their seeds for considerable distances, with the light-seeded spruce having a distance advantage (Lotan et al. 1981).

Fire Regime of Douglas-fir, white fir, Colorado blue spruce, and aspen
Coniferous forests in southern Rocky Mountains are characterized by frequent, low- to moderate-intensity surface fires. This vegetation type is typified by numerous ignition sources combined with recurring weather patterns that encourage fire spread (Sando 1978). Ecosystems in this regime have adapted to frequent fire by having fire-resistant characteristics such as thick bark or high crowns (Sando 1978).

Swetnam and Baisan (1994) hypothesize that both fuel production (especially grasses and pine needles) and fire fuel moisture are important climate-linked factors in ponderosa pine fire regimes, while fuel moisture is the primary factor controlling mixed conifer fire regimes. The canopy cover in mixed conifer is greater than in ponderosa pine forests, allowing the snow pack to persist longer into spring. Moreover, the shaded conditions limit the development of grass cover, and the short needles of Douglas-fir and true firs tend to compact quickly on the forest floor. This results in a fuel substrate that is less conducive to fire spread than in the grassy understory and loose litter layer of long needles found in ponderosa pine forests. Needle retention is also longer in both Douglas-firs and true firs (5-7 years or longer) than in ponderosa pines (2-3 years). The combination of these micro-environmental and fuel characteristics result in mixed conifer fire regimes that were unresponsive to previous year's moisture levels and associated fuel productivity, and fires primarily occurred when conditions were very dry (Swetnam and Baisan 1994).

In ponderosa pine and mixed conifer forests, precipitation was significantly reduced in the winter-spring period immediately prior to fire occurrence. In addition, winter-spring precipitation during the second year preceding major fire years in the ponderosa pine forest was significantly increased (Touchan et al. 1994). Variablility of fire documented in both mixed conifer forest and ponderosa pine forests was caused by 3 main factors: 1) anthropogenic stresses such as intense livestock grazing, 2) reduced fine fuels necessary for the spread of fire in the high frequency fire regimes, and 3) fire suppression. These factors allowed changes in species compositions and stand structure in both the ponderosa pine and mixed conifer forests. The build up of woody vegetation also contributed to the decline of native grasses, due to the increased shading and accumulation of thick mats of pine needle litter (Touchan et al. 1994)

Douglas-fir
Mature Douglas-fir has a high resistance to fire damage. However, saplings and small poles are sensitive to surface fires because of their thin bark, resin blisters, closely spaced needles, and thin twigs and bud scales. The low, dense branching habit of saplings and poles allows surface fires to carry into the crown. Older trees develop a thick, corky bark that protects the cambium against low to moderate-intensity fires (Wright and Bailey 1982). Douglas-fir foliage is considered to be highly flammable so even mature trees with branches extending the length of the bole (tree trunk) are susceptible to "torching" into the crowns (Crane and Fischer 1986). As with ponderosa pine, heavy fuel accumulations at the base of the tree increase the opportunity for fire injury (Zwolinski 1996). The shorter needles of many of the mixed conifer species, including Douglas-fir, results in a compact, low porosity fuelbed. Slow moving surface fires

with high residence times can remove the protective organic layer and damage shallow lateral roots.

Mature trees can survive moderately severe ground fires because the lower bole is covered by thick, corky bark that insulates the cambium from heat damage (A. D. Revill Associates 1978; Fischer and Bradley 1987). It takes about 40 years for trees to develop fire-resistant bark on moist sites in the northern Rocky Mountains (Fischer and Bradley 1987). Protection offered by thick bark is often offset by low growing branches and flammable foliage that make trees susceptible to crowning (Fischer and Bradley 1987; Lotan et al. 1981).

Douglas-fir regenerates on burned sites by wind-dispersed seeds. Fires will reduce fuel loadings and expose mineral soil allowing establishment of the shallow roots of seedlings. For best establishment, Douglas-fir needs minimum competition and some shade (Ryker 1975). Severely burned sites on south-facing slopes may be more favorable for ponderosa pine regeneration than Douglas-fir because of the warmer, drier conditions.

Overall, Douglas-fir is more fire resistant than spruces and true firs and equally or slightly less fire resistant than ponderosa pine. Fire management for Douglas-fir forests is complex, and prescribed fire is somewhat limited in scope. The variety of large wildfires can occur at any time in the developmental history of these ecosystems, and unlike the case with pure ponderosa pine forests, the timing of prescribed fire must relate to the life histories of more than one species. Large wildfires typically correspond to episodes of drought rather than simple patterns of fuel history. Prescribed burning is widely practiced for hazard reduction and site preparation following logging, and to some degree within programs of prescribed natural fire in wilderness areas. Broadcast burning is practiced, but not with the frequency typical of ponderosa pine forests.

White fir

Sapling and pole-sized white firs are fire sensitive. At this size trees have a smooth, thin, resinous bark and low-growing branches that can be easily ignited by surface fires. As trees mature the bark thickens and some self-pruning of lower branches occurs resulting in increased fire resistance. Shallow roots make white fir more susceptible to soil heating and root damage (Zwolinski 1996).

In mixed conifer forest, a natural fire regime of frequent, low-intensity surface fires prevents white fir from achieving dominance since it is less fire tolerant than associated species (Weaver 1951). This maintains the white fir habitat in a mid-successional stage where ponderosa pine or Douglas-fir dominate the overstory and white fir exists in the understory. White fir will eventually become dominant on the site if the fire interval is long enough to allow trees to reach a fire-resistant size (Wright and Bailey 1982).

Colorado blue spruce

Colorado blue spruce is very sensitive to fire and is generally killed with low-intensity burns (Zwolinski 1996). Post-fire regeneration via wind-dispersed seeds readily occurs on fire-prepared seedbeds (Fischer and Bradley 1987).

In riparian areas where blue spruce occurs, intervals between fires are about 350 to 400 years. Severe fires occur infrequently, and succession back to the original community is often relatively rapid (15 to 35 years). Depending on the site, blue spruce may be the dominant seral tree (Crane 1982). Successive fires may prevent blue spruce from dominance because it is fire intolerant. Historical fire frequency in mixed conifer forests was about 22 years, based on fire-scarred trees in the White Mountains of Arizona (Dieterich 1983). Fire suppression during the past 100 years has made the mixed conifer forest in which blue spruce occurs more susceptible to fire; however, blue spruce may be dominant in some areas because of the longer fire-free intervals.

The high susceptibility of blue spruce to fire damage is mitigated somewhat by the moist and cool sites where it grows (Crane and Fischer 1986). Pockets of blue spruce can escape fire if they occur in wet locations where fire spread is hampered.

Aspen

Aspen killed by fire will respond by vigorous root suckering and quickly form the dominant post-burn species on many sites (Zwolinski 1996). Fire removes apical dominance and, with a darker, warmer soil surface, stimulates roots to make stored food available for sucker generation. Moderate-intensity fires appear most effective in promoting suckering (Bartos and Mueggler 1981). Aspen appears to be less susceptible to injury when burned or damaged during winter dormancy. Young sapling size or smaller aspen can also regenerate through root crown and stump sprouting. In parts of the southern Rocky Mountains, aspen can survive in a suppressed state as an understory species in coniferous stands for relatively long periods of time, and can readily colonize a burned site through root suckering (Parker and Parker 1983). In some instances light surface fires can apparently retard conifer regeneration while stimulating aspen suckering (Parker and Parker 1983).

Many believe that the majority of aspen communities probably only burned naturally under extreme fire conditons (Brown and DeByle 1987). Mixed stands made up of aspen and conifers probably burned at more frequent intervals, with fire frequencies increasing as conifers replaced the aspen through natural succession (Brown and DeByle 1987). The natural stand replacement fire interval in many western aspen-mixed conifer or spruce-fir communities was approximately 70 to 200 years (Covington et al. 1983). Low-intensity fires may have occurred at 2 to 5 year intervals in some western, lower-elevation aspen-bunchgrass communities (Covington et al. 1983). Current fire frequency in many western stands is equivalent to only one fire in every 5,000 years (Brown 1985a). Research indicates that fire frequencies of 100 to 300 years are necessary for the regeneration and maintenance of many aspen communities (DeByle et al. 1987). Many references have been made to the comparative lack of fire in aspen communities. Some researchers believe that most of the flammable mixed conifer-aspen stands which burned in the late 1800s may only now be reaching critical fuel levels (Brown 1985a, DeByle et al. 1987).

Aspen has often been referred to as a fire-break species. Fires in aspen are generally infrequent, slow-spreading, and of relatively low-intensity (Bevins 1984, DeByle et al. 1987). Fires which burn thousands of acres of surrounding forest often burn less than a few yards into adjacent aspen communities (Jones and DeByle 1985). Crown fires in coniferous forests have reportedly dropped to the ground upon reaching aspen, and have subsequently been extinguished (Jones and

DeByle 1985e). Many factors influence the flammability of aspen communities including: slope, the amount of downed woody material present, grazing history, fuel moisture, fuel loading and distribution, weather, and aspen crown closure.

Riparian

Fire History

Prior to 1900, many of the riparian areas associated with the ponderosa pine zone experienced low-intensity fires at a rate of 2-5 per century (McCune 1983; Arno and Petersen 1983). These fires burned in a mosaic pattern leaving much of the vegetation and soil only lightly disturbed, and helped maintain a diversity of plant species far exceeding that found in adjacent upland forests. Riparian communities embedded in the semiarid ponderosa pine zone were historically dominated by relatively open stands of very large ponderosa pine that survived the low- to moderate-intensity fires (Arno 1986).

Presently, many of the disturbance dependent species (e.g., serviceberry, chokecherry, elderberry, and Rocky Mountain maple) are being replaced by dense understories and thickets of shade-tolerant trees. The overstory trees are often dead or dying in fire suppressed area and there is a buildup of downed fuels along with a dense conifer understory. These conditions allow modern wildfires to sweep through the entire streamside forest in a high-intensity burn, leaving little vegetation to protect streambanks and water quality (Arno 1986). Storms can readily degrade stream quality after high-intensity wildfires, which are now common in these ponderosa pine zone riparian areas (White 1995).

Fire Regime

Fire intensity, magnitude, and behavior vary with the composition, density, and structure of local vegetation, litter depth, soil composition, water table, and climate in riparian forests (Rassman 1993). Fire behavior varies from lightly-charring, slow-burning surface fires (Rassman 1993) to rapid, stand-destroying crown fires (Berndt 1971; Minshall et al. 1989) depending on the combination of these variables. Fuel loading is therefore highly variable and site specific. Wind, low humidty and hot air temperatures also affect fire behavior and extent.

Few data are available on the natural frequency of fire in riparian ecosystems; however, one may speculate that fire intervals were highly variable and depended on site-specific fuels and conditions. Riparian-initiated fires were presumably uncommon due to the high moisture content of riparian soils and vegetation, and the low frequency of lightning strikes in low-lying drainages and valleys bottoms where riparian areas occur (Joy 1996b).

Cottonwoods

Root sprouting occurs on all *Populus* species (Schier and Campbell 1976). Narrowleaf cottonwood will sprout after light- to moderate-intensity fires (Hansen et al. 1989; Fowells 1965). The morphologically similar eastern cottonwood and balsam poplar develop fire-resistant bark after 15 to 20 years of age (Collingwood 1937). It is likely that narrowleaf cottonwood does the same. However, young balsam poplars are susceptible to fire (Haeussler and Coates 1986), and young narrowleaf cottonwoods are probably also susceptible.

Seedling regeneration is favored following disturbances such as fire (Fowells 1965; Gruell 1980a). Fire thins the overstory, allowing more light penetration, and exposes the mineral soil such that seeds are able to establish if soil moisture is adequate (Fowells 1965). The bark of older cottonwoods can be up to 4 inches (10 cm) thick at the base, affording fire protection (Fowells 1965). Trees less than 20 years old are susceptible to fire (Collingwood 1937) but may resprout. Plains cottonwood (var. *occidentalis*) is able to produce sprouts from the rootcrown and the stump after fire (Dickman and Stuart 1983; Severson and Boldt 1977).

Willows

Sandbar or coyote willow (*Salix exigua*) sprouts from its roots after fire (Conrad 1987; Rowe and Scotter 1973; Zasada 1986). Its numerous wind-dispersed seeds are also important in revegetating burned areas (Rowe and Scotter 1973). The high soil and fuel moisture content characteristic of its streamside habitat reduces the chance of fire ignition and spread. Peachleaf willow (*Salix amygdaloides*) also sprouts from its roots following fire (Hansen et al. 1988). The high soil and fuel moisture content characteristic of its streamside habitat reduces the chance of fire ignition and spread. Its numerous wind-dispersed seeds are also important in revegetating areas following fire (Zasada 1986).

Case Studies (adapted from S.M. Joy 1996b)

Few studies have focused on fire-riparian relationships. Those that do are carried out opportunistically (Berndt 1971; Minshall et al. 1989; Bozek and Young 1994) due to the unpredictable and often uncontrollable nature of fire.

Accounts of lightning-ignited fires that spread from adjacent upland areas to riparian areas appear to dominate the literature. For example, in central Washington a lightning storm initiated a fire that "devastated" 115,000 acres ponderosa pine-Douglas-fir forest, including the vegetation surrounding several tributary streams (Berndt 1971). Similarly, Albin (1979) reported on the effects of a lightning-started fire near yellowstone Lake in Yellowstone National Park, Wyoming, that spread to several tributaries of the lake, and some lodgepole pine were "completely killed" (Albin 1979). Aspen-dominated riparian areas are believed to burn only when invaded with flammable conifer species that provide continuous fuel (DeByle et al. 1987).

Summary of Fire History of the Front Range

As stated in the previous section, detailed information on the history of fire affecting the vegetation of the Front Range is lacking (Veblen and Lorenz 1991). However, the pervasive influence of fire on the forests of the Front Range is reflected by the ubiquitous presence of charcoal beneath forests over the entire elevational gradient (Peet 1981). The available studies are not adequate for fully quantifying fire history in the Front Range; however, they do identify some consistent trends in mean fire return intervals (Veblen and Lorenz 1991) (see Table 9).

Average fire intensity varies among the vegetation zones of the Front Range. Most fires occurring in the open ponderosa pine woodlands are surface fires carried mainly by grass fuels and are unable to develop into crown fires due to the low density of trees and the lack of lower limbs on ponderosa pines (Veblen and Lorenz 1991). At higher elevations, in denser stands of ponderosa pine and particularly in mixed stands with Douglas fir, fires are more likely to become crown fires and are very patchy in spatial patterns as a result of the heterogeneous site and fuel conditions. Throughout the montane zone, most fires are light surface fires, but the infrequent crown fires have had a major impact on the landscape (Rowdabaugh 1978). A pattern appears of low-intensity, frequent fires at low elevations and high-intensity, infrequent fires at higher elevations. Variation in fire frequencies related to white settlement follows a consistent pattern in the Front Range (Rowdabaugh 1978; Laven et al. 1980; Skinner and Laven 1983). During the settlement period from the mid-nineteenth century to about 1915 there was a dramatic increase in fire frequency, attributed to the intentional burning by prospectors. For the montane zone, fire frequency increased several-fold so that mean fire-return interval decreased to less than 20 years (Veblen and Lorenz 1991).

Veblen and Lorenz (1991) indicate the importance of fire prior to c. 1920. In their collection of historical photographs most of the montane zone and a large part of the subalpine zone had been burned during the latter part of the nineteenth century. Thus, stands of dead standing, charred trees are commonly depicted. Most fires were started by white settlers beginning about 1860. Earlier, frequent fires were started by lightening or by Native Americans, but fire history studies show a substantial increase in fire frequency during the settlement period (Rowdabaugh 1978; Laven et al. 1980).

Following the massive burning of montane forests during Euro-American settlement period, fire frequency declined dramatically. Much of this decline is due to modern fire suppression efforts, but it is also partially due to the change in forest structure resulting from the nineteenth-century fires (Veblen and Lorenz 1991). Fire suppression during the past half century has resulted in dense populations of even-aged stands of ponderosa pine and Douglas-fir, accumulation of fuel, and the abundance of small trees that serve as fire ladders. The result is an increased likelihood of a stand-devastating crown fire.

Table 10. Summary of historical fire regimes of major vegetation zones on U.S. Air Force Academy.

Historical Fire Regimes (prior to 1900)			
Vegetation Type	**Fire Intensity**	**Fire Frequency**	**Location of Research Area**
Prairie grassland	low	1 every 1-6 yrs (Kucera 1981)	Central Great Plains
		1 every 5-10 yrs (Wright et al. 1978)	Central Great Plains
Oak Shrubland	high	1 yr (Sando 1978)	Rocky Mountains
Mountain-mahogany	moderate-high	1 every 25 yrs (Dieterich and Hibbert 1988) 1 every 5-40 yrs (Gruell et al. 1986)	Colorado Unknown
Mixed Coniferous			
Ponderosa pine	low	1 every 2-10 yrs (Swetnam 1990)	southwest U.S.
		1 every 66 yrs (Laven et al. 1980)	Colorado
		1 every 22 yrs (Goldblum and Veblem 1992)	Colorado
		1 every 5-12 yrs (Peet 1988)	northern Rocky Mountains
		1 every 38.9 yrs (Rowdabaugh 1978)	Colorado
Douglas-fir	low to moderate	1 every 30 yrs or less (Arno 1980; Pfister et al. 1977)	northern Rocky Mountains; Montana
Douglas-fir with ponderosa pine	low	1 every 10 yrs (Arno 1976)	Bitterroot N.F., Montana
Colorado blue spruce (dry site)	moderate to severe	1 every 22 yrs (Dieterich 1983)	southwest U.S.
Colorado blue spruce (wet site)	moderate	1 every 350-400 yrs (Crane 1982)	northern Rocky Mountains
Spruce-fir	moderate	1 every 100-500 yrs (Brown 1975) 1 every 30-40 yrs (Arno 1976; Gabriel 1976)	Washington Bitterroot N.F., Montana; Montana
Aspen with bunchgrass	low	1 every 2-5 yrs (Covington et al. 1983)	Unknown
Aspen	moderate	1 every 70-200 yrs (Covington et al. 1983)	Unknown
Riparian associated with ponderosa pine	low	2-5 fires every 100 yrs (McCune 1983; Arno and Peterson 1983)	Montana; Bitterroot N.F., Montana

Fire Effects on Noxious Plants

Japanese brome (*Bromus japonicus*)

Except in wet years, fire tends to reduce Japanese brome populations (Gartner et al. 1986). However, the reduction usually lasts for only 1 or 2 years (Gartner et al. 1986; U.S. Department of Agriculture, Soil Conservation Service 1982). Some seed is killed by fire, but seedbank

reserves, reproductive capacity, and competitive ability of Japanese brome are usually sufficient to allow for repopulation of an area within 2 years unless the site is reburned (Whisenant 1985; Whisenant and Bulsiewicz 1986). Studies conducted when precipitation was below normal reported reductions in Japanese brome populations for 2 post-fire years (Gartner et al. 1978; Gartner and White 1986). Since litter accumulations are more critical for germination and seedling establishment when precipitation is low, drastic population reductions can be expected when burning is followed by below-average precipitation (Whisenant 1990). Fire during wet years may not reduce Japanese brome populations. Studies conducted during years of high precipitation showed no change in Japanese brome density the summer after burning (Whisenant et al. 1984). Gartner et al. (1978) recommended burning Japanese brome in the ripe seed stage in order to maximize kill of seeds in panicles.

Kirsch and Kruse (1973) hypothesized that the successful establishment and spread of Japanese brome across the Northern Great Plains is a direct result of fire suppression. The resulting thick surface mulch created a more mesic microclimate for seeds and seedlings (Kirsch and Kruse 1973; Whisenant and Bulsiewicz 1986). Japanese brome populations will probably continue to increase in the absence of fire (Whisenant 1990). In the Flint Hills of Kansas, for example, bluestem (*Andropogon gerardii* and *Schizachyrium scoparium*) prairie grazed and burned annually has remained in excellent condition, while prairie grazed but not burned has been invaded by Japanese brome and Kentucky bluegrass (*Poa pratensis*) (Anderson 1965).

Whisenant (1985) stated that fire exclusion in northern mixed-grass prairie has improved conditions for Japanese brome establishment at the expense of native grasses. In the absence of intensive grazing, litter accumulations in northern mixed-grass prairie stabilize after 5 to 6 post-fire years (Abouguendia and Whitman 1979; Dix 1960). Whisenent (1990) has recommended burning every 5 years or less to reduce litter accumulations. This reduces Japanese brome populations, particularly when fall precipitation is low. However, he cautions managers to balance the benefits of litter against need to reduce Japanese brome when preparing fire management plans. Benefits of litter include soil stabilization and insulation, moisture retention, and promotion of climax perennials (Vogl 1974).

Smooth brome (*Bromus inermis*)

Smooth brome is fairly tolerant of fire during dormancy and early spring (Wasser 1982). Susceptibility increases once growth is initiated and the seed has germinated (U.S. Department of the Interior, Bureau of Land Management n.d.). Smooth brome reproduces primarily by rhizomes, which enhances the ability of plants to survive fire (Vogl 1974). The rhizomes of most grasses are usually 1 inch (2.5 cm) below the soil surface (Wright and Bailey 1980). In grasslands where the duration of fire and the resulting soil surface temperatures are usually minimal, this depth is sufficient to protect the primordial regions of the roots. In grasslands, the seeds of most plants survive fire (Daubenmire 1968; Wright and Bailey 1980).

Most information suggests that smooth brome is damaged by the heat of fire (Old 1969; U.S. Department of the Interior, Bureau of Land Management n.d.). Smooth brome is a cool-season grass and thus is most susceptible to fire damage during the spring and early summer when it is

actively growing. The susceptibility of plants to heat damage increases as plant moisture increases (Wright and Bailey 1980).

Fire stimulates the initiation of new shoots from the rhizomes (Gartner et al. 1978). Kirsch and Kruse (1973) reported that burning in May reduced the cover of smooth brome in the midgrass prairie of east-central North Dakota, and categorized it as a decreaser in response to fire. However, smooth brome recovered after spring burning in an Illinois prairie. It was less damaged than Kentucky bluegrass (*Poa pratensis*) because it began growth later in the growing season (Old 1969).

Cheatgrass *(Bromus tectorum*)

Cheatgrass is an annual grass and is able to complete its life cycle in the spring before the summer dry weather begins. Cheatgrass is well adapted to frequent fire and often dominates plant communities after fire (Young et al. 1969). Melgoza and Nowak (1991) concluded that after fire, cheatgrass reduces root length densities of native species (e.g., rabbitbrush and needle-and-thread grass). The success of cheatgrass after fire has often been attributed to its ability to rapidly occupy open spaces created by the removal of fire-intolerant plants (Stewart and Hull 1949; Klemmedson and Smith 1964; Young et al. 1969; Young and Evans 1973; Thill et al. 1984). Cheatgrass is a highly flammable species due to its complete summer drying, its fine structure, and its tendency to accumulate litter (Klemmedson and Smith 1964; Tisdale and Hironaka 1981). Fire reduces cured plants to ash, but fire intensity may not be great enough to consume the litter layer, even if associated shrubs burn (Young et al. 1976).

Because of its flammability, cheatgrass greatly increases the fire hazard on a site. The rate of spread, size, and frequency of fire all increase. In Oregon, cheatgrass ranges were found to be 500 times more likely to burn than non-cheatgrass ranges. A forest stand with a cheatgrass understory may suffer loss of regeneration because of frequent burning. Cheatgrass fires spread very rapidly and may extend into nearby stands of native vegetation and reduce the cover of valuable perennial species (Stewart and Hull 1949).

Besides increasing fire frequency, the length of time cheatgrass remains a hazard is longer than that for perennial grasses. Cheatgrass dries 4 to 6 weeks earlier than perennials and is susceptible to fire 1 to 2 months longer in the fall (Stewart and Hull 1949). Several growth and habitat characteristics of cheatgrass make it a fire hazard: 1) it produces large quantities of seed that usually develop into dense stands; 2) it can provide a continuous fuel between grassland and forest stands; 3) it grows in the 6 to 22 inch precipitation zone, an area with severe fire weather; 4) it cures early in the fire season; 5) its finely divided stems and flowering stem ignite readily when dry and; 6) it responds easily to any change in moisture conditions because of its structure.

There is a correlation between plant color and moisture status during the curing process. Cheatgrass passes from green to a purple hue to a straw color as it dries. The relationship is as follows:

Plant Color	Moisture Content (%)
green	100
purple	30-100
straw	< 30

The onset of purple coloring should be taken as a warning that hazardous fire conditions will develop within about 2 weeks. Observation of coloration should be done close-up. A stand may appear to be purple when most plant parts are still fairly green. In Montana, the average time required to change from purple to straw color (100 to 30 percent moisture) was 14 days (Mutch 1967).

Burning cheatgrass may reduce the next spring's production. On the Snake River Plain near Dubois, Idaho, fall-burned areas produced from 1/7 to 1/50 as much as unburned sites. The rate of spring growth was also retarded. Early in the season, plants were half as large as those from unburned areas. The difference became less evident as plants matured (Pechanec and Hull 1945). Early summer fires produce similar results. Plant numbers may be reduced, but those that do develop are often larger and produce great quantities of seed. The earlier the fire, the greater the degree of reduction (Stewart and Hull 1949). Fires in pure cheatgrass stands tend to be less common in the spring or early summer (Tisdale and Hironaka 1981). Fires generally occur in the summer after seed is shed and is less vulnerable to burning. Reduction of cheatgrass under these conditions is not great (Tisdale and Hironaka 1981). Cheatgrass competes with native species for soil water and negatively affects their water status and productivity. This competitive ability of cheatgrass greatly enhances its capability to exploit soil resources after fire and to enhance its status in the community (Melgoza et al. 1990).

Prescribed burning is currently being implemented at Phantom Canyon, a Nature Conservancy Preserve located in the Laramie Foothills, north of Fort Collins, Colorado. The experimental design includes burning twice a year; a very hot, high-intensity burn in the spring to kill mature plants, and a cool (with high humidity), low-intensity burn in the fall to remove the cheatgrass seeds in the litter layer and to promote warm season grasses (Heather Knight, manager, pers. comm.).

After a midsummer fire in northern Nevada, cheatgrass density was reduced, but individual plants were tremendously productive. Tiller and seed production both were enhanced by burning. On an unburned control plot, the maximum number of seeds on a plant was 250. On the burned plot, the minimum was 960 (Young and Evans 1978). In a northern Nevada study, cheatgrass seed was reduced approximately 96 to 99 percent, from 5,000 to 8,000 seeds per square meter to 20 to 300 seeds per square meter (Young et al. 1976). The amount of litter or ash left on a site is a good indicator of the amount of cheatgrass seed still surviving. Seed is concentrated in the litter, especially around shrubs. Since cheatgrass produces prolific quantities of seed, even a large reduction in the seed pool will not prevent it from regaining dominance on a site.

Canada thistle (*Cirsium arvense*)

Fire top-kills Canada thistle. After top-kill, plants resume growth from perennating buds located on the roots (Thompson and Shey 1989; Young 1986). Total herbage production was unaffected following winter and spring prescribed fires in Oregon. Although there were fewer mature plants, the high density of new vegetative shoots compensated for the loss in herbage production (Young 1986). Patches of Canada thistle were reduced in Minnesota after 4 years of consecutive spring burning of low- to moderate-intensity (Becker 1989). Density and above ground biomass were unchanged after a spring fire (May, before growth began) and increased after both summer (August, peak of growth) and fall (October, winter dormancy) fires in Manitoba. The increase on the fall fire was lower than on the summer fire (Thompson and Shey 1989). This species invades burned areas via wind-dispersed seed (McKell 1950).

Fewer total and functional flower heads were produced following dormant-season winter and spring fires (Young 1986). Flowering activity was also inhibited following a May fire in Minnesota (Pemble et al. 1981). Prescribed spring burning may be a useful means of slowing the spread of Canada thistle. Spring fires would reduce the number of mature plants. They would also reduce the number of functional flower heads, resulting in lower seed production and a slow-down in the spread of new plants. Dormant-season fire is also beneficial to many native grass species, which would make stands more productive. Increased grass production would interfere with Canada thistle growth and reproduction, and possibly decrease its rate of spread (Young 1986).

Spotted knapweed (*Acosta maculosa*)

Spotted knapweed resists low-severity fire because of its stout taproot. However, low-severity fire will top-kills spotted knapweed. Spotted knapweed colonizes after fire from seeds buried in soil or from off-site sources. Spotted knapweed shows moderate increases after fire (Noste 1982).

Spotted knapweed has been reported on burned sites several years after wildfires. For example, it appeared on a site in western Montana 3 to 5 years after a severe wildfire (Toth 1991). Spotted knapweed was present on a burn 3 years after a wildfire in the White Cap Wilderness, Idaho (Bradshaw n.d.). The presence or absence of spotted knapweed in the pre-fire vegetation of either wildfire was not reported.

Spotted knapweed infests areas at the rural-urban interface. Fire in spotted knapweed infested fields may burn severely or not at all, depending on fuel load and continuity. The fire severity depends on the amount of dry knapweed stems and the amount of fine grass fuels (Xanthopoulos 1986).

Xanthopoulos (1986) has developed a fuel model for spotted knapweed. The fuel model uses three components for calculating fuel load: old standing knapweed on site for at least one winter, newly grown knapweed, and litter and fine grasses. Independent variables are litter depth and cover, and spotted knapweed plant height and percent canopy cover. If short, fine

grasses have greater than 40 percent canopy cover, a short grass fuel model should be used instead.

In order to ensure that fires carry in spotted knapweed, they should be conducted in early spring prior to grass and forb growth. Early April test fires in spotted knapweed fields did not burn completely because of low wind speed and high moisture content of early growth by grasses and forbs. The spotted knapweed fuel model predicts flame heights for windspeeds less than 2.5 miles per hour (4 km/hr) to be less than 4 inches (10 cm), which is probably too low for fire to carry. In preliminary field tests of the fuel model, flame heights ranged from 1 to 4 feet (0.3-1.2 m) at wind speeds ranging from 3 to 8 miles per hour (5-13 km/hr) (Xanthopoulos 1986).

Prescribed burning alone is probably not effective for controlling spotted knapweed and may cause increases, but prescribed burning may be useful in conjunction with herbicides. In Montana, Carpenter (1986) tested the possibility that burning may reduce herbicide interception by old spotted knapweed stems and may increase seed germination, increasing the effectiveness of subsequent herbicide treatment. However, burning did not increase herbicide effectiveness. The April fire was followed by an unusually dry period so spotted knapweed did not germinate prior to the May herbicide treatment (Carpenter 1986).

Russian thistle (*Salsola kali*)

Russian-thistle aids in spreading fire. It burns easily because the stems are spaced in an arrangement that allows for maximum air circulation (Young 1991). Also, dead plants contribute to fuel load by retaining their original shape for some time before decomposing (Evans and Young 1970). The rolling action of the plant spreads prairie wildfire quickly. Russian-thistle colonizes a burn when off-site, abscised plants blow across it, spreading seed (Young 1991). The tendency of dead plants to aggregate against fencelines and buildings creates a fire hazard. Tumbling, ignited plants can spread fire, and may bounce across fire lines (Young 1991). Prescribed burning will not control Russian-thistle, since it colonizes from off-site and thrives in disturbed communities.

Fire presumably kills Russian-thistle and kills at least some of the seed retained in leaf axils. Russian-thistle colonizes a burn site within 1 to 3 years. It dominated a big sagebrush community in Idaho at post-fire year 2, contributing 58 percent of the total community biomass (Fraley 1978). On the Mesa Verde Plateau of Colorado, it codominated a burned area with Bigelow aster (*Machaeranthera bigelovii*) at post-fire year 3 (Erdman 1969). Once dominant, Russian-thistle retains dominance for an average of 1 more year. At post-fire year 3 or 4, populations decline until further disturbance (Young 1991).

Fire Effects on Fauna and Fire

The most commonly observed wildlife on USAFA are mule deer (*Odocoileus hemionus*), which are found in every vegetation zone. There is also a small population of white-tailed deer (*Odocoileus virginianus*) that inhabit riparian communities dominated by cottonwoods and willows along Monument Creek. Other seasonal residents include: black bear (*Ursus*

americanus), elk (*Cervus elaphus*), mountain lion (*Felis concolor*), bighorn sheep (*Ovis canadensis*), and pronghorn antelope (*Antilocapra americana*) (Ripley 1994).

USAFA also supports diverse populations of smaller mammals including: beaver (*Castor canadensis*), coyote (*Canis latrans*), bobcat (*Lynx rufus*), porcupine (*Erethizon dorsatum*), eight species of squirrel, and eastern and desert cottontail rabbits (*Sylvilagus floridanus* and *S. audubonii*) respectively. The most commonly seen raptors on USAFA are: Red-tailed Hawk (*Buteo jamaicensis*), Golden Eagle (*Aquila chrysaetos*), and Prairie Falcon (*Falco mexicanus*) (Ripley 1994).

General Effects of Fire on Fauna

Vertebrates

Immediate response of vertebrate animals to fire appears to span the spectrum from wild panic to calm movement away from the fire to positive movement toward the fire (Lyon et al. 1978). The kind of response is related to both the mobility of the animal and size of the fire, with smaller rodents such as squrirels, mice, and chipmunks being most likely to exhibit panic (Udvardy 1969; Komarek 1969; Tevis 1956). Larger, more mobile animals such as moose, deer, elk, and raccoon usually move calmly (Hakala et al. 1971; Vogl 1973; Sunquist 1967), while many insectivorous birds, quail, turkeys, and birds of prey appear to be attracted by fires (Phillips 1965; Komarek 1967; 1969; Stoddard 1963).

Direct mortality of animals in fires has been documented by some investigators (Hakala et al. 1971; Chew et al. 1958; Ahlgren and Ahlgren 1960; Lutz 1956). However, other investigators have remarked on the relative scarcity of dead animals in burned areas (Stoddard 1963; Keith and Surrendi 1971; Sims and Buckner 1973). In general, while some evidence of vertebrate mortality has been reported, the most common opinion is that vertebrates are rarely killed in fires, and where death does occur, it is usually negligible (Vogl 1967; Phillips 1965; Stoddard 1963).

Invertebrates

Effects of fire on invertebrate populations may be transitory or long lasting (Lyon et al. 1978). In general, invertebrates decrease because the animals or their eggs are killed by the flames or heat, and their food supply and shelter are diminshed. In some instances flying insects are attracted by heat, smoke, or damaged trees; thus populations of certain species may increase during and after a fire (Lyon et al. 1978).

Fire effects on butterflies are dependent on whether the butterfly is a specialist or generalist. Specialists decline strongly and significantly after fire, and this effect persists for 3-5 or more years. Species with the broadest habitat niche (invaders) are most abundant in the most recently burned areas and usually scarcest in longest unburned (Swengel 1996). Quick, shallow winter burns are the most beneficial to butterflies. Many butterfly food-plants thrive after burning (e.g, violets, primrose, and wild strawberry). Patchy burning is also beneficial, diversifying homogeneous grasslands by creating mosaics (Oates 1995).

Fire Effects on USAFA's Common Fauna

Mule deer (*Odocoileus hemionus*)

Prescribed burns in oak woodlands can create access to the understory forage of the oak woodlands (Klinger et al. 1989). Small burns are more beneficial than large burns to mule deer because they tend to use burned areas close to cover. The optimum width for burns in these communities may be less than 0.25 mile (0.4 km) (Blackburn et al. 1975).

Burning can control sagebrush in areas where it has dominated grasslands and reduced deer forage (U.S. Department of Agriculture, Forest Service 1973). Kufeld (1983) recommended burning Gambel's oak in autumn during or immediately following leaf fall and building fire breaks 26 feet wide (8 m) around the areas to be burned. Gambel's oak recovers quickly following fire, particularly at low elevations where mule deer winter, therefore its growth must be monitored and retarded to improve mule deer habitat (Kunzler and Harper 1980).

Burning in grassland communities reduces litter that otherwise inhibits new growth of grasses. To maintain forage in bunchgrass communities, burning at 4- to 6-year intervals in winter or early spring is recommended (Johnson 1989).Fire rejuvenates and improves these grasslands, which are important winter range in some areas (Johnson 1989; Willms et al. 1980). Burning sagebrush communities can result in significant increases of herbaceous plants by reducing decadent sagebrush that outcompetes more nutritious and palatable species (Smith 1985; U.S. Department of Agriculture, Forest Service 1973). However, in areas where sagebrush is the only cover, its complete removal can be detrimental to mule deer populations (U.S. Department of Agriculture, Forest Service 1973).

Shrubs and forbs in piñon - juniper communities tend to increase the first few years following fire, providing valuable browse (Blackburn et al. 1975; McCulloch 1969). Mule deer seem to use these areas 15 years after a fire (McCulloch 1969; Klebenow 1985). Klebenow (1985) reported that the beneficial effects of fire for mule deer in piñon-juniper stands can last as long as 115 years. However, Bunting (1987) concluded that burning of these stands becomes increasingly difficult as stands grow older because fine fuels in the understory are reduced. He stated that burning should take place at early successional stages and at intervals based on the fire tolerance of desirable forage species. Everett (1987) warned that preburn conditions in piñon-juniper stands will most likely determine the post-fire plant composition. If perennial shrubs are present before a burn, they will come back following fire. If no shrubs are present, perennial grasses will develop (Blackburn et al. 1975).

Fire can be used to stimulate browse, create openings in dense, inaccessible plant communities, and reduce slash, as well as increase nutrient content and palatability of forage (Gruell 1986). A mosaic of seral stages is best for mule deer (Gruell 1986). Gruell (1986) listed several factors that influence post-fire plant composition, including the severity, size, and season of the burn, fuel type, post-burn foraging intensity, and the preburn plant community composition. He stated that surface fires of moderate-intensity following thinning or selection cuts can improve Douglas-fir or ponderosa pine forests for mule deer by promoting regeneration of crown-sprouting shrubs and preparing the seedbed for herbs and shrubs.

Although uncommon, mule deer can be trapped and killed by fast-moving fires (Cowan 1956; Hines 1973). Deer seem to prefer foraging in burned areas, although preference may vary seasonally (Biswell 1989; Johnson 1989; Klinger et al. 1989). This preference may indicate an increase in plant nutrients which usually occurs following fire (Severson 1987). Hobbs and Spowart (1984) warned about making conclusions regarding the benefits of fire based on forage studies alone. Their study of fire on nutrition in Colorado revealed increases in the quality of deer diets due to changes in forage selection-not increases in nutrients of previously selected forage.

Elk *(Cervis elaphus*)

The most preferred elk forage species are enhanced by an increase in nutrients following fire (DeByle et al. 1989; Rowland 1983). Many studies, however, conclude that an increase in quantity of forage is more significant than an increase in quality (Bartos and Mueggler 1979; Canon 1985; Canon et al. 1989). Site preference studies show that elk usually prefer to graze on burned as opposed to unburned sites (Canon 1985; Canon et al. 1989; Rowland 1983). Grazing can reduce fuel buildup in grasslands, thereby decreasing the ability of a range to carry fire (Skovlin 1982). Fire in a Southwestern ponderosa pine forest increased forbs, grasses, and shrubs, created edge, and provided snags for cover. Elk increased in the burn, reaching a peak 7 years after fire when grasses were most abundant (Lowe 1975).

Fire improves the quality of forage under aspen stands (Canon 1985; Canon et al. 1989; DeByle et al. 1989). Canon (1985) cited several studies on fire effects in aspen communities. Fire regenerates decadent aspen stands, opens the understory, increases forbs and grasses by reducing shrubs, and increases aspen suckering, which may provide enough browse to compensate for overgrazing (Gruell and Loope 1974).

Prescribed fire is used routinely to create or enhance elk habitat in many Western states. Historical evidence shows that early Native Americans used fire to attract ungulates (McCabe 1982). Fire can be used to rejuvenate aspen stands, encourage early spring green-up of grasslands by reducing litter, slow or prevent conifer dominance in important foraging areas, increase palatability of foods, reduce the height of browse species, and stimulate regeneration through sprouting or heat scarification of seed (Jourdonnais and Bedunah 1990; Weaver 1987).

Where elk forage heavily in aspen stands, large areas should be burned to reduce grazing pressure by encouraging elk to disperse (Brown 1985b). Also, burning several small units nearby will improve elk dispersal and lessen grazing impact. Kramp et al. (1983) reported that elk prefer burns smaller than 8.6 acres (3.5 ha), and use of burns decreased with an increase in distance to cover.

Bighorn sheep (*Ovis canadensis*)

Prescribed fire can be a useful tool in managing bighorn sheep habitat (Peek et al. 1984). However, prescribed burning and its associated human activity in bighorn sheep range may increase stress levels in a population. Herd condition should be considered when planning time of fire (Woodard and VanNest 1990).

Many bighorn sheep populations originally occurred in areas with frequent fire intervals (Peek et al. 1984; Stucker and Peek 1984). Bighorn sheep inhabiting the Salmon River drainage of Idaho occupy a region where over 64 percent of their habitat has burned since 1900 (Stucker and Peek 1984). Fire suppression for over 50 years has allowed plant succession to alter many bighorn sheep habitats throughout North America (Chapman and Feldhamer 1982; Easterly and Jenkins 1992). Fire suppression, which has allowed conifers to establish on grasslands, has decreased the forage values on many bighorn sheep ranges (Easterly and Jenkins 1992).

Fire is an important factor in creating habitats that are heavily used by bighorn sheep (Chapman and Feldhamer 1982; Stucker and Peek 1984). Periodic burning keeps seral grasslands from becoming dominated by climax coniferous trees (Woodard and VanNest 1990). In April 1987, a prescribed fire was conducted on 235 acres (95 ha) of bighorn sheep winter range in Custer State Park, South Dakota. Burning expanded foraging habitat for bighorn sheep by curtailing encroachment of pondersosa pine onto mixed-grass prairie. Burning may also regenerate rangelands and enhance the production, availability, and palatability of important bighorn sheep forage species (Woodard and VanNest 1990). Bighorn sheep heavily utilized burned winter range the following two winters after a September 1974 fire at the East Fork of the Salmon River, Idaho (Peek et al. 1984). Over 66 percent of the plants on this burned range had been grazed by bighorn sheep. Utilization was consistently higher on burned sites than on adjacent unburned sites for 4 years after the fire (Peek et al. 1984). Burning can also increase the visibility for bighorn sheep. Research has shown that on burned sites bighorn sheep use areas more distant to escape terrain than on adjacent unburned sites (Woodard and VanNest 1990).

Fire can negatively affect bighorn sheep habitat when range condition is poor and forage species cannot recover, when nonsprouting species that provide important forage for bighorn sheep are eliminated, or when too much area is burned and forage is inadequate until the next growing season. Another potentially negative effect is when other species, especially elk, are attracted to prescribed burns intended to benefit bighorn sheep (Peek et al. 1984).

Prescribed burning has been widely used to increase the quantity and nutritional quality of bighorn sheep forage throughout North America (Easterly and Jenkins 1992). Prescribed crown fires conducted in winter in mature, coniferous stands adjacent to escape terrain may provide an inexpensive solution to maintaining or establishing bighorn sheep winter range. In areas where the available sheep range is large and provides alternative and distant wintering sites, fires should be prescribed or located in areas that would minimize the stress on sheep. Early spring fires, particularly on south and southwest aspects, may provide more early spring forage than would otherwise be available for bighorn sheep (Woodard and VanNest 1990). Burning immature forests and scrublands adjacent to bighorn sheep winter range could also provide migration corridors between winter and summer ranges (Stucker and Peek 1984).

Since both positive and negative effects can occur from burning bighorn sheep range, a well-thought-out plan must be developed before fire is considered for use on their range. Plans must consider the following: 1) condition of plants; 2) plant response to burning; 3) adjacent conifers (the possibility of creating more open range exists if conifer stands or tall shrub fields occur next to currently used ranges); 4) limiting factors (factors that may limit bighorn sheep populations should be identified, and an evaluation made as to how burning will effect these limiting

factors); 5) lungworm (lungworm infections can possibly be affected by altering bighorn sheep concentrations. If burns are small and concentrate bighorn sheep, results could be negative; if burns disperse populations, the effects could be positive) and; 6) competition by other ungulates attracted to burns (Peek et al. 1984).

Coyote *(Canis latrans*)

Fire may improve the foraging habitat and prey base for coyotes. Fires that reduce vegetation height and create open areas probably increase hunting efficiency for coyotes. Surface fires often open substrates for quieter stalking and easier capture of prey than can occur in closed forests (Landers 1987). Wirtz (1977) noted increases in consumption of birds and deer by coyotes after a chaparral fire in the San Dimas Experimental Forest, California. Increased consumption was presumably the result of increased vulnerability of prey with reduced cover, but no change was noted in small mammal consumption.

Periodic fire helps to maintain habitat for many prey species of coyote. Fires that create a mosaic of burned and unburned areas are probably the most beneficial to many coyote prey species. Several studies indicate that many small mammal populations increase rapidly subsequent to burning in response to increased food availability. Fire often improves hare and rabbit forage quality and quantity for two or more growing seasons (Landers 1987).

The 1988 fires in Yellowstone National Park have probably benefited coyotes (Mills et al. 1989). Fire in combination with drought likely increased available carrion the fall and winter following the fire. Additionally, the fires stimulated grass production, which should lead to an increase in small mammal populations (Mills 1989). Prescribed burning that favors small mammals by creating ecotones and different age classes of vegetation would increase the prey base for coyotes and make hunting easier by opening up the habitat (Quinn 1990).

Red-tailed Hawk (*Buteo jamaicensis*)

Fire directly reduces Red-tailed Hawk reproductive success if the fire crowns in occupied nest trees (Landers 1987). Fires that kill or otherwise alter unoccupied nest trees may disrupt reproduction if acceptable nest trees are scarce. Red-tailed Hawks are reported to be attracted to fire and smoke (Dodd 1988). They have been reported feeding on grasshoppers fleeing from fires (Landers 1987). Low-severity fires probably have little direct effect on Red-tailed Hawks. Landers (1987) commented that light winter burning probably does no substantial harm to raptors.

Although fire may reduce potential nest trees, it may also create snags for perch sites and enhance the foraging habitat of Red-tailed Hawks. Red-tailed Hawks often perch on snags created by lightning strikes (Baker 1974). They often use fresh burns when foraging due to increased prey visibility (Dodd 1988; Landers 1987; Nichols and Menke 1984). Regular prescribed burning helps to maintain habitat for many prey species of Red-tailed Hawks (Dodd 1988; Landers 1987; Lehman and Allendorf 1989). Several studies indicate that many prey populations increase rapidly subsequent to burning in response to increased food availability

(Dodd 1988; Landers 1987). Fire suppression in grasslands was detrimental to small bird and mammal populations due to organic matter accumulation and reduced plant vigor (Wagle 1981).

The suppression of natural fire in chaparral (e.g., oak shrubland) has resulted in reduced seral stage diversity and less edge (Dodd 1988), which has probably affected Red-tailed Hawks in these communities. Additionally, Red-tailed Hawks habitat is favored by fires that open up or clear piñon-juniper woodlands (Mason 1981). Raptors associated with piñon-juniper woodlands depend upon edges of openings created by fire and scattered islands of unburned woodlands (Dodd 1988).

In the first year following a severe fire in grassland, ponderosa pine, Douglas-fir, and mountain big sagebrush habitat types on the Salmon National Forest, several Red-tailed Hawks were observed within the burn. They were not observed in the area before the fire (Collins 1980). Following a fire in a mountain big sagebrush community on the Bridger-Teton National Forest, Red-tailed Hawks were more commonly observed using an area that experienced a severe fall fire than in a nearby area burned by a low-severity spring fire (McGee 1976).

Prescribed fire can be benefical to Red-tailed Hawk populations by enhancing habitat and increasing the prey base (Dodd 1988; Landers 1987). Prescribed burning plans should strive for creation of maximum interspersion of openings and edge, with high vegetative diversity. Habitats should be maintained in a random mosaic. In most cases, burning plans must be integrated with proper range management. Reseeding of perennial grasses as well as rest from livestock grazing may be necessary to achieve desired goals. Burning should be deferred until nesting is completed in areas where impact to breeding Red-tailed Hawks may occur (Dodd 1988).

Golden Eagle *(Aquila chrysaetos)*

Fire reduces Golden Eagle reproductive success if the fire crowns in occupied nest trees (Landers 1987). Fires that kill or otherwise alter unoccupied nest trees may disrupt reproduction if acceptable nest trees are few. Low-severity fires probably have little direct effect on Golden Eagles. Landers (1987) commented that light winter burning probably does no substantial harm.

High-intensity, catastrophic fires potentially affect nest trees, destroying perch and roosting trees. These snags are used by Golden Eagles for nesting, perching and/or roosting. Use of trees probably depends more on proximity to prey than condition (live or dead). Fires probably enhance the prey base and hunting efficiency of Golden Eagles. Golden Eagles were seen using recently burned sites in the Lincoln National Forest, New Mexico. Golden Eagles there were probably taking advantage of abundant prey associated with the growth of new vegetation on the burned site (Lehman and Allendorf 1987).

Fire suppression in this century has contributed to the loss of Golden Eagle breeding pairs in the Appalachian Mountains of the eastern United States. Historically, open areas used by Golden Eagles for foraging in those mountains were maintained by fire. After full suppression policies began, the openings reverted to brush and eventually to forest. Today, there are few openings in

the Appalachian Mountains; as a result, Golden Eagles have almost disappeared (Spofford 1971).

Prairie Falcon *(Falco mexicanus*)

Direct mortality from fire is rare for raptors (Lehman and Allendorf 1989). Adults can probably easily escape fire, and eggs and nestlings are rarely in locations that can burn. Grassland raptors such as Prairie Falcons have been adversely affected by fire exclusion wherever woodlands have encroached upon grasslands (Lehman and Allendorf 1989). Periodic fire may enhance the foraging habitat of Prairie Falcons and increase the prey base (Anderson, D. 1991; Dodd 1988; Lehman and Allendorf 1989). Several studies indicate that many small mammal and bird populations increase rapidly subsequent to burning in response to increased food availability (Dodd 1988; Lehman and Allendorf 1989). Additionally, fires in grasslands may increase prey availability by removing accumulated litter and reducing cover (Anderson, D. 1991). Fire suppression in grasslands is detrimental to populations of small bird and mammal herbivores due to organic matter accumulation and reduced plant vigor (Wagle 1981).

Although fire is often beneficial to Prairie Falcon prey species, Yensen et al. (1992) reported that in the Snake River Birds of Prey Area, southwestern Idaho, fire may reduce populations of Townsend's ground squirrels (*Spermophilus townsendii*), a major prey species of Prairie Falcons. To create or maintain desert grasslands, prescribed burning at an interval not less than 5 years is recommended. Periodic fire at approximately 5-year intervals will probably maintain an open condition, though burning over successive years may be necessary to eliminate woody invaders. Five-year intervals between fires allow for herbaceous plant recovery while not adversely affecting prey populations. The goal of prescribed burning in chaparral should be to create opportunities for perennial grasses to extend the open grass-shrub character. Complete elimination of climax chaparral species is not recommended. Periodic fire at approximately 5-year intervals will probably maintain an open condition. In most cases, burning plans must be integrated with proper range management. Post-fire seeding of perennial grasses as well as rest from livestock grazing may be necessary to achieve desired goals. Because of human disturbance, prescribed burning should be deferred until nesting is completed in areas where impact to breeding Prairie Falcons may occur (Dodd 1988).

Wild Turkey (*Meleagris gallopavo*)

Spring fires may destroy nests. Fast-moving fires may kill newly hatched poults, but once wild turkeys can fly, fires are probably not much of a problem; and losses to the population are negligible (Hurst 1981).

Prescribed fire can be used to stimulate the growth of food plants and promote early spring green up of grasses (Schroeder 1985). Fire can also reduce litter, exposing seeds and insects; and reduce brush so that turkeys can be wary of predators (Hurst 1981; Stoddard 1963). Fire can be used to create edges to increase nesting habitat (Stoddard 1961). It can also reduce parasites such as ticks and lice (Jacobson and Hurst 1979). Devet and Hopkins (DeGraaf et al. 1991)

recommended burning loblolly-longleaf pine stands every 3 years, and burning every 4 to 6 years in Piedmont regions.

Fire Effects on Dominant Plants of Major Vegetation Zones

The following information is adapted from the Fire Effects Information System (FEIS), USDA Forest Service Intermountain Research Station's Fire Sciences Laboratory (IFSL) in Missoula, Montana. This national interagency implementation is sponsored by the National Wildfire Coordinating Group and the USDA Forest Service. The following section is divided into the following major vegetation types: mixed coniferous forest, upland forest shrubs, upland shrubland, upland grassland, riparian shrub/tree, and riparian grass/forb. The species are arranged alphabetically by genus within each vegetation type.

Mixed Coniferous Forest

Abies concolor--white fir

Fire ecology or adaptations

Plant adaptations to fire

Sapling- and pole-sized white fir are fire sensitive. Trees of this size have smooth, thin, resinous bark and low-growing branches which easily ignite from burning undergrowth (Hopkins 1982; Raphael and White 1984). As the bark thickens and lower branches drop due to self-pruning, trees progressively become more fire resistant. Self-pruned trees that have attained a d.b.h. of 8 to 10 inches (20-25 cm) are moderately resistant to fire (Hopkins 1982). Bark on older trees may be 5 to 7 inches (12.7-17.8 cm) thick (Vines 1960) and afford large trees fire protection. Mature trees which survive fire, those which escape fire in small, unburned pockets, and trees which are adjacent to burned areas provide seeds to colonize burned sites (Bock and Bock 1977; Minnich 1977). Wind-dispersed white fir seeds generally travel a distance of only 1.5 times the height of the tree, but some seed is dispersed over much greater distances into burned areas (Fiske and DeBell 1989; Minnich 1977).

Fire regime

In mixed conifer forests of the southern Cascades and Sierra Nevada, fire frequency during pre-settlement times was from 6 to 20 years (Kilgore and Taylor 1979; McNeil and Zobel 1980; Wagener 1961b). Pre-settlement fire frequency in mixed conifer forests of the central and southern Rocky Mountains was from 7 to 22 years (Alexander et al. 1984; Dieterich 1983). These frequently occurring fires were generally of low-intensity because the short time span between fires resulted in low accumulations of dead and down fuels. High-intensity, stand-replacing fires were uncommon (Dieterich 1983). This regime of frequent, low-intensity fires promoted "open-grown" forests, as large numbers of small, shade-tolerant, fire-sensitive conifers like white fir were killed. Thus before fire suppression began in mixed conifer forests, ponderosa pine and Douglas-fir, which are more fire resistant than white fir, often dominated the

overstory (Agee 1982; Hopkins 1982). As a result of fire suppression since the turn of the century, white fir density has greatly increased in mixed conifer forests. Today, unnatural, heavy accumulations of dead fuels and abundant young white fir (which often form "dog-hair" thickets) greatly increase the chances for high-intensity, stand-replacing crown fires (Alexander et al. 1984; Kilgore 1973a; Parsons and DeBenedetti 1979).

Fuels and fire behavior

Annual litter accumulation under white fir is about 1 to 1.5 tons per acre (1.2 kg/ha) (Biswell et al. 1966; Kilgore 1973a). The needles are small and fine, and form a compact fuel bed that does not burn as readily as fine fuels under pines (Agee et al. 1978). Fire behavior and fuel consumption can vary greatly as fuel moisture and weather conditions vary at the time of burning. This variation is well demonstrated in a study by Kauffman and Martin (1989). They conducted a series of prescribed understory fires under a wide range of conditions in a 70-year-old mixed conifer stand codominated by white fir, ponderosa pine, and incense cedar.

Fire effects

Immediate fire effect on plant

Sapling- and pole-sized white fir have thin bark that provides little insulation for the cambium, and shallow roots which are susceptible to soil heating. Because of its shade tolerance, white fir is slow to self-prune lower branches. These low-growing branches, which have slender twigs and finely divided foliage, easily ignite from burning undergrowth (Kotok 1933; Wagener 1961a). Consequently, young white fir are often killed even by low-intensity surface fires (Atzet and Wheeler 1982; Kilgore 1973a; Thomas and Agee 1986). Larger trees are more fire resistant. Mortality results from crown scorch, girdled stems from cambial heating, or root damage from soil heating (Thomas and Agee 1986). Trees damaged or weakened by fire are susceptible to attack by insects and disease. Fire wounds in contact with the ground provide an entry point for decay fungi (Wenz and Pronos 1983). Fire-weakened trees that are attacked by insects can be killed within a few years (Thomas and Agee 1986).

In general, small-diameter white fir trees suffer high mortality following fire. Mortality decreases with increasing size. Prescribed low-intensity summer surface fires in mixed conifer forests of Crater Lake National Park, Oregon, resulted in high mortality of small white firs. Four years after burning, 86 percent of trees less than 2.2 inches (5.5 cm) d.b.h., 61 percent of trees larger than 2.2 inches (5.5 cm) d.b.h. and less than 33 feet (10 m) tall, 31 percent of trees 33 to 99 feet (10-30 m) tall, and 15 percent of trees taller than 99 feet (30 m) were dead (Thomas and Agee 1986). About two-thirds of white fir fire mortality in this study occurred within 1 year of burning (Thomas and Agee 1986). High mortality of small white firs was similarly observed in Kings Canyon National Park, California, 1 year after a low-intensity November prescribed fire in a giant sequoia-mixed conifer forest. Ninety-one percent of trees less than 6 inches (15 cm) d.b.h., 39 percent of trees 6 to 12 inches (15-30 cm) d.b.h., and 5 percent of trees larger than 12 inches (30 cm) d.b.h. were killed (Kilgore 1973b).

Discussion of fire effect

Plant response to fire

Following stand-replacing fires, white fir reestablishes via wind-dispersed seed. Exposed mineral soil seedbeds created by fire favor seedling establishment (Kauffman and Martin 1989)

however, seedling establishment and survival in sunny locations is often poor (Barbour et al. 1990). Seedlings establish quickly after fire if a canopy remains (Kilgore and Biswell 1971) but may take several years to establish if the canopy has been removed.

Less than 1 year following underburning in a giant sequoia grove, white fir seedling establishment in June was abundant, and 39 percent survived until October (Agee and Biswell 1969). Minnich (1977) studied conifer reproduction on burned areas in mixed conifer forests in the San Gabriel and San Bernardino mountains of southern California. He found that seeds originating from outside the burned area were responsible for abundant white fir reproduction on burns older than 10 years, but that reproduction was scant on burns less than 5 years old. Similarly, no white fir seedlings established within 5 years of a crown fire in the Sierra Nevada, even though several mature trees escaped the fire and thus provided a seed source (Conard and Radosevich 1982). Another Sierra Nevada study found negligible white fir reproduction 17 years following a stand-replacing fire, even though a seed source was readily available(Bock and Bock 1977).

In the mixed conifer zone of the Sierra Nevada, white fir seedlings are often abundant under montane chaparral shrubs which form brushfields after high-intensity crown fires. Conard and Radosevich (1981) found white fir reproducing abundantly on shrub-dominated sites, with a combined seedling and sapling density ranging from 445 to 4,453 per acre (1,100-11,000/ha) where crown fires had occurred 38 to 62 years earlier.

Discussion of plant response

Fire management considerations

Thinning: Because sapling- and pole-sized white fir are sensitive to even low-intensity fires, prescribed fire can be used as a thinning tool. In mixed conifer forests where white fir dominates the understory due to years of fire suppression, prescribed low-intensity surface fires will kill large numbers of white fir. This reduces the hazard of white fir providing a fuel ladder to ignite the crown of overstory trees and also restores tree species composition closer to that of pre-settlement conditions. When fire prescriptions cannot ensure that young white fir will not ignite the crown of overstory trees, cutting all trees under a certain size before burning reduces this fire hazard. In three different studies, white fir less than 9 inches (23 cm) d.b.h. (Kilgore and Biswell 1971) 6 inches (15 cm) d.b.h. (Mohr and Petersen 1984) and 11 feet (3.3 m) tall (Biswell et al. 1968) were felled before burning.

Preharvest underburning/site preparation: Underburning before timber harvesting with the shelterwood method in mixed conifer forests can be used to aid natural regeneration. Prescriptions developed in the Blue Mountains of northeastern Oregon recommend felling all understory trees less than 6 inches (15 cm) in diameter before burning (Mohr and Petersen 1984). The combination of cutting and burning removes all advanced regeneration, thus sanitizing the site of heart rot which is present in many 5- to 6-inch-diameter (12.5-15 cm) white fir (these trees are grand fir x white fir hybrids). Following harvest, seedling establishment of all conifers was abundant (Mohr and Petersen 1984). In some locations preharvest underburning is not recommended because it stimulates dormant shrub seeds to germinate and thus promotes the growth of shrubby vegetation which restricts the establishment and growth of conifers (Weatherspoon 1985)

Erosion: In the Siskiyou Mountains, soil erosion was found to be a problem following fires that remove duff layers on granitic soils (Atzet and Wheeler 1982).

Picea pungens--Colorado blue spruce

Fire ecology or adaptations:

Plant adaptations to fire

Blue spruce is easily killed by fire (Jones 1974; Wright and Bailey 1982). It has thin bark and shallow roots which make it susceptible to hot surface fires (Bradley et al. 1992). Blue spruce is slow to self-prune lower branches; therefore, surface fires can crown (Crane 1982). Blue spruce foliage has moderately volatile oils (Simpfendorfer 1989). Crowns are dense and highly flammable (Schmidt and Larson 1989). However, surviving blue spruce remain wind-firm in stands opened by fire (Bradley et al. 1992).

Fire regime

In riparian areas where blue spruce occurs, intervals between fires are about 350 to 400 years. Severe fires occur infrequently, and succession back to the original community is often relatively rapid (15 to 35 years). Depending on the site, blue spruce may be the dominant seral tree (Crane 1982).

Successive fires may prevent blue spruce from dominance because it is fire intolerant. Historical fire frequency in mixed conifer forests was about 22 years, based on fire-scarred trees in the White Mountains of Arizona (Dieterich 1983). Fire suppression during the past 100 years has made the mixed conifer forest in which blue spruce occurs more susceptible to fire; however, blue spruce may be dominant in some areas because of the longer fire-free intervals.

Fire effects

Immediate fire effect on plant:

Fire kills blue spruce. Low severity fires will kill saplings and seedlings (Alexander 1974). Slow burning of fine fuels will kill the shallow roots of blue spruce (Bradley et al. 1992).

Discussion of fire effect

Plant response to fire

Blue spruce does not sprout after fire (Simpfendorfer 1989). Rates of establishment will vary depending on proximity of seed trees and moisture. Seed must be transported from off-site. Blue spruce will establish by wind-dispersed seed that readily germinates on the mineral soil exposed by fire. Small mammals and birds may also carry cones or seeds into a burn.

Discussion of plant response

Fire management considerations

Fire is less frequent in montane mixed conifer forests at lower elevations where Douglas-fir is dominant with blue spruce than in ponderosa pine types. Quaking aspen is seral and present because of fire in these forests. Prescribed fire here would increase habitat and browse for wildlife (Severson and Rinne 1990)

Fuel prediction is difficult because of the large variation in natural fuel loadings in the forests where blue spruce occurs (Sackett 1979). Therefore, Sackett (1980) determined average squared diameters and specific gravities of blue spruce and seven other conifer species in Arizona and New Mexico. This established weight and volume of fuels using the planar intersect method. Greatest accumulations on the mixed conifer forest floor come from fermentation and humus layers (Ffolliottet et al. 1977). One fuel loading estimate was an average of 44 tons per acre (98 t/ha) (Youngblood and Mauk 1985).

Pinus flexilis--limber pine

Fire ecology or adaptations

Plant adaptations to fire

The thin bark of young limber pine trees does not protect from even cool fires. Since the bark at the base of older trees is often 2 inches (5 cm) thick, these trees can withstand stem scorch from low severity fires. Terminal buds are protected somewhat from the heat associated with crown scorch by the tight clusters of needles around them.

Fire regime
The vulnerability of limber pine to fire is reduced by the open structure of its stands and the dry, exposed habitats with sparse undergrowth in which it grows. Fires are infrequent and often of low-intensity with negligible spread. Fire intervals range from 50 to 200 years in Montana (Alexander et al. 1986). Fuel loadings from downed and dead materials are usually low. Hazardous fuel conditions are usually the result of dead herbaceous fuels (Fischer and Clayton 1983; Pfister et al. 1977; Youngblood and Mauk 1985)

Fire effects
Immediate fire effect on plant
Young limber pine trees rarely survive burns, partly owing to their thin bark (Minnich 1984).

Discussion of fire effect
Plant response to fire
Post-fire regeneration of limber pine is a consequence of seed caching by Clark's nutcrackers (Lanner and Vander Wall 1980).

Discussion of plant response
Regeneration of limber pine and Engelmann spruce was assessed in a high altitude area disturbed by fire which occurred on a south-facing slope on Niwot Ridge in Roosevelt National Forest along the Colorado Front Range about 1905. The uppermost elevation of the burn was about 11,000 feet (355 m), which is just below timberline. The slope gradient averages about 10 percent. Although regeneration at the uppermost elevation of the fire is less than at lower levels, colonization at all altitudes began shortly after the fire. There is no evidence that treeline has changed following the fire (Shankman 1984).

Fire management considerations
It is suggested that limber pines growing in open stands may be maintained by periodic fires which reduce the undergrowth. Where limber pine and Douglas-fir codominate, fire can act as a thinning agent slightly favoring limber pine over Douglas-fir in the younger age classes (Fischer and Clayton 1983).

Pinus ponderosa var. *scopulorum*--interior ponderosa pine

Fire ecology or adaptations:
Plant adaptations to fire
Interior ponderosa pine depends on frequent surface fires to maintain stand health and stability (Biswell et al. 1973; Cooper 1960). Consequently, interior ponderosa pine communities may have evolved flammable properties to encourage recurrent, low-intensity burning (Mutch 1970). The resinous needles provide an abundant, annual source of highly flammable fuel, with yearly accumulations in dense stands exceeding 3,500 pounds per acre (3,120 kg/ha) (Biswell 1973). Despite such characteristics, fire frequencies for interior ponderosa pine under natural fire regimes vary greatly according to site conditions and geographical area.

Fire regime

Fire has been a major ecological force in the pine forests of the Southwest. Results of fire scar analysis indicate that prior to 1876, fires occurred an average of every 2 years in northern Arizona (Dieterich 1980). Stands farther north have also been significantly influenced by fire. Average fire frequency in the Black Hills of South Dakota was estimated at 10 to 25 years (Wright 1978). Pine forests in Wind Cave National Park have burned every 13 to 21 years (Shilts 1976). In the central Rocky Mountains, however, the average mean fire interval is somewhat longer, perhaps because the more open nature of these stands results in less litter accumulation to carry fire (Wright 1978). Interpretation of fire scar data from interior ponderosa pine stands in the Front Range of Colorado indicated that prior to 1840, stands had a mean fire interval of 66 years (Laven et al. 1980). Interpretation of fire scar data from ponderosa pine stands in the Roosevelt National Forest, Colorado, indicate that prior to 1840, stands had a mean fire interval of 66 years (Laven et al. 1980). Goldblum and Veblen (1992) conducted a study near Boulder, Colorado and found the mean fire interval for pre-settlement averaged 22.0 years. Rowdabaugh's (1978) research in Rocky Mountain National Park indicates a mean fire interval for pre-white settlement to be 38.9 years. Veblen articulated in a recent telephone interview (April 3, 1997) that the mean fire interval for Great Plains/ponderosa pine ecotone in Boulder County should be every 7 years, at a minimum. However, Merrill Kaufman of the U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station, cautions that there is no standard mean fire interval that can be universally applied due to climatic and anthropogenic changes (pers. comm.).

There is a substantial difference in overall mean fire intervals among the three studies conducted in the Colorado montane zone. This could, in part, be explained by the different sizes of the study areas, as smaller units (within a study area) will tend to have longer fire return intervals (Arno and Peterson 1983; Dieterich 1983). The small size (50 ha) of the study area of Laven et al. (1980), compared to 600 ha for the Goldblum and Veblen (1992) may confound direct comparison of the mean fire intervals. The lower mean fire intervals for Four Mile Canyon may actually reflect more frequent anthropogenic fire. Fourmile Canyon was highly impacted by mining and railroads during the settlement era. In contrast, the study area in Roosevelt National Forest was less impacted by humans (Goldblum and Veblen 1992). In an ongoing research project Merril Kaufmann has collected fire scar data for 145 ponderosa pines at Cheesman Lake, southwest of Denver, Colorado. The Cheesman Lake sites historically experienced a series of large, moderate to high-intensity fires, with historical mean fire intervals between 20 to 120 years (pers. comm.).

Interior ponderosa pine has developed a number of adaptive traits which help to minimize fire damage to tissues. Thick, exfoliating bark and a deep rooting habit make large trees tolerant of most ground fires (Flint 1925). Mature trees tend to self-prune lower branches, thereby spatially separating foliage from burning ground fuels and reducing the potential for crown fires. Crown fires are further discouraged by long needles which are loosely arranged within an open-structured crown (Fischer and Clayton 1983). In addition, the foliar moisture content of this tree is relatively high (Saveland 1982). Trees burned during the dormant season are often able to survive extensive crown scorch damage because buds are large and enclosed within thin, insulative bud scales (Fowells 1965; Harrington 1987; Ryan 1982).

Fire creates a favorable seedbed for interior ponderosa pine by exposing bare mineral soil and removing competing vegetation. However, post-burn establishment is successful only when a good seed crop coincides with above average rainfall (Fowells 1965).

***Immediate fire effect on plant**:*
The effect of fire on interior ponderosa pine is generally related to tree size, fire intensity, and tree density (Alexander 1987). Low-intensity fires readily kill seedlings less than 12 inches (30 cm) in height (Biswell et al. 1973). Larger interior ponderosa pine seedlings can sometimes survive heat generated by low-intensity surface fires, especially dormant-season fires (Fischer and Clayton 1983; Fowells 1965). Larger seedlings, saplings, and pole-sized trees are damaged but not killed by low-intensity fires. Beyond the pole stage, interior ponderosa pine is quite resistant to the majority of ground fires (Schubert et al. 1970). Heat from fire can damage foliage, buds, cambium, and roots, although fire damage to roots is usually a factor only in very young trees, due to the deeply rooted nature of this tree (Flint 1925; Ryan 1982). In most cases, larger trees can withstand proportionally greater crown and stem damage than smaller trees because their thicker bark affords better insulation against cambial damage, and because their height elevates foliage and buds some distance from the flame zone (Wyant et al. 1986).

Site index can also affect post-burn mortality of interior ponderosa pine. Young, fast-growing trees on good sites typically survive fire better than overmature, slow-growing trees on poor sites (Wagener 1961a). Trees producing heavy cone crops are more prone to mortality because nutrients are diverted to cone development and maturation rather than to recovery (Wagener 1961a).

Discussion of fire effect
Plant response to fire
Post-burn mortality of interior ponderosa pine varies considerably depending on the extent of crown and bole damage. However, the exact relationship between the extent of fire damage (bole and crown) and mortality is not well understood (Ryan 1982). Damage to cambium is a more important factor in post-fire mortality in small-diameter classes, and crown damage more important in pole-sized and larger trees.

Bole damage: Interior ponderosa pine's resistance to fire is due in large part to the early development of thick, exfoliating bark that insulates the cambium from excessive heat damage. Besides being relatively nonflammable, pine bark does not conduct heat readily because of its low density and low moisture content (Saveland 1982). Bark on some old-growth trees may be 3 or more inches (6+ cm) thick (Lanner 1983). Lethal exposure time (minutes) is approximately equal to 3 times the squared bark thickness in centimeters, or 2 times the squared bark thickness in inches (Ryan 1982). Doubling the bark thickness increases the lethal exposure time by a factor of four (Hare 1965). Diameter increases alone have a significant impact on the partitioning of heat around the stem (Gill 1974). Cambial damage is most likely to occur when high-intensity fires are maintained at the base of a tree. Cambium must be completely girdled in order to kill trees, making smaller trees more susceptible to mortality (Wyant et al. 1986). Trees only partially girdled have a good chance of survival (Hermann 1954; Van Wagner 1963). Trees can tolerate basal girdling of less than 25 percent if crown and root damage are minimal. As the extent of girdling increases and larger patches are killed, even those trees with less than 60

percent crown scorch (high survival potential) are likely to succumb. Damage which occurs more than several feet up on the bole appears to increase post-burn mortality more than a similar amount of damage near the base (Wagener 1961a).

Cambial damage is rather inconspicuous initially, and charring of the bark is usually a poor indicator of its occurrence (Ryan 1982). Thin-barked trees may be killed without any external sign of charring, whereas thick-barked trees may be extensively charred but sustain only slight damage. Narrow strip killing beneath deep bark crevices is the most common form of cambial damage (Wagener 1961a). An immediate assessment of cambial damage can be made by checking areas where stem charring is most severe. Healthy cambium is somewhat moist and cream colored, whereas dead cambium is dry, brown or gray, and has a sour, fermented smell (Boldt and Van Deusen 1974). The first external sign of cambial damage is often large amounts of pitch exuding from the fringes of deeply charred bark. Another indication is an abrupt change in the pattern of bark beetle borings (Ryan 1982; Wagener 1961). Callus formation around the killed cambium eventually causes the bark to fall away, exposing sapwood (Lachmund 1921; Lachmund 1923). The bark may begin to loosen and slough within days to weeks on thin-barked trees, but this process may take several years on thick-barked trees. Flammable resin accumulations around the wound make these areas highly susceptible to subsequent fire damage. The occurrence of fire before the wound has entirely healed results in the formation of a fire scar or "cat face". Repeated fires tend to enlarge fire scars. Trees are highly susceptible to fire mortality if existing scars exceed 25 percent of the base circumference (Ryan 1982). Otherwise, if the exposed sapwood of existing scars is sound, fire does not have a measurable impact on survival. If the wood is rotten or infested with insects, however, the tree will probably burn through at the base (Lachmund 1923; Ryan 1982). Following prescribed fire, an estimated 1 percent of trees previously scarred by fire will probably fall over (Biswell 1967).

Crown damage: Crown scorch is widely considered to be the principle cause of pine mortality following fire (Dieterich 1979; Herman 1954; Lynch 1959; Van Wagener 1963; Wagener 1961a). Crown scorching generally weakens the tree, but mortality is usually negligible in healthy, vigorous trees when less than 30 percent of the crown length is killed (Ryan 1982). The literature contains highly variable reports on the extent of crown scorch damage trees can sustain (Davis et al. 1968; Herman 1954; Lynch 1959). Following prescribed burns in Colorado, Harrington (1987) reported that up to 90 percent scorch damage resulted in minimal mortality in trees that had sustained crown damage only. The author cautioned, however, that mortality would be likely if such extensive damage were combined with some degree of bole damage. Sixty-one percent of trees sustaining 100 percent scorch during fall burning survived.

Although crown scorch damage is important, numerous studies indicate that bud kill is even more crucial than crown scorch in determining the survival potential of fire-damaged trees (Hare 1961; Methven 1971; Wagener 1961a). Buds of interior ponderosa pine are large and protected by heavy bud scales that have lethal temperatures 68 degrees F(20 degrees C) higher than that of needles (Methven 1971). Consequently, extensive scorching of pine foliage sometimes occurs with only light damage to buds and twigs, allowing vigorous trees to maintain shoot growth on defoliated branches (Wyant and Zimmerman 1984). Some trees can sustain scorch damage of up to 90 percent as long as 50 percent of buds and twigs survive (Wagener 1961a). The pre-burn condition of a tree strongly influences survival potential. Immature, fast-growing trees generally

tend to survive the same proportion of scorch better than older, slow-growing trees (Ryan 1982). Young interior ponderosa pines have a high chance of surviving 100 percent crown scorch if 20 percent of the buds survive; older trees will survive the same damage if 40 percent of the buds survive (Herman 1954). The height of bud kill also influences the survival potential of trees by reducing the crown ratio. Ryan (1982) recommends maintaining a crown ratio of 30 and 40 percent in vigorous young trees and older, mature trees, respectively.

Season of burn: Interior ponderosa pine usually survives fires during the dormant season, largely because insulating scales form once leader growth stops (Ryan 1982) and because dormant-season fires are usually relatively cool (Dieterich 1979; Harrington 1987). Trees are least resistant to thermal damage during early spring and most resistant in the fall when dormant (Hare 1961). Trees can withstand up to 50 percent crown scorch from fall burning but only 30 percent crown scorch from spring burning (Mohr 1984).

Post-fire seedling establishment

Post-fire regeneration of interior ponderosa pine is exclusively through seed. Foliage-scorched trees are an important seed source for post-burn reestablishment (Rietveld 1976; Wagener 1961a). Although seed trees may not live longer than two post-fire seasons, juvenile cones continue to mature on trees with less than 66 percent of the live crown scorched. These seeds are generally highly viable. Fire encourages natural regeneration by exposing mineral soil and removing litter accumulations and competing vegetation. The degree to which interior ponderosa pine depends on fire to prepare suitable seedbeds may vary according to geographic region (Crane 1982; Laven et al. 1980; Peet 1981).

Front Range of Colorado

Along the Front Range, interior ponderosa pine seedling establishment on low-elevation woodland sites tends to occur episodically (Hoffman and Alexander 1976). On these dry sites, good seed crops during favorable moisture years may be more responsible for seedling establishment than fire. Litter buildup on these sites is typically light, resulting in infrequent fires and an abundance of exposed mineral soil seedbeds (Little 1979). Although fire undoubtedly influences the interval between periods of successful regeneration, it is probably not solely responsible for the episodic regeneration characteristic of this area (Laven et al. 1980; Peet 1981).

Post-fire growth and recovery

Since crown scorch damage decreases photosynthesis, generation of new needle tissure is necessary for recovery (Wyant et al. 1983). On sites in Colorado, crown condition improved after 2 years in 74 to 91 percent of trees in each of five damage classes (Potter and Foxx 1979). Wyant et al. (1983) reported that in Colorado prescribed fall underburning did not substantially affect the physiological processes of dormant buds. In fact, the post-fire environment was highly conducive to crown growth. Mean fascicle lengths and length and diameter of bud shoots produced during the first post-burn growing season were substantially greater on burned than on unburned plots. The authors of this study speculated that crown growth would be enhanced for at least two post-burn seasons (Wyant et al. 1983).

The response of interior ponderosa pine after burning is not well understood or documented. Underburning may enhance the water status of individual trees by reducing understory competiton and by decreasing transpiration loss by reducing crown surface area, thereby increasing the total moisture available to the remaining crown. Furthermore, if water supplies are adequate, tree growth is enhanced by the higher air and soil surface temperatures associated with burned areas (Steinbrenner and Rediske 1964). On sites where underburning was used to thin stands, reduction in understory competition resulted in increased diameter and height growth of pine (Cooper 1960).

Interior ponderosa pine growth is severely limited by low nitrogen availability (Heidmann 1976). Accumulation of litter is often cited as the cause of retarded organic nitrogen mineralization on many pine sites. Usually there is a net post-burn loss of total nitrogen from the forest floor, but a concomittant post-burn increase in available soil inorganic nitrogen is often reported. These post-burn nitrogen surges generally benefit tree growth (Covington and Sackett 1986).

Discussion of plant response

Underburning in interior ponderosa pine stands can release substantial amounts of nutrients bound up in surface organic matter. On sites in the Southwest, fall prescribed fire often reduces organic debris by 28 to 37 percent (Covington and Sackett 1984). Increased microbial activity immediately following prescribed burning may further reduce forest floor debris. Covington and Sackett (1984) indicated that microbial mineralization was probably responsible for decreasing forest floor organic matter by an additional 440 grams per square meter within 7 months of a prescribed burn, thereby reducing fuels without nutrient volatilization. Refer to Covington and Sackett (1984;1986) and Ryan and Covington (1986) for further details.

Fire management considerations

Repeated prescribed underburns can be used to thin interior pondersa pine and maintain open stocking of trees. Such burns would reduce numbers of seedlings, remove dense understories in sapling- or pole-sized stands, and thin low-vigor overstory trees. In the absence of fire, overstocked seedling patches usually develop into "doghair" thickets of interior ponderosa pine saplings. Trees in these stands are less vigorous and more susceptible to mortality from low-intensity fires than trees in more open stands (Biswell et al. 1973; Boldt and Van Deusen 1974; Cooper 1960).

Fire exclusion has profoundly influenced the stability of interior ponderosa pine stands (Cooper 1960). Covington and Sackett (1984) listed the following management problems associated with reduced fire frequencies: (1) overstocked sapling patches, (2) reduced growth, (3) stagnated nutrient cycles, (4) increased disease, insect infestations, and parasites, (5) decreased seedling establishment, (6) increased fuel loadings, (7) increased verticle fuel continuity due to dense sapling patches and, (8) increased severity and destructive potential of wildfires.

Populus tremuloides--quaking aspen

Fire ecology or adaptations

Plant adaptations to fire

Fire has played a significant role in the establishment and maintenance of aspen forests (Brown and Simmerman 1986). Although aspen appears to represent climax vegetation in parts of the

West, in many areas it grows as a seral species which depends on fire or other major disturbance for regeneration. Past fires are believed to be responsible for the abundance of even-aged stands in many parts of western North America (DeByle et al. 1987). Mature or decadent aspen stands now predominate thoughout most of the western United States, and vigorous stands are becoming increasingly rare (Jones and DeByle 1985). Many researchers attribute the decline of aspen to increased fire suppression, wildlife browsing, and grazing, which has inhibited suckering and created reductions in fine fuels.

Fire regime

Many believe that the majority of aspen communities probably only burned naturally under fairly extreme fire conditons (Brown and DeByle 1989). Mixed stands made up of aspen and conifers probably burned at more frequent intervals, with fire frequencies increasing as conifers replaced the aspen through natural succession (Brown and DeByle 1989). The natural stand replacement fire interval in many western aspen/mixed conifer or spruce/fir communities was approximately 70 to 200 years (Covington et al. 1983). Low-intensity fires may have occurred at 2 to 5 year intervals in some western, lower-elevation aspen-bunchgrass communities (Covington et al. 1983). Evidence suggests that in central Utah and possibly elsewhere in the Intermountain West, aspen may have burned with low severity at 7 to 10 year intervals prior to 1885 (DeByle et al. 1987). Natural fire in aspen forests of the Northeast ranged from 26 to 100 years (Rouse 1986). Current fire frequency in many western stands is equivalent to only one fire in every 5,000 years (Brown 1985a). Research indicates that fire frequencies of 100 to 300 years are necessary for the regeneration and maintenance of many aspen communities (DeByle et al. 1987). Many references have been made to the comparative lack of fire in aspen communities. Some researchers believe that most of the flammable mixed conifer-aspen stands which burned in the late 1800's may only now be reaching critical fuel levels (Brown 1985a, DeByle et al. 1987).

Aspen has often been referred to as a "firebreak species." Fires in aspen are generally infrequent, slow-spreading, and of relatively low-intensity (Bevins 1984; DeByle et al. 1987). Fires which burn thousands of acres of surrounding forest often burn less than a few yards into adjacent aspen communities (Jones and DeByle 1985). Crown fires in coniferous forests have reportedly dropped to the ground upon reaching aspen, and have subsequently been extinguished (Jones and DeByle 1985). Many factors influence the flammability of aspen communities including: slope, the amount of downed woody material present, grazing history, fuel moisture, fuel loading and distribution, weather, and aspen crown closure (Brown and Simmerman 1986).

Fire generally stimulates the production of aspen suckers and the rejuventation of the stand. However, the effect of fire may be somewhat variable in aspen communities when the broad elevational and climatic range of this species is considered (Covington et al. 1983). Although the thin-barked stems are often killed by even fairly light fires, the roots are well-protected through the insulation of overlying soil layers. Sapling size or smaller aspen can also regenerate through root crown or stump sprouting. These modes of regeneration promote survival and rapid growth after fire or other disturbance. Although regeneration through seed is uncommon in drier western North America, this species can regenerate through seed following fire in the Great Lake states and elsewhere in eastern North America where moisture regimes are more favorable for seed germination and subsequent development. Fire may also play a significant role in

controlling viruses and other pathogens in this clonal species (Schier 1975b). The removal of diseased trees may slow the rate at which pathogens are transferred to new suckers producing a much healthier stand (Schier 1975a).

Aspen is described as a "good competitor," and in many areas, soon forms the dominant vegetation after fire, even where pre-burn vegetation may have consisted of only scattered aspen in a coniferous stands (DeByle et al. 1987; Schier and Winokur 1985). In parts of the southern Rocky Mountains, aspen can survive in a suppressed state as an understory species in coniferous stands for relatively long periods of time, and can readily colonize burned sites through root suckering (Parker and Parker 1983). In some instances light surface fires can apparently retard conifer regeneration while stimulating aspen suckering (Parker and Parker 1983).

The time required for aspen communiites to revert back to a pre-burn condition has not been well-documented. In many lightly burned communities, researchers believe that pre-burn conditions may be reached within 10 years (Bartos and Mueggler 1981). Several decades may be required for more heavily burned areas to reach a pre-burn condition (Bartos and Mueggler 1981). Many factors including the age and health of aspen, and site characteristics such as soil, and topography can significantly influence the rate of succession in aspen communities.

***Immediate fire effect on plant**:*
Aspen is thin-barked and extremely sensitive to all but the coolest ground fires (Jones and DeByle 1985). Even fairly low-intensity fires can seriously damage or kill aspen. The effects of fire on aspen depend on such factors as the amount of fire-caused injury, season of burn, bark morphology, size, vigor, and form of the tree, clonal variation and fire intensity and severity (Rouse 1986). The probability of mortality is closely related to char height and the circumference charred (Brown and DeByle 1987).
Fire can damage the cambium, buds, or leaves, and when the living tissue is heated to approximately 147 degrees F (64 degrees C), mortality occurs immediately (Rouse 1986). The length of time the tree is heated is reportedly the primary factor influencing cambial damage. Fires of long duration generally induce mortality in aspen (Brown and DeByle 1987). Experimental tests indicate that aspen tissue dies when heated to approximately 120 degrees F (49 degrees C) for a period of one hour (Rouse 1986).

The exact causal mechanism of death to aspen following heat injury to boles is poorly understood (Brown and DeByle 1987). Often, mortality does not occur immediately after fire. Occasionally buds in the crown will survive at least temporarily, and leaf out prior to the death of the tree (Brown and DeByle 1987). This might indicate that at least in some cases, heat injury can immediately disrupt the function of phloem tissue, but can allow xylem to function so that food reserves generate a new flush of foliage (Brown and DeByle 1987). Brown and DeByle (1987) reported that following several fires in Wyoming and Idaho, aspen continued to die over a four year period, although most individuals succumbed by the second post-fire year. Even when aspen is not killed outright by the effects of fire, the bole may be sufficiently damaged to permit the entrance of wood-rotting fungi (Kovalchik 1987). According to Jones and DeByle (1985), basal scars which lead to destructive heart rot can be made on even large aspen by "the lightest of fires." Basal fire scars may also permit entry of borers and other insects which can further weaken the tree (Brinkman and Roe 1975).

Larger aspen are generally less severely damaged by fire than are smaller trees (Rouse 1986). In addition to having slightly thicker, and thus more protective bark, these larger trees have more cambium, some of which may survive and continue to function, even when parts of the tree are killed (Rouse 1986). Bark thickness at breast height and at 4 inches (10 cm) above the ground, correlates well with d.b.h., and resistance to fire-induced mortality (Brown and DeByle 1987). Brown and DeByle (1987) found that as d.b.h. increases beyond 6 inches (15 cm), aspen becomes more and more resistant to fire-induced mortality. Trees of small diameters also generally die more quickly than do large trees when damaged by fire (Brown and DeByle 1987). A greater length of time is required for trees of larger d.b.h. to reach lethal temperatures (Brown and DeByle 1987). Research suggests that larger diameter trees died at significantly higher rates when burned by moderately severe fires than when burned by fires of lower severity (Brown and DeByle 1987).

Growth form and vigor also significantly influence the effect of fire on aspen. Less vigorous trees tend to heal more slowly than do healthier trees (Rouse 1986), and may be rendered more vulnerable to subsequent insect infestation or disease. Aspen which lean typically receive greater heat at the bark surface than do straighter trees when flames are generated directed below the stem (Rouse 1986).

The position of an individual tree on a slope, or within a stand, can also influence the degree of damage caused by fire. Even when damaged, trees located near the boundaries of a fire can often maintain a live crown (Brown and DeByle 1987). These peripheral trees may continue to receive food supplies from the roots of their unburned neighbors (Brown and DeByle 1987). Aspen growing on slopes generally show greater damage than do trees on flatter areas. According to Brown and DeByle (1987), flames moving uphill often curl up the lee side of trees when fanned by upslope wind, and produce charring to a greater height. The effect of slope is particularly pronounced (up to 31-44% higher char heights) after fires of higher severity (Brown and DeByle 1987). This relationship is presented in the following table:

	probability of mortality	
d.b.h	0.90	0.95
Average char height		
10	5	12
15	14	21
20	23	30
25	32	39
Uphill char height		
10	6	16
15	19	29
20	31	42
25	44	55

(Brown and DeByle 1987).

Evidence suggests that in most cases, aspen is more often killed by fires of increasing intensity and severity. Flame length, flame height, duration of heat received, and the type of fuels all greatly influence the effect of fire on aspen. Fires in aspen communities generally tend to be of relatively short duration, and of low- to moderate-intensity (Brown and DeByle 1987). Intermediate size fuels, 4-3 inches (1-8 cm in diameter), are sparse in many aspen communities (Brown and DeByle 1987). Jones and DeByle (1985) report that crown fires running through conifers are often extinguished after burning into aspen for only a few yards.

In general, aspen appears to be less susceptible to injury when burned or otherwise damaged, during winter dormancy (Rouse 1986). Research indicates that a series of burns at 2 to 3 year intervals may be especially damaging to aspen, and can in some instances reduce or eliminate resuckering capabilities (Brinkman and Roe 1975, Jones and DeByle 1985).

Discussion of fire effect

Aspen mortality is believed to be related to fire intensity and severity. Fire severity refers to the overall effect of fire on the ecosystem and describes the transfer of heat downward (Brown and DeByle 1987). Fire intensity, or the upward movement of heat is often described by fire line intensity and relates empirically to flame length (Brown and DeByle 1987). Fire severity as it relates to aspen mortality is as follows:

1) high severity fire - a) litter and duff completely consumed, mineral soil exposed and possibly altered; b) shrubs killed, foliage and stems < .4 in (1 cm) in diameter consumed; c) aspen stems charred for at least 3 ft (1 m) in height, all stems apparently killed by fire.

2) moderate severity fire - a) litter and fine woody material consumed, some fine stem wood charcoal remains, duff deeply charred or consumed; b) shrubs killed and 40 to 80% of their canopy consumed; c) most aspen stems charred but to a lesser degree than severe burn, majority appear to be killed.

3) low severity fire - a) leaf litter charred but not completely consumed; b) unburned litter and patches of unburned vegetation may remain; c) shrubs partially killed, some leaves and twigs remain; d) some aspen charred, but less than 1/2 stems appear to be killed (Brown and DeByle 1987).

High-intensity fires in Wyoming and Idaho resulted in the death of all aspen immediately after, or within the first year following fire (Brown and DeByle 1987). The number of trees that had died increased noticeably from the first to second years and increased slightly during the third and fourth years after fire (Brown and DeByle 1987). Aspen mortality was as follows after fires described as "of low to moderate severity" in Ontario:

1) low-intensity fire - approximately 1/2 of the aspen canopy killed, 1/4 damage and partially defoliated.

2) moderate-intensity fire - all aspen stems making up the canopy were killed, ground was scorched and duff burned off locally (Horton and Hopkins 1966).

Typical injuries following fire include charring and browning of bark. Both have been correlated with fire severity and subsequent mortality (Brown and DeByle 1987). According to Brown and DeByle (1987), the amount of circumference charred may be thought of as an indirect indicator of the heat pulse causing mortality to a particular aspen.

Char height, which reflects flame height and fire intensity, also correlates well with aspen mortality (Brown and DeByle 1987). Char heights of 12 inches (30 cm) were found to have a 90% probability of killing aspen greater than 10 inches (25 cm) d.b.h. Often, greater char heights were produced where head fires were aided by high winds, or where steep slopes allowed strong runs (Brown and DeByle 1987). Generally reduced char heights are produced during flanking and backing fire and short head fire runs (Brown and DeByle 1987). Char height as it relates to slope position and fire severity, is presented in the following table:

Char height						
post-burn	Uphill			Average		
year	Low	Moderate	High	Low	Moderate	High
Dead trees						
1	49	46	152	34	34	116
2	43	43	152	30	34	116
3	43	43	152	30	30	116
43	43	152	30	30	116	
Live trees						
1	22	34	76	17	23	46
2	17	16	--	14	10	--
3	18	17	--	14	10	--
4	11	16	--	9	10	--

(Brown and DeByle 1987).

Brown and DeByle (1987) reported that flame heights of 4 inches (10 cm) were required to kill aspen of 4 inch (10 cm) d.b.h. with 90% probability in the Wyoming-Idaho study area, whereas flame heights of at least 24 inches (60 cm) were required to kill trees of 10 inch (25 cm) d.b.h. Average flame heights of 18 - 24 inches (45-60 cm) were necessary to kill most aspen at this study site (Brown and DeByle 1987).

Brown and DeByle (1987) found that aspen bark was killed to the cambium if the tree received a flame height of 30 inches (75 cm), as well as on 12 inch (30 cm) fires lasting 3 minutes or more. Bark was not completely killed to the cambium on some trees receiving 72 inch (30 cm) flame heights of 1 to 2 minute duration, however. Brown and Simmerman (1986) noted that flame lengths of at least 1.7 to 2.1 ft (0.5-0.6m) are needed to actually kill aspen trees. Trees may be killed at fireline intensities of 18 to 28 Btu/ft/s (Brown and Simmerman 1986).

Discussion of fire effect

Plant response to fire

The specific response of aspen to fire depends on such factors as site characteristics, fire intensity, genotype, season of burn, and stand vigor. Generally, however, aspen responds to fire by vigorous root suckering when stems are killed. A number of researchers report that in many areas, it is almost impossible to prevent root suckering through burns of even high intensities (Horton and Hopkins 1966). Aspen frequently responds as a vigorous colonizer, and can form the dominant post-burn species on some sites where it was only a minor constituent of pre-burn communities (Brinkman and Roe 1975). Although regeneration by seed occurs following fire in some moist areas of eastern North America, suckering is the predominant mode of reestablishment throughout most of its range. Reestablishment in western North America is essentially restricted to vegetative means (Covington et al. 1983).

Soil temperatures are believed to exert an important influence on sucker production in aspen. The darkened soil surface created by fire warms up earlier in the spring and may stimulate chemical activity in the roots which makes stored food reserves more available for suckering (Covington et al. 1983). Elevated soil temperatures produced by fire may serve as a stimulus for root suckering (Horton and Hopkins 1966). Fire also generally kills phloem tissues thereby interrupting the downward flow of auxin, and removing apical dominance (Brown 1994). Cytokinin is no longer moved upward in the xylem and root concentrations increase, promoting suckering (Brown 1994).

Aspen roots are generally well-protected from fire by insulating layers of soil. Many researchers believe that aspen is unlikely to sustain significant root damage from fire (Gruell and Loope 1974; Tucker and Jarvis 1967). Horton and Hopkins (1966) contend that a long drought, combined with large accumulations of heavy slash, is necessary to produce soil temperatures high enough to kill most roots. In many areas, a layer of 0.5 inch (1cm) of soil can insulate aspen roots from high surface temperatures (Horton and Hopkins 1966). The insulative value of soil also increases with greater soil moisture content (Horton and Hopkins 1966). Soil moisture contents greater than 25% can apparently protect almost all aspen roots from lethal temperatures (Horton and Hopkins 1966). Other researchers report than some severe fires may, on certain sites, damage or kill shallow aspen roots (Parker and Parker 1983). Most suckers develop from

fairly shallow roots and some high-severity fires may damage enough of the roots to limit the number of potential suckering locations (Brown and DeByle 1987).

Research indicates that fires of moderate-intensity are most effective in promoting suckering (Bartos and Mueggler 1979; Bartos and Mueggler 1981; Brown 1985b; Horton and Hopkins 1966). Suckering also occurs after fires of high-intensity, where fuel levels are sufficient to generate such fires. High-intensity fires generally produce fewer suckers than do moderate-intensity burns, but more than those produced following low-intensity burns (Brown 1985b). Some researchers report that suckers which develop after high-intensity burns, exhibit less vigor than those produced in response to fires of lower intensities (Horton and Hopkins 1966). Light burns with low ground char typically produce patchy or sparse suckering (Brown 1985b). Low-intensity burns often fail to reduce the competing understory species, thus hampering aspen regeneration (Horton and Hopkins 1966). Light fires in aspen result in stand thinning and the development of an all-age stand (Brown 1985b). Fires of higher intensities more often produce even-aged stands (Brown and Simmerman 1986).

Sucker density varies according to root vigor and abundance, root depth, hormone levels, carbohydrate reserves, genotype, soil temperature, and light (Brown 1985b). During the first few years after fires, sucker densities have ranged from 15,000 to 150,000 per ha (Brown and DeByle 1989). Most suckers develop within 2 years after fire (Brown and DeByle 1989). Sucker densities may increase as suckers expand, or extremely high levels of suckers may be thinned through natural morality (Brown 1985a). Mortality often reduces the number of suckers after the first year (Brown and DeByle 1987). When aspen stands are burned during the first half of the growing season, many suckers develop later in the year, while others are produced the following year (Brown and DeByle 1987). Aspen stands burned during late summer or fall develop suckers the following growing season (Brown and DeByle 1987).

When the above ground portion of aspen is killed by fire, the energy reserves present in the clonal root system contribute to the vigor and number of root suckers produced (Olmsted 1979). Aspen of low vigor often tend to sucker less than do more vigorous trees (Rouse 1986). Newly formed bud primordia produced by healthy roots give rise to the most suckers (Brown 1994).

Aspen burned early in the growing season generally sucker later in that same season and again the following year (Brown and DeByle 1987). Aspen burned in mid-season generally produce suckers just prior to the end of the growing season, and trees burned later in the season when dormant produce suckers the folowing year.

Evidence suggests that repeated consecutive burns can significantly reduce or eliminate aspen.

The following reductions in aspen sucker production were noted after repeated spring burns in Minnesota:

# of spring burns	% aspen reduction
2	68
3	86
4	94

(Jones and DeByle 1985)

According to Brinkman and Roe (1975), a series of fires at 2 to 3 year intervals can eliminate the suckering ability of the aspen's root system. Repeated burns can impair moisture absorption and retention when the humus layer is reduced or removed (Brinkman and Roe 1975). Annual burning in the aspen parklands of east-central Alberta, has reportedly slowed or stopped the invasion of aspen into adjacent grasslands (Anderson and Bailey 1980). In many areas, reburning may be very difficult if not impossible due to the slow rate at which litter accumulates (Horton and Hopkins 1966).

Aspen can also regenerate through stump and root collar sprouting, although these modes of regeneration are relatively uncommon since the bases of the parent stems are often scorched (Horton and Hopkins 1966). Regeneration following several Ontario wildfires was as follows:

	Root suckers		Stump and collar sprouts	
	Light burn	Moderate burn	Light burn	Moderate burn
Total # of stems	800	1,323	229	189
Avg. ht (ft) of tallest stem	2.8	3.3	2.6	2.4

(Horton and Hopkins 1966).

In some portions of the east, aspen can reestablish a site through wind-dispersed seed. Burning results in exposed mineral soil which serves as an effective seedbed (Godman and Mattson 1976). Severe burns in parts of Canada can create especially favorable sites for seedling establishment (Horton and Hopkins 1966). In northeastern Wisconsin, aspen seedlings sometimes make up as much as 20 to 35% of all seedlings present 6 years after fire on sites without an aspen overstory (Godman and Mattson 1976).

Trees damaged by fire are more vulnerable to disease or insect infestation. In some areas root rot (*Armillaria* sp.) may become a significant problem in fire-weakened stands (Rouse 1986). However, fire can also be effective in eliminating some insects, such as cutworms, which overwinter in duff (Rouse 1986), and other pathogens such as certain viruses (Schier 1975b).

New aspen suckers are a nutritious and readily available food source for many large ungulates. In some areas, aspen renegeration may be seriously impacted by large populations of elk (Bartos and Mueggler 1981). Smaller burns are particularly vulnerable to elk damage when elk numbers are high. Livestock also tend to concentrate on newly burned areas while feeding on the succulent aspen sprouts, and can inhibit or prevent regeneration in some instances.

Discussion of plant response

Aspen root suckers generally develop on roots at approximately 0.08-5 inch (0.2-13cm) depth (Brown and DeByle 1987). A number of researchers have observed that suckers tend to develop on deeper roots after fires of high severity, with greater severity increasing the depth of development (Schier and Campbell 1978, Schier et al. 1985). Schier and Campbell (1978) reported that depth of root-suckering ranges from 0 to 9 inches (0-23cm) as follows:

	suckers				parent roots			
	mean #/point	age (yrs.)	number	mean Hgt. (cm)	% with advent. roots	mean depth (cm)	mean dia (mm)	% with new roots
site 1	1.47	1 2	74 126	47.2 56.0	65.5	7.8	17.2	37.5
site 2	2.02	1 2	158 42	66.2 87.6	49.5	7.2	12.4	50.5

(Schier and Campbell 1978).

At a second study in Wyoming and Idaho, suckers originated at an average depth of 18 inches (45.5 cm) following low severity fires, and at a depth of 2 inches (5.89 cm) after moderate burns (Brown and DeByle 1987). Jones and DeByle (1985) reported an average suckering depth of 4 inches (10 cm) after a severe burn, and an average depth of 2.5 to 3 inches following a burn of moderate severity. Depth of suckering is also influenced by fire severity as reflected in litter consumption (Brown 1994). A Wyoming study revealed that average suckering depth was greatest where all litter was consumed by fire as illustrated below:

Litter consumed (%)	Average sucker depth (cm)
0-10	6.0
10-90	7.3
90-100	10.7

(Brown 1994)

Exact reasons for this relationship have not been documented, although several hypotheses have been proposed. Deeper roots may sucker more readily because relatively greater numbers of shallow roots have been killed by heat, or litter removal may create a warmer soil surface which promotes suckering (Brown 1994). Aspen root suckering in the west, tends to occur at deeper depths than in eastern North America (Schier and Campbell 1978; Schier et al. 1985). Interestingly, fires in many aspen communities of eastern North America tend to be of low-intensity and relatively short duration (Rouse 1986).

A number of studies have suggested that root suckering is related to fire severity, with moderate-intensity fires producing the most prolific suckering. Temperatures of approximately 150 degrees F (66 degrees C), are often considered to be lethal to aspen root tissue (Horton and Hopkins 1966). However Horton and Hopkins (1966) reported that temperatures of 165 to 170 degrees F (74-77 degrees C), when attained at the root surface, cause only minimal damage and

did not appear to affect suckersing. Horton and Hopkins (1966) report the following levels of root suckering following burns of varying intensities:

1st year suckering	control	light burn	intense burn
# per acre	1,100	5,000	3,900
frequency %	37	94	100
mean max. ht. (inches)	16	20	10

	uncut plots			cut plots		
	control	light burn	moderate burn	control	light burn	moderate burn
# per acre	300	7,000	16,000	6,800	15,800	20,500
freq. - %	8	54	87	50	79	83

(Horton and Hopkins 1966).

Bartos and Mueggler (1979) observed suckering after fires of several intensities in western Wyoming. Suckering decreased after the first post-burn year to 1/3 of the pre-burn numbers by the third year after fire. This high-intensity burn yielded 30,000 stems per hectare, or approximately twice as many suckers as the pre-burn levels (Bartos and Mueggler 1979). Burns of moderate-intensity produced 27,000 per hectare the first year after fire, 66,000 per hectare the second year after, with natural thinning to 30,00 per hectare by the third year after fire (Bartos and Mueggler 1979). However, Brown and DeByle (1987), reported that in a Wyoming-Idaho study, sucker density was not significantly related to fire severity. The average sucker density before burning was 3,710 per hectare, but averaged 24,700 per hectare by the end of the first post-burn year (Brown and DeByle 1987). Bartos (1979) found that in the central Rocky Mountains, light intensity burns did little to stimulate sucker production. Moderate to high-intensity burns produced short-term increases in suckering but soon returned to unburned control levels (Bartos 1979).

Fire management considerations

Increased fire suppression and heavy grazing have contributed to the predominance of mature and decadent aspen stands in the western United States. Young, vigorous aspen communities are now relatively uncommon (Jones and DeByle 1985). Most aspen communities require a major disturbance such as fire for their maintenance. The value of prescribed fire as a tool for regenerating decadent aspen stands is now widely recognized. In many areas of the West, burning is both the most economical, and environmentally sound means of maintaining aspen. Brown and DeByle (1989), note that fire is "the only tool that is likely to be applied on perhaps a third of the aspen acreage in the West." Prescribed fire, when applied correctly, can regenerate many decadent aspen stands creating a habitat mosaic of particular value to wildlife and short-term increases in forage for domestic livestock.

The use of prescribed fire has been somewhat limited in the past due in part, to the widely held belief that aspen is difficult to burn (Brown and Simmerman 1986). Often, pure aspen stands burn well only under extreme conditions when the danger of fire spreading into adjacent, more flammable timber types is great (Brown and DeByle 1989). Burning may be limited to only a

few days a year, and in some years there may be no opportunity to burn (Brown and Simmerman 1986).

Elements necessary for a successful burn include: 1) adequate loading of fine fuels in both the 1 and 10 hour timelag classes, 2) continuity of fine fuels, 3) live fuels must be cured with an average herbaceous fuel moisture content of 50% or less, 4) and an appropriate method of ignition must be selected and applied (Brown and DeByle 1989). Fuels and flammability are highly variable in aspen types (Brown 1985b, Brown and DeByle 1989). The flammability of aspen communities may also be related to the age and health of the stand. Generally, flammability increases as the community starts to break up and fuels increase (Rouse 1986). In some aspen communities the probabilty of a successful burn is quite high while in others, the chances of a good fire is low and alternate means of promoting regeneration such as mechanical treatment, should be selected (Brown 1985b). Fire may sometimes be combined with partial cutting to create conditions favorable for aspen regeneration (Schier et al. 1985). Fire in aspen logging slash of 10 tons per acre (22t/ha), killed residual overstory and created conditions conducive for aspen regeneration (Schier et al. 1985). Fire creates a favorable seedbed for germination of aspen seed in eastern North America where rainfall regimes permit regeneration through seed.

Evidence suggests that fires of moderate-intensity and ground char are most effective in promoting aspen sucker formation (Bartos and Mueggler 1979; Brown 1985a). Brown (1985a), reports that where aspen regeneration is the primary objective of a prescribed fire, managers should attempt to kill all or at least most of the aspen in a stand. At least 80% should be killed for optimum suckering (Brown and Simmerman 1986). The mosaic of burned and unburned vegetation often produced when fires of "adequate intensity " occur in the treatment areas, creates a diversity of aspen age classes (Brown 1985a).

The correct identification of aspen fuel types capable of sustaining spread is critical to the success of a burning program (Brown and DeByle 1987). Recent research conducted through the Intermountain Fire Sciences Laboratory and elsewhere, has contributed to a better understanding of the conditions necessary for successful burns in aspen types. The publication "Appraising fuels and flammability in aspen, a prescribed fire guide," by J.K. Brown and D.G. Simmerman, (1986), serves as a useful guide for appraising flammability in aspen types and should be reviewed by managers interested in conducting prescribed burns in aspen communities. Brown and Simmerman (1986), have grouped aspen communities into 5 fuel types: 1) aspen/shrub, 2) aspen/tall forb, 3) aspen/low forb, 4) mixed (aspen/conifer)/shrub, 5) mixed/forb.

Aspen/shrub is described as the most flammable type, with mixed/shrub the next most flammable, and aspen/low forb and mixed/forb the least flammable of the five types (Brown 1985a). Aspen/tall forb is moderately flammable, having only half the estimated fire intensity potential of the aspen/shrub class (Brown 1985a).
Several important differences exist between fire in mixed conifer - aspen stands and pure aspen stands. Conifers more often torch out producing a more intense fire (Brown 1994). Mixed types generally have slightly more downed woody fuels than do aspen, although the amount of downed woody fuel present can vary greatly from stand to stand (Brown 1994). Evidence also suggests

that heavy fuel loading can increase the rate of spread in mixed types but not in pure aspen types (Brown and Simmerman 1986).

According to Brown (1994), only the aspen/shrub class produces sufficient fireline intensity to support a sustained fire at a midflame windspeed of 3 km/h. Sustained spread is generally not possible in aspen types unless flame lengths exceed 1 to 1.5 ft (.3-.5m) (Brown and Simmerman 1986). Flame lengths of 1.7-2.1 ft (.5-.6m) or more for at least 1 minute are required to kill aspen (Brown and Simmerman 1986).

The timing of a burn can be critical in aspen communities. Conditions favorable for burning generally occur in late summer and fall when temperatures are still relatively warm and the understory is well-cured (Brown 1985b). Aspen may also be burned during the spring in parts of the Southwest, although herbaceous fuels still matted down from the winter's snow tend to fairly difficult to burn (Brown 1985b). Curing can significantly influence fireline intensity in some aspen communities (Brown 1994). Curing reportedly exerts a greater influence in the aspen/tall forb and aspen/shrub fuel classes than in the mixed/shrub and mixed/forb classes (Brown 1994). Curing rates depend on a number of factors. The understory in aspen stands with open canopies generally cures 2 to 4 weeks earlier than in stands with closed canopies (Brown 1994). In aspen stands, grasses typically cure faster than forbs (Brown and DeByle 1989). Precipitation increases both forb and grass moisture levels in July and August, but has little effect by September (Brown and DeByle 1989). The moisture content of understory shrubs in aspen, generally remains high until the leaves change color in the fall (Brown and DeByle 1989). Hard frosts often cure all understory vegetation quickly (Brown 1994). Fire in aspen communities spreads slowly when the lower 2 inches (5cm) of grass greens up (Brown and DeByle 1989).

Grazing can reduce fire potential by 80 to 90% of ungrazed conditions and can reduce predicted fire intensity and rate of spread by at least 1 rating level (Brown and Simmerman 1986). Many researchers recommend eliminating grazing 1 year before carrying out prescribed burns (Brown 1994). Grazing and browsing can also significantly harm aspen regeneration following fire on many sites. Grazing should generally be deferred after fire in aspen. Up to three years are required for aspen to grow beyond the reach of sheep, and at least 4 to 5 years are required for aspen to grow out of reach of cattle (Brown and Simmerman 1986). Elk browsing can also inhibit regeneration where populations are large (Bartos and Mueggler 1981). Suckering success can be maximized by burning larger areas or by burning clusters of smaller areas where elk browsing is intense (Brown and Simmerman 1986).

Areas of mineral soil exposed by pocket gophers can reduce fire potential when rodent populations are fairly large (Brown 1994). Higher winds may be needed to carry fire in these areas with discontinuous surface fuels.

The type of ignition should be carefully chosen for maximum success. Hand-held and aerial ignition have both been used. Helitorch ignition can sometimes compensate for problems with fuel loading or moisture content (Brown and DeByle 1989). Aerial ignition can enhance fire spread on marginal sites by creating larger flames (Brown and Simmerman 1986).

Herbaceous understory production generally increases, at least in the short term, after fire in aspen communities (Brown 1985b). The amount of increase and length of time required for optimum production is highly variable. Researchers have reported increases of up to 200% within 3 years of moderate-intensity fires in some aspen community types (Mueggler 1985). Fires of moderate-intensity appear to produce the greatest increases in understory production (Mueggler 1985).

Pseudotsuga menziesii var. *glauca*--Douglas-fir

Fire ecology or adaptations:

Plant adaptations to fire

Mature Douglas-fir is generally more fire resistant than spruces and true firs, equally or slightly less fire resistant than ponderosa pine (Wellner 1971). Mature trees can survive moderately severe ground fires because the lower bole is covered by thick, corky bark that insulates the cambium from heat damage (A. D. Revill Associates 1978; Fischer and Bradley 1987). It takes about 40 years for trees to develop fire-resistant bark on moist sites in the northern Rocky Mountains (Fischer and Bradley 1987). Protection offered by thick bark is ofen offset by low growing branches and flammable foliage that make trees susceptible to crowning (Fischer and Bradley 1987; Lotan et al. 1981).

Douglas-fir relies on wind-dispersed seeds to colonize burned areas where trees have been killed. Mineral soil exposed by burning provides a good seedbed. Germination of artificially sown seed was about 60 percent on burned seedbeds but only 10 percent on unburned duff (Boyce and Neuenshwander 1989). Seedling establishment begins a few years after fire and is restricted to within a few hundred yards of seed trees.

Fire regime

Dry Douglas-fir habitat types in the northern Rocky Mountains experienced low- to moderate-intensity ground fires at less than 30-year intervals (Arno 1980; Pfister et al. 1977). Where ponderosa pine is a major associate, fires at 10-year intervals were common (Lotan et al. 1981). These frequent ground fires maintained relatively open stands of Douglas-fir or, more frequently, seral stands of ponderosa pine since pine saplings are more fire-resistant then Douglas-fir saplings (Arno 1980; Fischer and Bradley 1987; Loope and Gruell 1973). Fire suppression has resulted in long fire-free periods which have allowed Douglas-fir regeneration to become wellestablished. In some areas, dense thickets have formed, which provide a continuous fuel ladder to the crown of overstory trees. Thus, fire suppression has increased the potential for severe, stand destroying wildfires.

In the southern Rocky Mountains, frequent surface fires in dry Douglas-fir habitat types maintained seral stands of ponderosa pine and/or southwestern white pine. Fire suppression has allowed the development of dense, Douglas-fir sapling thickets (Moir and Ludwig 1979). Cool and dry Douglas-fir habitat types in central and southern Utah do not experience frequent, low-intensity ground fires characteristic of the northern Rocky Mountains. These habitats typically have discontinuous ground fuels and poor grass cover which hamper fire spread (Youngblood and Mauk 1985). The mean fire interval for moist and dry Douglas-fir habitat types in southwestern Montana is about 40 to 45 years. On moist sites tree growth is rapid enough so

that some trees reach fire-resistant size before the next fire, allowing open stands to develop. Tree growth is much slower on dry sites. During pre-settlement times, Douglas-fir did not reach fire resistant size before the next fire. As a result, dry Douglas-fir habitat types were maintained as seral grasslands. Trees were restricted to rocky microsites with sparse herbaceous fuels. Fire suppression has allowed Douglas-fir to spread from these fire-safe sites and form extensive pole-sized stands in mountain grasslands (Arno and Gruell 1983).

Fire regimes in moist Douglas-fir habitat types are variable, ranging from low to moderate surface fires at relatively frequent intervals to severe crown fires at long intervals (Kilgore 1981). Large fires burn at several intensities, changing with shifts in stand structure, fuel loads, topography, and weather (Arno 1980). The result is a mosiac of fire treatments. Intense crown fires or repeat fires generally favor seral assocaites such as aspen or lodgepole pine. In the Bob Marshall Wilderness in Montana, Douglas-fir-dominated sites were converted to lodgepole pine by three fires at 30- to 40-year intervals. Another site in the same Wilderness was converted from a Douglas-fir-western larch forest to a forest dominated by lodgepole pine as a result of a single intense fire (Gabriel 1976). The mean fire interval for moist Douglas-fir climax forests in the Coram Experimental Forest in northwestern Montana is 140 years (Arno 1980).

Fire effects

Immediate fire effect on plant

The effects of fire on Douglas-fir vary with fire severity and tree size. Saplings are often killed by surface fires because their low branching habit allows fire to carry into the crown. Photosynthetically active bark, resin blisters, closely spaced flammable needles, and thin twigs and bud scales are additional characteristics that combine to make saplings vulnerable to surface fires (Fischer and Bradley 1987). Douglas-fir saplings are more susceptible to mortality from surface fires than ponderosa pine saplings (Arno and Gruell 1983; Weaver 1968).

Chance of survival generally increases with tree size. Because they have thicker bark and larger crowns, large trees can withstand proportionally greater bole and crown damage than small trees. Following a low- to moderate-intensity ground fire in an open mixed conifer stand in Colorado, 64 of 103 Douglas-fir trees died within 2 years of burning. Live trees averaged 9.5 inches (24 cm) d.b.h. and 32 feet (9.8 m) in height, while dead trees averaged 5.6 inches (14.3 cm) d.b.h. and 22.6 feet (6.9 m) in height. In general, surviving trees tended to be taller and have larger bole diameters than trees that died (Bevins 1980).

Discussion of fire effect

Crown fire

Crown fires can kill trees over extensive areas. On thoroughly burned over areas, however, scattered survivors can be expected in unburned pockets or in areas where fires moderated (Crane et al. 1983; Wellner 1971).

Ground fire

Moderate-intensity surface fire kills varying proportions of pole-sized and larger Douglas-fir trees (Wellner 1971). Crown scorch tends to be the principal cause of death, probably because ground fires intense enough to kill a tree by girdling it will also scorch the entire crown (Wyant et al. 1986). Crown scorching from summer fires is more damaging than late summer or fall fires because more buds are killed. During late summer the buds are set and subsequent-year needles are well protected (Peterson and Arbaugh 1986). Cambial damage from bole charring is the second most prevalent cause of death. Ryan et al. (1988) found that if more than 25 percent of the cambium on Douglas-fir trees at breast height was dead, there was a high probability that the tree would die. Trees damaged and weakened by fire are susceptible to insect attack. If attacked by bark beetles, fire-weakened trees often die (Peterson and Arbaugh 1986). Root damage can also affect survival. Douglas-fir frequently has shallow lateral roots that can be damaged if the organic layer burns (Ryan et al. 1988).

Fuels and fire behavior

The amount of crown and bole damage a tree receives is related to fire behavior and the abundance of understory fuels. If there are heavy fuel accumulations around the base of the tree, severe cambial damage can occur from ground fires that burn primarily in the forest litter. Trees infested with dwarf mistletoe commonly have large accumulations of dead, fallen brooms around their base. When ignited, this fine debris burns hot, charring the bole. It can also provide a fuel ladder to torch the crown (Wickner and Leaphart 1976).

Fuel type and arrangement, and related fire behavior, vary greatly in dry Douglas-fir habitat types. Where surface fuels are discontinuous, many trees survive the resultant irregular burning (Wickner and Leaphart 1976). Fire and grazing history greatly influence the fuel buildup. In northern Idaho, Douglas-fir was more susceptible to fire damage in stands subjected to years of livestock grazing than in ungrazed stands (Weaver 1968). Ungrazed stands remained open and park-like, and had a nearly continuous distribution of small fuels which carried fire well. Prescribed fires had flame lengths up to 36 inches (91 cm) but spread rapidly and only scorched the lower crowns of large trees. On grazed sites open stands were converted to dense pole stands with sparse understories and numerous sapling thickets. These stands had a greater accumulation of duff and large woody fuels which contributed little to fire spread. This resulted in a less intense but slow-spreading fire which was more damaging to trees, probably because of the long residence time, which can kill trees through cambial heating (Peterson and Arbaugh 1986). On the grazed site, numerous trees up to 4 inches (10 cm) d.b.h., and a few more than 6 inches (15 cm) d.b.h. were killed.

Plant response to fire
Seedling establishment following fire is dependent on the spacing and number of surviving seed trees. Following large, stand-destroying fires, Douglas-fir seedling establishment is slow. Seedlings are restricted to the burn edge or near surviving trees within the main burn (Crane et al. 1983).

On logged-over sites, Douglas-fir readily established after slash burning, but establishment tends to be better where Douglas-fir is a seral species, such as in grand fir or subalpine fir habitat types, and on north- and east-facing slopes (DeByle 1981; Shearer 1976). On dry, south- and west-facing slopes some shade is often needed for seedlings to survive (Hatch and Lotan 1969). Many tree associates are more dependent on mineral soil for seedling establishment than Douglas-fir. Burning may increase the percentage of associates such as Engelmann spruce, grand fir, ponderosa pine, lodgepole pine, and western larch (DeByle 1981).

Discussion of plant response
Prescribed burning
Published guides outline prescribed burning objectives and techniques for killing invading Douglas-fir in bunchgrass habitat types (Gruell et al. 1986) and for understory burning for site preparation, fuel reduction, and habitat improvement in ponderosa pine-western larch-Douglas-fir forests (Kilgore and Curtis 1987). When burning understory in pine-larch-fir forests, the Douglas-fir should be larger than 16 inches (40 cm) in diameter when fuels exceed 30 tons/acre. Heavy fuels within 6 feet (1.8 m) of the base of trees should be removed.

Tree mortality
Published models can be used to predict fire-caused mortality of Douglas-fir (Peterson and Arbaugh 1986; Ryan et al. 1988). Crown scorch, bole damage, and insect damage are the best variables for determining mortality/survival.

Duff reduction
In larch-Douglas-fir forests in western Montana, broadcast burning in clearcuts or in standing timber can be safe and practical when small diameter fuel (less than 4 inches (10 cm) moisture content is between 10 and 17 percent (Norum 1981). Below 10 percent moisture content, fire behavior may become extreme and control problems arise; above 17 percent fuels are difficult to ignite. When burning between these moisture content values, duff consumption can be predicted by measuring the moisture content of the lower duff and the quantity of small fuels (Norum 1981). Shearer (1981) found that when the lower duff is below 50 percent moisture content, almost all the duff is burned, but when above 100 percent moisture content, less than half the duff burns.

Fire behavior of logging slash
Fresh, cured coniferous logging slash is generally very flammable because of its characteristic loose arrangement and high percentage of needles and twigs. Flammability decreases with time as needles drop to the forest floor and as a result of compaction by winter snow. In experimental burns with 32.5 tons of slash per acre (80 tons/ha) and similar relative humidities (52-70%), the rate of spread in fresh, cured Douglas-fir logging slash was 20.7 seconds/foot, while the rate of spread in 1-year-old slash was 70 seconds/foot (Fahnestock 1960).

Upland Forest Shrubland

Acer glabrum--Rocky Mountain maple

Fire ecology or adaptations
Plant adaptations to fire
Rocky Mountain maple is well adapted to fire. Following aerial crown kill, root crowns often produce numerous sprouts, sometimes over 100 per plant (Leege 1968; Leege and Hickey 1971; Lyon 1966; Lyon and Stickney 1976; Mueggler 1965). This long-lived seral species often persists in the understory of late seral or climax coniferous stands. Its sprouting ability allows it to become part of the immediate post-fire community when the conifer overstory is removed or killed (Mueggler 1965; Steele and Geier-Hayes 1989).

Fire effects
Immediate fire effect on plant
Most fires top-kill Rocky Mountain maple (Crane 1982; Lyon 1971). Hot fires, which effectively transfer heat below the mineral soil surface, may damage Rocky Mountain maple root crowns and thus prevent sprouting of some plants (Stickney 1981).

Discussion of fire effect
Many researchers report that Rocky Mountain maple sends up numerous root crown sprouts following fire (Crane 1982; Leege 1968; Leege and Hickey 1971; Lyon and Stickney 1976). However, Stickney (1981; 1985) reports that after a severe August wildfire in northwestern Montana, some Rocky Mountain maple plants survived, but suffered high mortality and contributed little cover to the early seral vegetation. He describes this maple as a "fire sensitive plant", noting that survival and response is reduced by hot surface fires which effectively transfer heat below the mineral soil surface and thus damage plant regenerative parts.

Plant response to fire
Studies in northern Idaho brushfields show that Rocky Mountain maple sprouts 4 to 8 weeks after spring prescribed burns (burning when plants are still in winter dormancy) and sprouts the following spring after fall burning (burning when leaves are changing color or have been shed) (Leege and Hickey 1971). Following a high-intensity, stand-destroying wildfire in western Montana, Rocky Mountain maple sprouted by fall of the same year in ravines and along roadcuts which received some upslope moisture (Crane et al. 1983).

After spring prescribed burning in northern Idaho brushfields, plants averaged over 100 root crown sprouts per tree, while fall burning stimulated slightly fewer, but larger, sprouts (Leege 1968; Leege 1969; Leege and Hickey 1971). Other researchers report numerous sprouts 1 year following summer fires,(Crane and Fisher 1986). This maple often takes on a tall and spindly form in late seral and climax coniferous stands. Therefore its crown volume often remains lower for several years after burning since plants assume a low and bushy form resulting from multiple root crown sprouts from 1.5 to 6.5 feet (0.5-2 m) tall (Crane et al. 1983; Leege and Hickey 1971;

Lyon 1971; Miller et al. 1988). Following a moderately severe prescribed burn in August, in a Douglas-fir forest of south-central Idaho, Rocky Mountain maple sprouts averaged 1.8 feet (0.5 m) tall after one growing season and were 5.8 (1.7 m) feet tall after 7 years (Lyon 1971). Thus with a smaller and bushier growth form, Rocky Mountain maple had a much lower crown volume following burning. Similarly, 5 years after a high-intensity, stand-destroying July wildfire in western Montana, Rocky Mountain maple shrub volume was only 40 percent of that on nearby unburned sites (Crane et al. 1983).

Discussion of plant response

Following a severe August wildfire in northwestern Montana, Rocky Mountain maple had low survival, slow redevelopment, and its percent cover remained well below pre-burn levels (Stickney 1981; 1985). Several other researchers, however, report that this maple sends up numerous root crown sprouts following fire (Crane 1982; Leege 1968; Leege and Hickey 1971; Lyon 1966). This difference in response may possibly be due to differences in fire severity, soil moisture, and season of burning. Fall and spring prescribed burning in Idaho, when plants are dormant, consistently results in prolific sprouting (Leege 1968; Leege 1969; Leege and Hickey 1971). Stickney (1981) reports that Rocky Mountain maple survival and response is reduced by hot fires which effectively transfers heat below the mineral soil surface and thus damages plant regenerative parts. Researchers in British Columbia have found that low temperature fires result in tall root crown sprouts of 6.5 feet (2 m), but that hot burns stunt root crown sprout growth, resulting in sprouts that are only 20 inches (50 cm) tall (Miller et al. 1988).

Fire management considerations

Rocky Mountain maple is a highly valued big game browse species. To improve winter range conditions, prescribed burning can be used to stimulate sprouting of plants that are decadent or have grown out of reach of browsers (Leege 1968; Leege 1969; Leege and Hickey 1971; Miller et al. 1988). Within 3 years after prescribed burning of seral brushfields in northern Idaho, Rocky Mountain maple sprouts were over 7 feet (2.1 m) tall and thus out of browsing animals' reach (Leege 1969). Conversely, in a Douglas-fir forest in south-central Idaho, Rocky Mountain maple sprouts from a summer prescribed burn, took 7 years to reach 5.8 feet (1.8 m) in height (Lyon 1971). Although nutritional analysis of Rocky Mountain maple sprouts from burned plants versus unburned plants shows only slight improvements in the quality of the forage, elk show a preference for feeding on the sprouts of burned plants and will eat larger twigs from burned versus unburned plants (Asherin 1973; Leege 1969).

Rocky Mountain maple responds well to both fall and spring burning; however, fall burning eliminates the food supply on winter ranges during the first winter, as plants do not sprout until the following spring. Prescribed spring burning in northern Idaho at 5-year intervals has not adversely affected this species (Leege 1979).

Cercocarpus montanus--mountain-mahogany

Fire ecology or adaptations
Plant adaptations to fire
Mountain-mahogany is usually temporarily damaged by fire (Ralphs et al. 1975). Mountain-mahogany burns less readily than other shrubs (Lindenmuth and Glendening 1962; Pase and Lindenmuth 1971) and sprouts vigorously from the root crown after most fires (Bradley et al. 1991; Crane 1982; Pase and Lindenmuth 1971).

In open, dry habitats where mountain-mahogany is likely to occur, fires in pre-settlement times were of low severity because of fuel discontinuity. Today, many formerly open stands are dominated by conifers and decadent shrubs which provide greater fuel loads. When fires occur, they are likely to be more severe (Bradley et al. 1991).

Fire effects
***Immediate fire effect on plant*:**
Fire generally top kills mountain-mahogany (Pase and Lindenmuth 1971). Live crowns of plants occurring in Arizona chaparral dominated by shrub live oak (*Quercus turbinella*) and mountain-mahogany were reduced by 88.5 to 99 percent during prescribed fires of various intensities.

Discussion of fire effect
Plant response to fire
Mountain-mahogany sprouts vigorously after fire. Pre-fire canopy cover consisted mainly of ponderosa pine; tree foliage was either completely scorched or consumed by the fire in the two stands where true mountain-mahogany occurred (Potter and Foxx 1979).

Mountain-mahogany seedlings may establish after fire, although seedling establishment may be relatively low. One year after a prescribed fire in Arizona chaparral, seven mountain-mahogany seedlings per acre were observed (Pase and Lindenmuth 1971). Five seedlings per acre emerged in post-fire year 2. No seedlings emerged in post-fire years 3, 4, or 5. Seedling mortality may be high.

Discussion of plant response
The response of mountain-mahogany to fire may vary seasonally. High- and low-severity fire treatments applied to mountain-mahogany during the dormant season in north-central Colorado were more effective in increasing biomass production than those applied during the growing season (Young and Bailey 1975).

Juniperus communis ssp. *alpinus*--common juniper

Fire ecology or adaptations
Plant adaptations to fire
Common juniper is generally described as "susceptible" to fire (Crane 1982; Hoffman and Alexander 1987). The degree of damage received increases with progressively greater fire intensity. Where common juniper is killed by fire, some seed presumably survives in the soil on-

site and germinates when conditions become favorable. Other seed is brought to the site through birds (Crane 1982) or, perhaps less commonly, mammal dispersal agents. A relatively long germination period may be required (Crane and Fischer 1986), and germination is frequently poor (Dietz et al. 1980; Pack 1921). These factors contribute to slow post-fire reestablishment on many sites. Although relatively little documentation exists on the post-fire successional role of common juniper, this species most often becomes prominent in later stand ages of certain lodgepole pine forest of Colorado (Clagg 1975).

Fire effects

Immediate fire effect on plant

Very light fires have relatively little effect on common juniper (Stark and Steele 1977; Kessell and Potter 1980). The amount of damage this species incurs increases with increasing fire intensity (Stark and Steele 1977). Under certain circumstances fires of even low-intensity can kill or seriously damage common juniper. Little is known about the ways in which site characteristics, weather conditions, or season of burn might influence the effect of fire on this species.

Discussion of fire effect

Plant response to fire

Common juniper does not sprout after disturbance. Specific evidence is somewhat limited, but it is probable that seed protected by overlying layers of soil can survive at least some fires. Common juniper typically produces an abundance of long-viability seed, and after low-intensity fires, some seed may germinate. Post-fire recovery is believed to be quite slow, although specific documentation is lacking. Recovery is much slower following high-intensity fires. Relatively slow growth and poor germination characteristics make this species slow to assume prominence after most fires. Possible differences in post-fire response according to variety, season of burn, weather conditions, soils, and vegetative associates, have apparently not been well studied.

Discussion of plant response

Kessell and Potter (1980) reported that mild fires in Douglas-fir/common snowberry habitat types of Montana can actually produce an increase in common juniper. Other researchers have found that following light fires in certain forested areas of western Montana, sites often exhibit fairly minimal shrub damage and have at least some surviving common juniper (Stark and Steele 1977). Results of these studies suggest higher survival rates on lightly burned sites (Stark and Steele 1977).

Upland Shrubland

Holodiscus dumosus--bush oceanspray

Fire ecology or adaptations
Plant adaptations to fire
The fire ecology and fire adaptations of bush oceanspray are poorly documented. Fire is reported to be the principal factor initiating succession on many of the high-elevation sites occupied by this shrub in the Southwest (Hanks 1966), and bush oceanspray is apparently able to maintain itself into late successional stage on many of these rocky, forested sites (Alexander et al. 1984)

Fire regime
It is currently unknown whether post-fire reestablishment is from sprouting from surviving plants, or from on-site or off-site seed sources, or both. In the White Mountains of New Mexico, bush oceanspray was a conspicuous component of the shrub layer 2 to 3 years after fire (Hanks and Dick-Peddie 1974).

Fire effects
Immediate fire effect on plant
The intricately branched nature of this shrub suggests that most plants are readily top-killed by relatively low-intensity fires.

Plant response to fire
Fire response of bush oceanspray is poorly documented. A similar species, oceanspray (*H. discolor*), increases by prolific sprouting.

Juniperus monosperma--one-seed juniper

Fire ecology or adaptations
Plant adaptations to fire
One-seed juniper is considered susceptible to fire (Wright 1972). Fire mortality is generally high for both young and old trees (Johnson et al. 1962), although the thicker bark of older trees may afford some protection (Crane 1982). Older trees often have very little understory vegetation beneath the crown and fuels are quite limited (Crane 1982), consequently, they frequently limit the rate of spread (Vallentine 1971). A history of heavy grazing can also reduce the flammability of mature one-seed juniper stands by reducing fine fuels (Aro 1971). In some locations, mature stands may not burn due to a lack of fuels (Johnsen 1987).

In some areas, juniper is protected from fire by site factors. For example, one-seed juniper frequently grows on rocky breaks or escarpments where fire frequency is very low (Wright 1972). Wright (1972) reported that fire-intolerant plants such as one-seed juniper often occupy isolated topographic breaks which may be surrounded by grasslands that burn at fairly frequent

intervals (Wright 1972). One-seed juniper is described as a climax species on such rocky terrain not subject to frequent fires (Wright 1972).

Post-fire reestablishment is primarily through seed and is relatively slow (Humphrey 1953). Most establishment is from seed dispersed from off-site by birds and mammals, but some establishment may occur from seeds buried on-site and protected from the heat of fire by overlying soil layers. Establishment may be relatively poor even when good germination occurs, and growth is typically very slow (Schott and Pieper 1986). One-seed juniper is usually described as a nonsprouter (Wright 1972), but older trees have been known to sprout infrequently after fire (Crane 1982; Vallentine 1971).

Fire effects

Immediate fire effect on plant

One-seed juniper is generally damaged or killed when above ground foliage is consumed (Johnson et al. 1962). Trees less than 4 feet (1.2 m) in height are particularly susceptible to fire mortality (Dwyer and Pieper 1967). The length of exposure to fire required to kill trees generally increases with size (Jameson 1966).

Average mortality following a low-intensity June wildfire in oak-juniper woodlands of Arizona was 76 percent (Johnson et al. 1962). Researchers observed 92 percent mortality after a stand of small, bushy one-seed juniper was burned (Blackburn and Bruner 1975). Springfield (1976) noted that typically 70 to 100 percent of one-seed junipers less than 4 feet (1.2 m) in height are killed by fire. Mortality rates are apparently somewhat more variable among larger trees. Kills ranging from 30 to 100 percent have been reported in trees 5 to 6 feet (1.5-1.8 m) in height (Springfield 1976).

Relatively little is known about the ways in which fire intensity, specific weather conditions, understory vegetation, and season of burn influence the effect of fire on one-seed juniper. In general, fires of greater intensity appear to be most damaging to one-seed juniper. Researchers have found that 100 percent mortality results when 60 percent of the crown is scorched (Jameson 1966). Temperatures lethal to one-seed juniper tissue are also related to the degree of desiccation (Jameson 1961) and thus season of burn may also influence the amount of damage the plant sustains. Laboratory experiments have shown that lethal temperatures tend to be lowest during late spring and summer (Jameson 1961).

Discussion of fire effect

Fires which occur during late spring or summer may be most damaging to one-seed juniper. Average lethal temperatures range from 153.7 degrees F (67.6 degrees C) to 164.5 degrees F (73.6 degrees C) (Jameson 1961).

Plant response to fire

Regeneration of one-seed juniper is generally through seed when above ground vegetation has been killed or seriously damaged by fire. The length of time required for post-fire recovery of one-seed juniper has not been well documented. However, researchers have observed that following other types of disturbance such as mechanical removal, one-seed juniper only rarely becomes established within the first few years after the disturbance (Schott and Pieper 1986). A

similar response is probable after fire. One-seed juniper did not become prominent after an intense wildfire in Utah until post-fire year 40. This species did not regain dominance on this site until approximately 70 years after the fire (Vallentine 1971). Evidence suggests that factors such as soil type and pre-burn community composition may significantly influence the length of time required for recovery (Schott and Pieper 1987). Seedling establishment may be favored in the shade of dead vegetation, including one-seed junipers (Crane 1982). Once established, one-seed juniper can bear seed as early as 10 years of age on some sites (Schott and Pieper 1987).

Discussion of plant response

In some instances, one-seed juniper can survive when partially defoliated by fire (Dwyer and Pieper 1976). Often the extent of damage is not evident until post-fire year 1 or 2 (Dwyer and Pieper 1967).

Fire management considerations

One-seed juniper has been invading grasslands of the Southwest during much of the past century (Jameson 1962). In some locations grasslands or open woods have been replaced by thick stands of piñon and juniper. Historically, fire is believed to have played an important role in maintaining juniper savannas, and grasslands (Johnsen 1962; White 1965). Fire suppression is believed to have contributed to the relatively recent spread of one-seed juniper. Mueggler (1976) reported that a fire-free period of 85 to 90 years may be required for the formation of a "well-developed" juniper woodland.

Attempts have been made to enhance forage production through the use of fire. In some instances, forage production has increased after one-seed juniper and other species were removed by fire. Aro (1971) reported increased forage production of up to 500 pounds per acre following fire in some piñon-juniper sites in Arizona. However, in other instances little long-term increase in forage production has been noted. The presence of a good stand of residual grasses prior to burning may be important for a favorable reponse.

Removing one-seed juniper by burning individual plants appears to be most applicable to smaller trees less than 10 feet (3.1 m) in height, in stands of up to 100 per (Jameson 1966; Springfield 1976). Small junipers reportedly burn rapidly (Jameson 1966; Springfield 1976). Larger trees frequently take a considerable length of time to burn, making this method prohibitively expensive (Jameson 1966).

Light surface fires have been used in attempts to eliminate young one-seed juniper which have invaded nearby grasslands. Adequate fuel is generally available in these areas and young seedlings and saplings 3.2 to 6 feet (1-1.8 m) or less in height are readily killed (Blackburn and Bruner 1975; Vallentine 1971). Fire-induced mortality in larger trees may be somewhat variable, however (Jameson 1962). In some areas grazing should be deferred for a time to allow for sufficient buildup of fuels (Springfield 1976). One-seed juniper does not survive in grasslands where fires occur frequently (Wright 1972).

Fire may be used in piñon-juniper woodlands in which one-seed juniper is well represented when weather conditions are favorable and sufficient fuels are present. Dense stands of approximately 400 or more trees per acre, with a flammable understory, are most suitable for broadcast burning

(Vallentine 1971). In areas with a past history of heavy grazing where little fuels remain and few residual grasses occur, burning may be difficult and ineffective. Burns tend to be most successful when carried out in June when temperatures are high and the humidity low (Aro 1971; Ralphs and Busby 1979). Unfortunately, the risk of escape may be great during this time period (Aro 1971), and the window for burning very narrow (Ralphs and Busby 1979).

Pinus edulis--piñon pine

Fire ecology or adaptations

Plant adaptations to fire

Piñon pine is generally very susceptible to fire damage. The effect of fire, however, depends largely upon stand structure and understory composition (Wright et al. 1979). Following fire, piñon pines are absent from early successional stages. Seedlings establish primarily via the post-burn food caches of birds and rodents; successful establishment requires a nurse plant. The rate of piñon pine reinvasion of burned areas is determined by relay floristics (Everett and Ward 1984).

Fire effects

Immediate fire effect on plant

Piñon pines are generally very susceptible to fire mortality (DeVelice and Ludwig 1983; Minnich 1984). Tree mortality, however, is largely determined by the extent to which trees dominate a site. Where piñon pine trees have recently invaded grassland communities, young trees less than 4 feet (1.2 m) tall are easily killed. As tree dominance increases and the understory is gradually suppressed, the ability of the understory to carry fires intense enough to kill larger trees also decreases (Wright et al. 1979).

Plant response to fire

Piñon pines reestablish from seed cached by rodents and birds after fire. Reentry into the plant community depends upon perennial nurse plants associated with mid-successional and late-successional stages. Although trees become established 20 to 30 years after burning, coverages and densites are relatively low until approximately 60 years after fire, at which time tree dominance begins to exceed that of the understory. Piñon pines dominate a site within 100 to 150 years of burning (Koniak 1985; West et al. 1975).

Discussion of plant response

Since piñon pines usually require protected sites near the base of 1- to 2-foot tall (0.3-0.6 m) shrubs for successful establishment, piñon pines can be expected to establish sooner on sites where mid to late seral stages existed prior to burning. Piñon pines begin to dominate sites very early in the sere, usually when trees are only 3 to 6.5 feet (1-2 m) tall. As tree roots grow into moisture supplies, understory species decline at an increasingly rapid rate. Areas converted to grasslands should be burned every 20 to 40 years depending on the rate of piñon pine re-invasion (Wright et al. 1979).

Fire management considerations

Prescribed fire is no longer an effective managment tool on many piñon pine-juniper sites where prolonged tree dominance has not only reduced the ability of the understory to carry fires but has

also seriously lowered the ability of the understory to respond to fire. When fires do occur in closed stands, intensities are often so severe and the soil seed reserves so depleted, that without successful seeding, highly flammable annual grass communities are often established (Everett and Clary 1985; West 1984). On many overgrazed woodlands, seeding burns is the only way to restore a desirable perennial grass stage.

Prescribed fire can be used on some sites, depending on stand structure and understory composition. In open stands with 700 to 1,000 pounds per acre (786-1,123 kg/ha) of fine fuels, low-intensity spring fires are very effective in eliminating piñon pine less than 4 feet (1.2 m) tall. Mortality of taller trees in open stands is relatively low unless fuels have accumulated around tree bases (DeVelice and Ludwig 1983; Wasser 1982). Optimal burning conditions in open stands are: air temperatures of 70 to 74 degrees F (21-23 degrees C); relative humidiy of 20 to 40 percent; and wind speed of 10 to 20 mph (16-32 km/h).

Closed stands with little understory are extremely difficult to burn and carry fires only on hot summer days when burning conditions are often hazardous (e.g., air temperatures of 95 to 100 degrees F (35-38 deg C); low relative humidity; and wind speed of 8 to 20 miles per hour (13-32 kmh)). Burning under these conditions usually results in extremely intense crown fires. Fires carry better as tree density increases and as the proportion of piñon to juniper within a stand increases. This increased ease of burning has been attributed to both the greater flammability of piñon pine and the increased site potential associated with piñon pine-dominated stands (Blackburn and Bruner 1975).

In the Great Basin woodlands, the best candidates for effective prescribed burning are ecotonal areas where trees have invaded sagebrush-grassland communities; sites usually have a shrub and tree cover ranging from 45 to 60 percent (Bruner and Klebenow 1979). These sites can be burned effectively with low-intensity spring burns which eliminate the tree overstory; in most cases, reseeding is not required for adequate post-fire understory response (Everett and Clary 1985).

Quercus gambelii--Gambel's oak

Fire ecology or adaptations

Plant adaptations to fire

Gambel's oak are clonal and long-lived. They create heavy leaf litter which creates microenvironments unfavorable for most native herbaceous species. A single oak plant will occupy serveral hundred square feet and create a homogeneous vegetation cover of oak and a few understory herbaceous species (Esco Associates 1992). Gambel's oak generally sprouts vigorously from stem bases or from underground lingotubers and rhizomes following fire (Brown 1958; Clary and Tiedemann 1986; Neuenschwander n.d.; Reynolds et al. 1970; Vallentine and Schwendiman 1973). Both rhizomes and lignotubers are protected from the effects of fire by overlying soil; rhizomes are generally buried at depths of 4 to 20 inches (10-50 cm) and are well adapted to survive most fires (Tiedemann et al. 1987). Fire promotes root sprouting and the formation of buds on rhizomes (McKell 1950; Reynolds et al. 1970). In some locations, particularly in the southern part of its range, Gambel's oak can readily reoccupy a site

through seed protected in buried caches of rodents (Neuenschwander n.d.). Post-fire regeneration through seed does not appear to be significant in northern portions of its range.

Immediate fire effect on plant
Gambel's oak is extremely fire tolerant and is seldom actually killed by fire. It is likely that only extremely severe fires with maximum fuel consumption would produce enough heat to kill the buried rhizomes (Harrington 1985).

Discussion of fire effect
During initial spring, summer and fall burns in Colorado ponderosa pine stands, rhizomes of Gambel's oak were unharmed by fires that killed above ground vegetation except in areas of high fire severity such as near stumps (Harrington 1985).

Plant response to fire
Gambel's oak typically sprouts from underground rhizomes and lignotubers, or from stem bases after fire (Wright and Bailey 1982; Young 1983). Rhizomes generally lie between 4 and 20 inches (10-50 cm) in depth and are protected from all but the most severe fires (Tiedemann 1985). Fire appears to stimulate the growth of many dormant buds on rhizomes (McKell 1950). Gambel's oak often sprouts profusely from the bases of burned stems (Tiedemann et al. 1987). Very large tree-like Gambel's oak found most commonly on optimal sites toward the southern edge of the species' range may be less likely to sprout after fire than the small shrublike growth form which is most common elsewhere. Gambel's oak greater than 12 inches (31 cm) d.b.h. tend to be poor sprout producers (Reynolds et al. 1970).

In some areas, particularly in the southern part of the species' range, seeds may play a role in reestablishment after fire. Reproduction from seed is apparently closely related to animal activity on the site prior to the burn. Seeds buried in rodent caches are protected from most fires by overlying soil and are major seed sources in many areas (Neuenschwander n.d.).

Sprouting may be extremely rapid after disturbance. Wagstaff (1984) reports that "significant growth" can occur within several weeks. In central Utah, sprouts have appeared as early as 10 days after fire (Tiedemann et al. 1987). Sprouts up to 4.72 inches (12 cm) in length were found less than 3 weeks after a July fire (Wagstaff 1984). Only 2 months after a July wildfire and ensuing summer rains, Gambel's oak sprouted profusely from lignotubers with some sprouts reaching 2 to 3 feet (0.6-0.9 m) in height. This fire had consumed all above ground vegetation, all herbaceous plants, and litter. However, Gambel's oak quickly reestablished this site in densities greater than those present prior to the burn (Tiedemann 1985).

Recovery time varies according to fire severity, climatic factors, and site characteristics. In general, Gambel's oak recovers more rapidly at lower elevations on warmer, south-facing slopes (Harper et al. 1985). Recovery of Gambel's oak may take anywhere from 6 to 40 years (Kunzler and Harper 1980; Wright et al. 1979). In Northern Utah, McKell (1950) reported 50 percent recovery only 2 years after fire and 75 percent recovery 18 years after the burn. Recovery time in drier forest and sagebrush-grassland zones of the Intermountain region averages 30 to 40 years (Wright et al. 1979). Gambel's oak recovers from fire in 6 to 35 years in central Utah (Kunzler

and Harper 1980) and has completely recovered to pre-burn levels within 10 years on some northern Utah sites (Christensen 1949).

Gambel's oak can occasionally flower a second time following summer fires. At one northern Utah site an August 1 fire defoliated but did not kill Gambel's oak stems. New shoots with flowers were produced by August 27th (Christensen 1955). Vigorous shoot growth generally leads to at least a short-term increase in productivity after fire in Gambel's oak communities. The productivity of Gambel's oak on Colorado rangeland increased 7l percent 2 years after a burn, 33 percent 5 years after fire, and was greater than 20 percent 10 years later (Kufeld 1983).

Gambel's oak often grows in very dense thickets on steep slopes in Colorado and elsewhere. Fire may have promoted the growth of these thickets since fire tends to burn rapidly and covers larger areas on steep slopes (Brown 1958). Gambel's oak sprouts prolifically after fire and increases in density after large burns of this kind. Increases may be dramatic. In a Colorado study, Gambel's oak increased 100 to 150 percent in density and 10 to 40 percent in frequency following a single burn (Harrington 1985).

Single burns in any season generally result in prolific sprouting. Increases in oak density and frequency typically occur, with only temporary decreases in cover (Harrington 1985). Consecutive, annual burns may inhibit the growth of Gambel's oak in some instances (Christensen 1949; Harrington 1985). Current evidence suggests that repeated, relatively high-severity fires may be necessary to reduce Gambel's oak (Harrington 1985; Steinhoff 1978). Gambel's oak still exhibited short-term increses in density and frequency after two consecutive spring and fall burns in Colorado (Harrington 1985). Some researchers contend that at least three treatments (burning, mechanical removal, or chemical control) in successive years may be required to kill Gambel's oak (Steinhoff 1978).

Gambel's oak is most severely harmed by successive fires during periods when carbohydrate reserves are low. Gambel's oak was basically unharmed by two consecutive spring burns in Colorado. Sprouts produced after these burns had 6 to 8 weeks to recover before the onset of fall dormancy, and plants were able to accumulate carbohydrate reserves before winter. Summer burns during the period of rapid regrowth when carbohydrate reserves are low appear to be most damaging to Gambel's oak. Sprouts cannot mature by fall dormancy and the plant must survive 9 to 10 months without carbohydrate replacement (Harrington 1985).

Repeated summer burning may significantly inhibit growth or eventually even kill Gambel's oak. However, in many areas, consecutive summer burns may be difficult to accomplish. Litter fuel accumulations are often very light during a single growing season, and prescribed fires must be set during extremely hazardous burning conditions in order for fire to carry. Harrington (1985) reported that biennial burns may be the best option for fire application in many areas. Research on Gambel's oak response to more than two consecutive annual burns is lacking.

Discussion of plant response

Recovery time of Gambel's oak varies with fire severity, climate, and site characteristics (Kunzler and Harper 1980). Initial shoot growth of Gambel's oak after fire is generally rapid. Shoot growth at a northern Utah site averaged 18 inches (46 cm) after the first growing season.

An average increase of 20.5 shoots per yard square declined to a difference of only 0.5 plants on the burned area 18 years after the burn (McKell 1950). Gambel's oak reached heights of 8 to 10 feet (2-3 m) 13 to 15 years after fire in Arizona piñon-juniper woodlands. Occurrence of Gambel's oak was 6 percent on burned plots and 19 percent on unburned plots (McCulloch 1969).

Productivity of Gambel's oak generally increases after fire. A Colorado study compared relative productivity (kg/ha) of Gambel's oak on units which were burned, sprayed, or chained. Results of this study are as follows (Kufeld 1983):

	time in relation to burn			
	1 yr before kg/ha	2 yr after kg/ha	5 yr after kg/ha	10 yr after kg/ha
untreated	345.86	338.99	206.80	287.99
burned	196.13	213.13	162.40	204.46
sprayed	231.53	226.60	123.20	163.40
chained	371.93	484.46	272.73	539.79

After 2 yr		After 5 yr		After 10 yr	
kg/ha	%	kg/ha	%	kg/ha	%
-6.87	-2	-139.06	-40	-57.87	-17
+20.28	+7	+45.23	+33	+39.44	+20
-1.09	-3	-15.12	-16	-31.41	-14
+118.74	+29	+50.54	+21	+226.86	+67

Fire management considerations

Fire can significantly influence both the growth form and stand characteristics of Gambel's oak. Fire generally promotes the shrublike growth of this species and tends to thicken existing stands through proliferation of new sprouts. Reestablishment of ponderosa pine may be difficult on these sites and persistent scrub oak communities may develop (DeVelice and Ludwig 1983; Winward 1985). Fire also creates conditions favorable for the invasion of Gambel's oak in many Southwestern spruce-fir communities (Christensen 1949). Gambel's oak can serve as a ladder fuel and can contribute to crowning in ponderosa pine forests of the Southwest. A dense oak understory greatly increases the probability of fire in these communities. Gambel's oak is particularly flammable when late spring frosts kill sprouting shoots, which die but remain on the plant until late summer or fall (Winward 1985).

Symphoricarpos albus-common snowberry

Fire ecology or adaptations

Plant adaptations to fire

Common snowberry is a survivor of low- to moderate-intensity fires. It sprouts from rhizomes located in mineral soil (Fischer and Clayton 1983).

Fire effects

Immediate fire effect on plant

Common snowberry is moderately resistant to fire kill. Rhizomes survive low- to moderate-intensity fires but may not survive severe fires (Crane and Fischer 1986; Fischer 1981).

Plant response to fire
Common snowberry sprouts vigorously from its rhizomes (Fischer and Clayton 1983) and sprouts less vigorously from its root crown (Crane and Fischer 1986). It may produce some fruit in the first post-fire year (Crane et al. 1983).

Discussion of plant response
Common snowberry increased in volume by 12 times within 5 years of a wildfire in western Montana (Crane et al. 1983). In a ponderosa pine/common snowberry community of northern Idaho, common snowberry increased in biomass from 8.1 g/m2 to 14.6 g/m2 in a 4-year period following a prescribed burn (Merrill 1982). Following a wildfire in Oregon, common snowberry was much more abundant in pine/mixed fir communities not seeded with grasses than in seeded pine/sedge, pine/mixed fir, and mixed fir communities (Anderson and Brooks 1975). Light-intensity spring fires benefited common snowberry in Idaho seral brush communities (Hooker and Tisdale 1974). Clearcutting followed by broadcast burning promoted the greatest overall shrub biomass compared to other postharvest treatments in a western Montana study (Richardson 1980). Burning in summer on dry slopes may not benefit common snowberry as much as burning in more mesic communities (Merril 1982).

Fire management considerations
Fire potentials based on stand site data and fuel loading are listed for several habitat types in Montana in which common snowberry is an indicator (Fischer 1981; Fischer and Bradley 1987). Brown (1976) listed above ground weights and leaf weights to basal stem diameter ratios for assessing flammability. Bradley (1984) predicted plant survival based on rhizome measurements and depth. Common snowberry should be marked as a potential spot-fire hazard when it occurs near fire lines (Neuenschwander n.d.).

Upland Grassland

Andropogon gerardii var. *gerardii*--big bluestem

Fire ecology or adaptations
Historically, fires occurred frequently in the tallgrass prairie and were essential in maintaining these grasslands (Daubenmire 1968). Across the Great Plains, lightning-caused fires may have occurred as frequently as every 1 to 6 years (Kucera 1981). Having evolved in a grassland environment subjected to frequent fires, big bluestem is well adapted to fire. After above ground foliage is consumed by fire, new growth is initiated from rhizomes. The well-developed rhizomes are generally 1 to 2 inches (2.5-5 cm) below the soil surface (Albertson 1937; Weaver 1958). In general, grassland fires are of low-intensity because the flames pass quickly, and the soil temperature 1 inch (2.54 cm) below the surface rises very little (Kucera 1981). Plants burned during the spring when dormant quickly send up vigorous new growth because of stored carbohydrate reserves in below ground roots. If burned during the summer when plants are actively growing, plants normally survive by initiating new growth from rhizomes. However, regrowth may be slower and less vigorous than in plants burned when dormant (Ewing and Engle 1988).

Fire effects
Immediate fire effect on plant
Under dry conditions, up to 100 percent of live and dead above ground stems and leaves may be consumed by fire (Ewing and Engle 1988; Hulbert 1969). Rhizomes, however, survive, even when above ground tissue is completely destroyed.

Discussion of fire effect
Fires that occur during the active growth stage in the summer are most damaging to big bluestem because of the limited carbohydrate reserves available for new growth during this time period (Daubenmire 1968). Burning tallgrass prairie in Oklahoma during late summer resulted in nearly complete combustion of above ground biomass (Ewing and Engle 1988). On grazed plots with little fuel accumulation, tillers were completely defoliated but did not suffer apical meristem damage. On ungrazed plots with heavy fuel accumulation, tillers did suffer apical meristem damage. Tiller density was significantly lower 2 months after burning on these high fuel plots, and new growth consisted largely of newly initiated tillers.

Plant response to fire
Big bluestem initiates new growth from surviving rhizomes after above ground foliage has been damaged or consumed by fire. After summer burns, depending on the amount of damage inflicted, regrowth is either from undamaged apical meristems or from the initiation of new tillers from underground rhizomes (Ewing and Engle 1988). Plants burned during the spring or fall when the above ground foliage is dead resume growth in the spring as normal. In general, spring burning has a stimulating effect on the growth and competitive vigor of big bluestem (Kucera 1981).

Big bluestem plants in recently burned areas start growth earlier in the spring, develop faster, and produce more herbage than plants in unburned areas. This earlier and increased growth is most often attributed to increased solar radiation reaching the soil following the removal of standing dead material (Hulbert 1969; Hulbert 1988; Knapp 1985; Knapp and Gilliam 1985; Old 1969; Peterson 1983) which results in higher soil temperatures (Kucera and Ehrenreich 1962; Pemble et al. 1981). Following spring burning in native bluestem prairie in Missouri, soil surface temperatures in burned areas compared to unburned area averaged 7.1 degrees F (3.9 degrees C) warmer in April, 11.4 degrees F (6.3 degrees C) warmer in May, 8.3 degrees F (4.6 degrees C) warmer in June, and 7.1 degrees F (3.9 degrees C) warmer in July (Kucera and Ehrenreich 1962). Increased soil temperatures promoted earlier root growth and activity and thus earlier emergence of shoots (Knapp and Seastedt 1986; Peet et al. 1975; Reis et al. 1988). Also, the emerging shoots received more sunlight because they are not shaded by the standing dead shoots and leaves of the previous year. Knapp (1985) reported that following spring burning in Kansas tall grass prairie, big bluestem had a greater photosynthetic rate and increased stomatal conductance and leaf thickness in response to increased solar radiation.

In the tallgrass prairie, late spring burning, when warm-season grasses are about to resume growth, is the most beneficial to big bluestem. Big bluestem stands burned at this time show the highest above ground biomass gains compared to unburned stands or stands burned at other times of the year (Aldous 1934; Anderson and Birkenholz 1983; Towne and Owensby 1984). In the true prairie region, spring burning of big bluestem stands that have not been burned within the past few years almost always results in increased yields. Two to three fold increases are common (Hadley and Keickhefer 1963; Hulbert 1969; Kucera and Ehrenreich 1962; Old 1969; Peet et al. 1975; U.S. Department of Interior, Bureau of Land Management n.d.). Increases in big bluestem growth following late spring burning are associated with increases in (1) the density of tillers (Hulbert 1969; Knapp 1984; Svejcar and Browning 1988), root and rhizome biomass (Hadley and Keickhefer 1963), and (3) size and number of leaves (Aldous 1934; Knapp 1985; Knapp and Gilliam 1985; Svejcar and Browning 1988).

Increased flower stalk production is also common following burning (Curtis and Partch 1950; Ehrenreich 1959; Ehrenreich and Aikman 1957; Hadley and Keickhefer 1963; Hulbert 1978; Old 1969). Following burning in Wisconsin, flower stalk production increased six-fold (Curtis and Partch 1950) after a spring burn in Iowa, flower stalk production increased seven fold but returned to normal by the third post-burn growing season (Ehrenreich 1959). Although numerous researchers report dramatic increases in seed production, Abrams (1988) observed that seedlings were less frequent in areas burned annually or on a 4-year cycle than in unburned areas.

Big bluestem rapidly produces large amounts of foliage following fire. Generally, it takes 2 to 5 years after burning for litter accumulation to equal that of unburned areas (Daubenmire 1968). When the fire interval is greater than about 5 years, bluestem prairie becomes unproductive because the large accumulations of standing dead material stifles growth. However, fires occurring more frequently than every 2 years will probably lower biomass production.

Discussion of plant response

Post-burn herbage production indicates that the season of burning, the amount of annual precipitation the area receives, and site conditions greatly influence big bluestem's response to burning. Big bluestem increases as influenced by different annual burning dates in Kansas are summarized below (Towne and Owensby 1984):

Kansas prairie burned annually 48 of 56 years	
time of burning	% big bluestem in the grass community
prestudy (unburned)	16.7
poststudy (unburned)	18.0
late spring, about May 1	46.0
mid-spring, about April 10	24.0
early spring, about March 20	23.0
winter, about December 1	35.0

Farther west, in the mixed-grass prairie which receives less precipitation, big bluestem generally increases following fire if precipitation is average or above average but decreases if drought conditions occur. Response is somewhat variable though, and dependent upon site characteristics. On upland sites in the mixed-grass prairie where big bluestem occurs infrequently, it is generally not stimulated by burning (Steuter 1987). Big bluestem is more common on lowland sites, and because lowlands receive some surface water runoff, burning these sites in the spring results in increased big bluestem biomass (Engle and Bultsma 1984; Steuter 1987). In the mixed-grass prairie of north-central South Dakota, big bluestem production increased following spring burning on overflow sites, even though 2 years of below average precipitation followed (Engle and Bultsma 1984). Results from this study are summarized below:

burning date	phenological state	leaf length (cm)		flower stalks/m^2		current years growth (g/m^2)
		8/80	8/81	8/80	8/81	8/80
May 15, 1980	dormant	21.3	45.2	132.4	69.8	128.8
June 16, 1980	5-10 cm tall	16.8	37.0	95.2	53.8	71.4
control		42.4	43.4	2.0	15.123 .0	

Fires that occur during the summer cause the most harm to big bluestem. Summer burning during a below average precipitation year in north-central South Dakota resulted in the reduction of big bluestem on overflow sites, and shifted dominance towards cool-season grasses (Steuter 1987). In the true prairie of Oklahoma, late summer burning resulted in initial reductions of big bluestem tiller density; however, by the end of the growing season, tiller density returned to normal (Ewing and Engle 1988). Measurements taken the following August showed that productivity was within the seasonal normal range. In the Black Hills of South Dakota, big

bluestem basal cover remained unchanged following an October burn (Bock and Bock 1983), and increased slightly following spring burning (Gartner 1977).

Fire management considerations
Late spring is the best time to burn big bluestem stands. The later in the spring burning occurs, just prior to the emergence of new growth, the greater the post-burn herbage production. If burns are conducted too early, production may decrease as a result of the increased evaporation of soil moisture in the interval between the fire and the resumption of new growth (Kucera 1981). Spring burns leave the soil exposed for the least amount of time and thus reduce soil moisture levels over the growing season less than winter, early, or mid-spring burns (Anderson 1965; McMurphy and Anderson 1965). Regardless of season of burn, however, soil moisture content is consistently lower in burned areas than in unburned areas. Thus in drought years, in areas of low precipitation, or in areas where soil moisture is limiting, big bluestem will probably not show the high post-burn biomass increases generally reported for tallgrass prairie sites.

Late spring burning can be used to increase grass productivity and improve cattle use. Cattle prefer vegetation on burned sites over that on unburned sites (Launchbaugh and Owensby 1978). Compared with weight gains of cattle grazing in nearby unburned pastures, weight gains of cattle grazing on late spring burned pastures were 17 percent higher in Oklahoma (Steuter 1987) and 11 percent higher in Kansas (Launchbaugh and Owensby 1978).

Bouteloua gracilis--blue grama

Fire ecology or adaptations
Plant adaptations to fire
Blue grama is generally dormant during early spring and in the hottest, driest part of summer. It is also dormant in cold winter months. When warm-season grasses such as blue grama are burned while dormant, living plant parts are often unaffected (White and Currie 1983). Reestablishment occurs through rhizomes, which may be unaffected or even stimulated by fire, and by germination of wind-dispersed, water-dispersed, or animal-dispersed seed (Weaver and Albertson 1944).

Immediate fire effect on plant
Fire generally top-kills blue grama. Rhizomes are usually unharmed. This species is generally less damaged when burned during the spring than when burned during the summer or fall (Daubenmire 1968; White and Currie 1983). During the spring when soils are moist, heat penetration into the soil is slight, and damage to underground parts is minimal (White and Currie 1983). Blue grama is usually unharmed by fires in years with above normal winter and spring precipitation (Wright and Bailey 1980). It can be severely damaged by fires that occur during drought years (White and Currie 1983; Wright and Bailey 1980).

Discussion of fire effect
An Arizona study indicated that lethal temperatures of culm bases of sensitive perennial grasses such as blue grama closely approximated existing air temperatures at or near ground level during the hot, dry months of summer. During cooler, moister periods, the lethal temperatures of culm

bases were much higher. Over 2 years, the lethal temperature for blue grama growing in piñon-juniper woodland ranged from 108 to 128 degrees F (60.0-71.2 deg C) (Jameson 1961).

Plant response to fire
Blue grama's response to fire varies. It may be harmed, largely unaffected, or increased by fire. Its response depends on season of burning, soil moisture, temperature, plant community composition, and five severity. Recovery is more rapid during years of above average precipitation. Recovery time ranges from 1 to 4 years or more depending on conditions (White and Currie 1983; Wright and Bailey 1980).

Discussion of plant response
A Texas study compared post-fire blue grama yields after fires carried out in dry and wet years (winter-spring precipitation 0 to 40 percent above normal). Blue grama was tolerant of fire during wet years; there was no loss in herbage yield by the end of the first growing season (Wright and Bailey 1980).

When a Central Great Plains blue grama-buffalo grass prairie was burned during the spring of a dry year, yields recovered to 35, 62, and 97 percent at the end of the first, second, and third post-fire growing seasons (Wright and Bailey 1980). In a New Mexico study, blue grama herbage was reduced by 30 percent the first year after an early spring fire, but with adequate moisture recovered completely by the second year (Dwyer and Pieper 1967). At the end of the first growing season after an October fire in Nebraska, productivity of blue grama on burned plots was less than on adjacent control plots (Morrison et al. 1986). However, in Montana, White and Currie (1983; 1981) found that spring burning reduced blue grama yield early in the growing season, but increased productivity later in the same season.

Fire management considerations
Application of fertilizers such as ammonium sulfate significantly influences yields of blue grama on burned and unburned plots. In New Mexico, plots burned but not fertilized produced 9.0, 11.3, 15.6, and 56.3 percent more blue grama during October, January, June, and April, respectively, than did control plots (Dwyer 1971). Plots burned and fertilized produced 103, 116.9, 164.4, and 173.6 percent more forage in October, January, June, and April than did the control. Plots fertilized in June and left unburned produced 200.1 percent more blue grama than the control (Dwyer 1971).

The nutrient content of blue grama is altered by fire. June crude protein levels following a low-intensity April fire in New Mexico were 16.1 percent compared with 14.8 percent on unburned control plots (Dwyer and Pieper 1967). In a Nebraska study, several species of grasshoppers showed a marked preference for blue grama from burned areas, suggesting potentially higher nutritional value of grama on burned sites (Morrison et al. 1986). A 3- to 4-month rest from grazing is recommended after fire (Wright 1974). Burning blue grama during drought years should be avoided if productivity is to be maintained (White and Currie 1981).

Bouteloua hirsuta-hairy grama grass

Fire ecology or adaptations

Hairy grama increases, decreases, or remains unaffected by fire, depending on climatic conditions, season of burn, severity of fire, and growth form. Hairy grama has short rhizomes in the northern part of its range, but grows in clumps resembling a bunchgrass in the south (U.S. Department of Agriculture, Forest Service 1937). Rhizomatous grasses are generally unharmed or increased by fire. In many areas hairy grama is harmed by fire during dry years but largely unaffected by fire in wet years (Wright and Bailey 1980). Recovery time is also variable but has been reported as 3 years in New Mexico and Texas desert shrub communities (Ahlstrand 1982).

Fire effects

Immediate fire effect on plant

The effects of fire on hairy grama appear to be highly variable. Research indicates that some fires have harmed or reduced hairy grama whereas others have enhanced its growth or have had no discernible impact on this species. The effect of fire depends primarily on growth habit, fire severity, climatic conditions, and season of burn.

Plant response to fire

Hairy grama has short rhizomes in the northern portion of its range (U.S. Dept. of Agriculture, Forest Service 1937). Rhizomatous grasses are generally favored by fire, as fire is believed to stimulate the initiation of new shoots at primordial regions of the root systems. The variability in growth form may account for some of the differences in plant response described in the literature.

Discussion of plant response

In the shortgrass and mixed prairies of the Central Great Plains, hairy grama is harmed during dry years but is probably tolerant of fire during wet years (Wright and Bailey 1980). Hairy grama increases slightly or remains largely unaffected by fire in the tallgrass prairie of the Central Great Plains (Wright and Bailey 1980). It reportedly increases in response to fire regardless of the date of burning in parts of western South Dakota (Worcester 1979).

Productivity of hairy grama approached control levels 1 year after an October wildfire in the Sandhills of Nebraska (Morrison et al. 1986) is as follows:

Phytomass (g/m^2)				
	June	July	August	October
Burned area	0.3	1.3	4.5	8.8
Control	4.8	8.1	4.2	12.8

Although recovery time varies, Ahlstrand (1982) reported that hairy grama in a Chihuahuan desert shrub communities had recovered to pre-fire levels by the third post-fire season.

Calamagrostis purpurascens-purple pinegrass

Fire ecology or adaptations
Purple pinegrass can establish on burned sites by wind-dispersed seeds (Daubenmire 1953). It can also sprout from on-site surviving rhizomes after fire.

Fire effects
Immediate fire effect on plant
Purple pinegrass mortality following fire has not been widely documented. Fire, however, will presumably kill above ground vegetation of purple pinegrass. Severe fires may kill below ground rhizomes.

Plant response to fire
Purple pinegrass will typically colonize sites after fire through wind-dispersed seeds (Daubenmire 1953). After low-severity fires this grass will presumably sprout from on-site surviving rhizomes. In Alaska, purple pinegrass typically invades dry south-facing slopes after fire. This grass will persist here until the canopy closes; this process, however, appears to be very slow in many places (Daubenmire 1953).

Fire management considerations
In subalpine and alpine habitats where purple pinegrass commonly occurs, vegetation recovers slowly from disturbance because of the cold climate and short growing season (Crane 1982; Daubenmire 1953). The exposed nature of these sites may increase the possibility of lightning strike, but the lack of fuels reduces the likelihood of fire spreading through the stand. The subalpine grasslands that form the early successional stage may last a century or more (Crane 1982).

Calamovilfa longifolia-prairie sandreed

Fire ecology or adaptations
Prairie sandreed is considered to be fire tolerant when dormant. Plants probably reestablish after fire via the initiation of new growth from rhizomes. A limited amount of research has been conducted concerning the influence of fire on prairie sandreed. Research results on post-burn frequencies are contradictory at best. Recovery rates after fire appear to be influenced by season of the burn, associated plant species, and mean annual precipitation. Spring burning in the tallgrass prairie of eastern North Dakota increased prairie sandreed's frequency by more than 100 percent compared with unburned plots (Kirsch and Kruse 1973), while in the mixed prairie of western North Dakota spring burning slightly reduced prairie sandreed's frequency and fall burning significantly reduced its frequency (Lyon and Stickney 1976).

Fire effects
Immediate fire effect on plant
Fire kills above ground parts of prairie sandreed, but the rhizomes probably survive most fires.

Plant response to fire

Prairie sandreed starts growth earlier in the spring following a fire. After an October lightning fire in Nebraska, prairie sandreed's growth rate increased the following spring, but the rate slowed so that by August it was equal to that of the unburned area (Morrison et al. 1986).

The effect of fire on frequency of prairie sandreed appears variable. After spring fires, some researchers have found it to increase in frequency, while others have found it to decrease in frequency. In Alberta, prairie sandreed increased greatly in canopy cover and frequency following 25 years of annual early spring burning (burning conducted soon after snowmelt) (Anderson and Bailey 1980). Here percentage seed head presence was much higher on burned versus unburned areas. Whether reproduction by seeds increased is not known, but fire removed litter buildup and exposed mineral soil, creating a more favorable seedbed. Burning also reduced prairie sandreed's blade length (Anderson and Bailey 1980). Two and one-half months after a spring burn in western North Dakota, prairie sandreed was slightly reduced in frequency compared with the unburned plots (Dix 1960). Four years after an August 14 fire in this same area, frequency of prairie sandreed was still reduced significantly. However, after a spring fire in eastern North Dakota, frequency of prairie sandreed was increased by more than 100 percent (Kirsch and Kruse 1973). After a spring wildfire in Nebraska, prairie sandreed increased by 11 percent on south-facing slopes but decreased by 8 percent on hilltops and north-facing slopes (Bragg 1978), the decrease was attributed to competition from associated species.

Discussion of plant response

Fire management considerations

Summer or fall burning is harmful to prairie sandreed, causing a decrease in frequency and phytomass. The response to spring burning is less clear. Spring burning appears to be beneficial or only slightly detrimental. Prairie sandreed occurs in both tallgrass and mixed-grass prairie regions. In the tallgrass prairie, fire seems to favor the tall warm-season grasses, and prairie sandreed probably benefits from spring burning (Bragg 1978; Kirsch and Kruse 1973). In the mixed-grass prairie, prairie sandreed may or may not benefit from spring burning (Anderson and Bailey 1980; Dix 1960).

Fire may also affect plant-water relationships of prairie sandreed. Since prairie sandreed is usually found on sandy soil, virtually all the water percolates into the soil regardless of litter buildup. The removal of litter due to fire may lead to increased evapotranspiration rates, leading to increased water stress (Morrison et al. 1986).

Carex pensylvanica ssp. *heliophila*-yellow sedge

Fire effects

***Immediate fire effect on* plant**

Fires typically top-kill yellow sedge. In the plains, the heat of combustion is confined entirely to the surface, thus not damaging the rhizomes (Hensel 1923). However, yellow sedge does not do well after hot fires because its roots and rhizomes do not penetrate deep into the soil (Abrams and Dickmann 1982).

Plant response to fire

Yellow sedge exploits fire-generated gaps in the litter layer through aggressive clonal propagation (Matlack and Good 1989). Recovery is usually within 1 or 2 years (Steiger 1930). Seed germination also occurs but is rare.

In lower north Michigan, yellow sedge is less dominant on burned sites than on clearcut sites. Fire does not appear to result in "regressive succession" (Abrams and Dickmann 1982). Some sources report yellow sedge as a fire persister (Potter and Moir 1961). Others report yellow sedge as a fire decreaser or increaser (Johnson 1992).

Discussion of plant response
The information regarding post-fire response of yellow sedge is contradictory. In oak woods, increases in yellow sedge following burning have been reported; these increases may have been related to an increase in soil pH (Eyre 1980; Swan 1970). Yellow sedge is abundant following prescribed fires where trees and shrubs have failed to sprout (Collins and Good 1986). In central Minnesota, nominal decreases in yellow sedge were reported (Johnson 1992). At a site in North Dakota, there were no significant increases or decreases in comparison to unburned stands (Eyre 1980).

Fire management considerations
In restoration of prairie vegetation, burning every 3 to 5 years discourages woody vegetation and encourages grasses and forbs, including yellow sedge. Mid to late summer fires appear to benefit cool season graminoids the most. Spring fires do not benefit yellow sedge and can be more harmful to wildlife and more explosive than summer fires (Blewett 1978).

Danthonia parryi--Parry oatgrass

Fire ecology or adaptations
There was no information found regarding Parry oatgrass specifically. In fact, scant information is available on any species of oatgrass. However, timber oatgrass (*Danthonia intermedia*) is reported to be moderately resistant to fire (Volland and Dell 1981).

Fire effects
Plant response to fire
Timber oatgrass (*Danthonia intermedia*) is rated as moderate in post-fire regeneration response in the Pacific Northwest, with a 35 to 64 percent chance that at least 50 percent of the population will survive or reestablish after fire (Volland and Dell 1981). It reportedly takes 5 to 10 years to approximate pre-burn frequency or coverage (Volland and Dell 1981).

Timber oatgrass was reported to increase after a midsummer fire (Volland and Dell 1981). Two seasons after a spring burn in northwestern Montana, timber oatgrass was reported to have increased by 7.5 percent (Bushy 1985).

Muhlenbergia montana-mountain muhly

Fire ecology or adaptations
Mountain muhly may sprout after aerial portions are burned. It is densely tufted (Hickman 1993; Munz 1973) and old sheath bases are persistent (Welsh et al. 1987) they may protect basal buds from fire damage. On the other hand, in hot dry conditions the dead litter of a mountain muhly plant can produce a hot fire which may damage or kill the plant (Vose and White 1991).

Fire effects
Plant response to fire
Mountain muhly density generally decreases from pre-fire values during the first few years after fire (Gaines et al. 1958; Oswald and Covington 1984), but it may increase over original values (Andariese 1982). Mountain muhly usually takes at least 3 years to fully recover from fire (Gaines et al. 1958). However, after prescribed fire in central Arizona, mountain muhly had recovered pre-fire biomass within 10 months (Harris and Covington 1983).

Discussion of plant response
In central Arizona mountain muhly was sampled in September 1981, on sites that previously had been prescribed burned in ponderosa pine pole timber and mature stands. Thinning treatments and grazing management varied from site to site. All burned sites were matched with similar, unburned controls. The following mountain muhly production (kg/ha) means were reported (Andariese 1982); standard errors are in parentheses:

Ponderosa pine pole stands			
	2 yr burn	5 yr burn	7 yr burn
Burn	0.21 (0.06)*	0.60 (0.22)	4.56 (1.06)*
Control	1.60 (0.82)*	0.28 (0.12)	1.22 (0.38)*
Mature ponderosa pine stands			
Burn	0.88 (0.30)	0.16 (0.15)	0.84 (0.27)
Control	8.24 (4.20)	0.10 (0.05)	0.67 (0.20)

* indicates significant difference ($p<0.05$) between burn and control.

In 2-year-old burns, mountain muhly production was less than on control sites. In 5- and 7-year-old burns, mountain muhly production was greater than on control sites (Andariese 1982).

In central Arizona mountain muhly occurred on an area that was prescribed burned October 18 and 19, 1977. Mountain muhly density was sampled before the fire, in 1974 and again after the fire in 1980. Backfires and short strip headfires were used; estimates of fuel consumption ranged from 50 to 75 percent. Most ponderosa pine regeneration was not killed. Mountain muhly density was 1.01 stems per square meter in 1974. In 1980, after the fire, stem density was zero (Oswald and Covington 1984).

In central Arizona mountain muhly biomass and nutrient concentrations were measured during the first growing season after burning on plots in a ponderosa pine/Arizona fescue habitat type. The stand had been unburned since a fire in 1876. The overstory consisted of uneven-aged

ponderosa pine distributed in even-aged groups of mature trees, poles, or saplings. Mountain muhly was dominant in the herbaceous vegetation within openings. Controlled burning occurred in November 1976. The fire consumed surface needles on 94 percent of the area, and exposed mineral soil on 16 percent of the area. Fuels less than 1 inch (2.54 cm) in diameter were reduced 63 percent. Fuel reduction was greatest under mature trees, where fuel loads were heaviest; it was intermediate in pole stands, and least in sapling stands. Mountain muhly standing crop was sampled on 11 burned and 7 unburned plots during June and September of post-fire year 1. Mountain muhly standing crop and measured nutrient concentrations (% oven-dry weight) were as follows 7 months after fire (Harris and Covington 1983).

June 1977						
	Mature Timber		Pole		Sapling	
	Unburned	Burned	Unburned	Burned	Unburned	Burned
Standing Crop (kg/ha)	3.97	3.26	2.18	0.47	0.26	2.09
N (%)	0.99	1.19	1.02*	1.31	1.07	1.15
P (%)	0.25*	0.35	0.28*	0.34	0.27	0.32
K (%)	0.69*	0.82	0.68*	0.82	0.62*	0.82
Ca (%)	0.13*	0.18	0.14*	0.19	0.14*	0.18
Mg (%)	0.10*	0.14	0.12	0.14	0.12	0.12

* Indicates significant difference ($p<.05$) between burned and unburned sites within a stratum.

By 10 months after fire, no significant differences in standing crop were found between unburned and burned plots, and few significant differences in nutrient concentrations persisted (Harris and Covington 1983).

Fire management considerations

Mountain muhly is a principal grass in some ponderosa pine/grass types where recurring fires have maintained savanna. In north-central Arizona mountain muhly will usually carry surface fires in November, and perhaps during the winter and spring (Biswell 1973).

In the Santa Catalina Mountains of Arizona, decades of fire protection have resulted in many ponderosa pine stands with a dense undergrowth of ponderosa pine saplings. In the 1960's, an inventory of ponderosa pine stands burned within the past few decades revealed that those burned by lightning-caused fires, which occur at a relatively high frequency in the range, were primarily open and park-like, with an herb layer dominated by mountain muhly. However, intensity of a major incendiary fire was so severe that it killed all ponderosa pines in the stand and enhanced development of oak scrub, which shaded out mountain muhly (Niering 1981).

Schizachyrium scoparium--little bluestem

Fire ecology or adaptations

Historically, fire played a major role in the ecology of grasslands in which little bluestem occurs (Daubenmire 1968). Little bluestem is well adapted to fire when dormant, since food reserves are stored in below ground roots and crowns. Dormant plants burned in the spring resprout from buds within the insulated crowns, thereby revegetating the burned community. Burning little bluestem during summer when plants are actively growing significantly reduces basal cover (Ewing and Engle 1988). Plants are particularly sensitive to summer fires because apical meristems are 1 inch (2.54 cm) above the soil surface, and hence easily consumed by fire (Abrams and Hulbert 1987; Anderson and Fly 1955).

Fire effects

Immediate fire effect on plant

Little bluestem is minimally affected by fire if burned dormant. It increases or only slightly decreases in frequency following dormant-season fires (Anderson et al. 1970; Ehrenreich and Aikman 1963; Kucera and Ehrenreich 1962; Wright 1974a). Spring and winter fires consume above ground plant parts in proportion to the amount of moisture in the standing dead material and in the soil. In Nebraska, Bragg (1982a) reported that about 98 percent of above ground biomass was consumed by April fires, while 84 percent of above ground biomass was consumed by fire in June when plants were moist and actively growing. Results from prescribed fire in late spring in South Dakota indicate that as soil and fuel moisture content at the time of ignition increases, the amount of above ground vegetation consumed decreases (Worcester 1979). Fires in areas with a soil moisture content of 33 percent and a fuel moisture content of 30 percent consumed 47.2 percent of little bluestem above ground biomass. However, fire in areas with a soil moisture content of 46 percent and a fuel moisture content of 45.6 percent consumed only 31.4 percent of above ground biomass.

Discussion of fire effect

Most often exhibiting a bunch habit, the plant crown of little bluestem is relatively resistant to fire under moist conditions (Towne and Owensby 1984). Under dry conditions, fire can burn the crowns more easily, injuring basal buds that are below the soil surface during dormancy (Wright and Bailey 1982). During summer little bluestem is particularly susceptible to fire damage because apical meristems are elevated about 1 inch (2.5 cm) above the soil surface (Branson 1953) and therefore exposed to the fire's flames and heat (Wright and Bailey 1982). Late summer fires in Oklahoma resulted in little bluestem suffering 58 percent basal area reduction on plots with low fuel quantities and 95 percent reduction on plots with high fuel quantities (Ewing and Engle 1988). Regrowth within 2 months was minimal resulting from surviving tillers. Few new tillers were initiated.

Plant response to fire

The extent to which seed contributes to revegetating post-burn stands is unknown, but Ehrenreich and Aikman (1963) reported that seeds from burned stands have higher germination percentages than seeds from nearby unburned stands. Little bluestem plants burned when dormant generally start growth earlier in the spring and produce more herbage than plants on nearby unburned areas. Early resumption of spring growth has been observed during the first

growing season following late spring burns in Iowa (Ehrenreich 1959) and South Dakota (Worcester 1979), an early spring burn in Missouri (Kucera and Ehrenreich 1962), and an October lightning-caused fire in Nebraska (Morrison et al. 1986).

Earlier and increased growth is most often attributed to increased solar radiation reaching the soil following the removal of standing dead material (Ehrenreich and Aikman 1963; Hulbert 1988; Old 1969). As a result of increased solar radiation, soil temperatures on burned areas are higher than on unburned areas (Daubenmire 1968). Following spring burning in native bluestem prairie in Missouri, soil surface temperatures on burned areas compared with unburned areas averaged 7.1 degrees F (3.9 degrees C) warmer in April, 11.4 degrees F (6.3 degrees C) warmer in May, 8.3 degrees F (4.6 degrees C) warmer in June, and 7.1 degrees F (3.9 degrees C) warmer in July (Kucera and Ehrenreich 1962). Increased soil temperatures promote earlier root growth and activity, and thus earlier emergence of shoots.

Numerous authors have reported increases in flower stalk abundance (up to 1200 percent) following spring burning in tallgrass prairie (Curtis and Partch 1948; Ehrenreich and Aikman 1963; Hulbert 1988; Pemble et al. 1981). These increases are attributed to increased nitrogen availability and to the removal of the litter layer around the growing points, which increases the amount solar radiation received (Hulbert 1988; Old 1969). Long-term observations indicate that increases in flower stalk and herbage production are short-lived. Following spring burning in Iowa, flower stalk production increased dramatically the first growing season but returned to normal by the third growing season (Ehrenreich and Aikman 1963). Stem density declines as the amount of standing dead material increases. In eastern Kansas, little bluestem shows a linear decrease in abundance with time since burning (Gibson and Hulbert 1957). When little bluestem's abundance was compared on grasslands with different burning frequencies, its greatest abundance was on stands annually burned. In the Kansas Flint Hills, Towne and Owensby (1984) observed that on plots burned annually 48 out of 56 years before 1982, total herbage production of little bluestem was greatest in 1981 (compared to the previous 56 years), indicating long-term annual burning is not detrimental to little bluestem.

Discussion of plant response

Post-burn frequencies indicate that little bluestem's response to fire is influenced most by the season of burn and by the mean annual precipitation (MAP) of the burned area. When precipitation is normal or above normal, little bluestem generally increases after spring burns in the tallgrass prairie where precipitation is normally greater than 20 inches annually. An exception is the Flint Hills of Kansas, which receive about 30 inches (75 cm) of MAP. Here late spring burning neither increased nor decreased yields, but mid or early spring burning reduced yields significantly (Anderson et al. 1970; Owensby and Anderson 1967). In other areas of the tallgrass prairie, yield increases greater than 100 percent have been observed the first year following late spring burns in North Dakota (Kirsch and Kruse 1973), Minnesota (Svedarsky et al. 1986), Missouri (Kucera and Ehrenreich 1962), and Iowa (Ehrenreich and Aikman 1957). Farther west in the mixed-grass prairie, little bluestem is stimulated by fires only in areas receiving over 16 to 20 inches (40-50 cm) of MAP during years with above normal precipitation (Bailey 1978; Wright and Bailey 1980). In the Black Hills of South Dakota, which receive 15 to 17 inches (38-43 cm) of MAP, a late spring burn (May 27) increased little bluestem yields by 31 percent (Schripsema 1977). In the mixed-grass prairie, burning followed by periods of below

normal precipitation decreases yields compared to unburned stands (Anderson 1972; Wright 1974b). Late spring burning, when warm-season grasses are about to resume growth, appears to be most beneficial to little bluestem.

Fire management considerations
Late spring fires have yielded the greatest increases in little bluestem abundance (Anderson 1965; Anderson et al. 1970; Branson 1953; Kirsch and Kruse 1973; Kucera and Ehrenreich 1962). The later the burn occurs in the spring (just prior to the emergence of green shoots), the greater the herbage production will be following burning. If burning is too early, lower production may result, due to increased evaporation of soil moisture in the interval between the fire and the resumption of new growth. Soil is exposed for least amount of time following late spring fires; therefore soil moisture levels over the growing season are reduced less after late spring burns than after winter, early spring, or mid-spring burns (Anderson 1965; McMurphy and Anderson 1964).

For grazing purposes, late spring burning can be used to increase grass productivity and improve cattle use in areas receiving over 16 to 20 inches (40-50 cm) of annual precipitation. Grazing distribution may be improved because cattle prefer grasses on burned areas over grasses on unburned areas (Launchbaugh and Owensby 1978). Anderson et al. (1970) observed increased steer weight gains on late spring burns in Kansas, and improved range condition due to an increase in warm-season grasses. Chemical analysis of little bluestem following annual spring burning on April 10 in Kansas shows that its nutrient value increased slightly. When sampled in July, plants from burned stands had a protein content of 6.3 percent compared to 5.7 percent for plants from unburned stands (Smith and Young 1959). Chapin and Van Cleve (1981) observed a 10 percent increase in nitrogen, 11 percent in phosphorus, and 10 percent in calcium 3 months after burning.

Stipa comata--needle-and-thread grass

Fire ecology or adaptations
Needle-and-thread grass is susceptible to fire and can be severely damaged (Wright et al. 1979). This non-rhizomatous bunchgrass regenerates through seed after fire. Re-establishment is enhanced by unique seed characteristics and good seedling vigor (Hironaka et al. 1983).

Fire effects
Immediate fire effect on plant
Needle-and-thread is generally killed when above ground vegetation is consumed by fire. Fire effects depend on the season of burn and phenology, as well as on fire intensity and severity. Site conditions and climatic factors can also play a significant role. Needle-and-thread grasses are among the least fire resistant of bunchgrasses (Wright et al. 1979).

Morphological characteristics of needle-and-thread grass make this plant especially susceptible to fire. The dense culms can burn for 2 to 3 hours after a fire passes. Temperatures up to 1,000 degrees F (538 degrees C) are sometimes reached up to 45 minutes after the passage of the fire front (Wright et al. 1979). Above ground vegetation is generally consumed by fire. The dense, fine foliage can transfer heat below the soil surface and increase damage to the plant. Needle-

and-thread grasses often exhibit subsurface charring (Wright and Klemmedson 1965). Many plants die outright or survive with only a few living culms (Wright et al. 1979).

This species begins growth in the spring or early summer and lacks the pronounced dormant period in late summer which is typical of many other grasses. Consequently, fire is most injurious in midsummer and least detrimental in late spring or fall (Volland and Dell 1981). If needle-and-thread is burned while in the fruiting stage (generally in midsummer) when root carbohydrate reserves are low, damage can be significant. When plants are burned later in the season after carbohydrate reserves have accumulated, fire effects are less severe (Wright et al. 1965). Burning after a fall rain can minimize damage (Wright 1971).

Plant size can frequently be an important factor in determining mortality. Large plants are thought to be more susceptible to fire-induced mortality or injury than smaller plants (Wright et al. 1979). Large plants typically reach higher crown temperatures due to the presence of additional fuel volumes and a higher ratio of dead to living plant materiel (Wright and Klemmedson 1965).

According to Wright (1970), mortality is generally caused by heat except when herbage removal alone is sufficient to kill the plant. Heat usually kills plant tissue at the center before killing tissues at the periphery. The amount of heat generated by the fuels within the needle-and-thread plant is sufficient to cause mortality regardless of the amount of heat generated by the fire front sweeping over the plants (Wright 1971).

Discussion of fire effect

A number of researchers have examined the varying effects of fire on needle-and-thread grass in several seasons and geographic locations. Generally, fires conducted in midsummer were most harmful. In Wind Cave National Park, South Dakota, cover of this species was reduced by 38.2 percent after an early April fire, and 12 percent after a mid-April fire (Shown 1982). Growth was delayed more than 2 to 3 weeks past the green-up date on the unburned control plots, and recovery had not been completed by as late as August. In another study, needle-and-thread grass increased significantly 1 year after a different mid-April fire in the same area (Schripsema 1978). Some rather specific research has been focused on the effect of varying levels of heat on needle-and-thread grass. Results indicate that mortality can be due to heat alone during most months of the year (Wright 1970). Position within a natural burn, which is presumably closely related to heat intensity received by the plant, also influences the effect of fire on needle-and-thread grass (Wright et al. 1965). Soils may also influence the effect of fire. In fall burns in North Dakota the frequency of needle-and-thread was reduced 11 percent on sandy soils but increased 10 increased on clay loam (Wright and Klemmedson 1965). Limited evidence suggests that when precipitation amounts are below normal, the effects of fire tend to be more severe.

Plant response to fire

The specific response of needle-and-thread to fire depends on such factors as season of burn, size of plant, and intensity of the heat received. If not killed outright by fire, this plant generally recovers relatively slowly. Depending on the site, 3 to 8 years may be required for plants to reach pre-fire levels (Wright et al. 1979).

When above ground parts are consumed by fire, mortality generally results. Frequently, a small number of culms survive and the plant slowly recovers. Smaller plants with a lower ratio of dead to living plant material and less fuel volume generally respond more favorably than do larger plants (Wright and Klemmedson 1965). After fire, regeneration of this nonrhizomatous bunchgrass is through seed.

Discussion of plant response

Recovery time varies depending on site characteristics, climatic conditions, and grazing history. In western North Dakota needle-and-thread required 4 years to fully recover from burning. Under moderate grazing, 4 to 8 years were required for this species to reach pre-burn levels at some southern Idaho sites (Wright et al. 1979). No negative impacts of fire could be detected 12 years after a different southern Idaho fire (Linne n.d.).

Cover and frequency of needle-and-thread grass are generally reduced by fire for some period of time. In western South Dakota, needle-and-thread grass cover decreased immediately after the fire but increased relatively rapidly 2 years later (Forde et al. 1984; Shown 1982).

After burning, productivity generally shows a short-term decline. In southern Idaho, productivity of needle-and-thread grass was reduced during the first year after burning on sites where sagebrush had been eliminated, but then began to increase gradually (Wright and Klemmedson 1965). Needle-and-thread produced 10 to 26 pounds (5-12 kg) per acre more on southern Idaho plots burned 12 years earlier than on unburned control plots. On the South Dakota burns, both the size and number of inflorescences increased during the first post-fire year and decreased the second year (Schripsema 1978). Decreases in the number of inflorescences were reported in North Dakota and Idaho (Dix 1960; Wright and Klemmedson 1965).

Fire management considerations

Generally when needle grasses are burned, a 50 percent reduction in basal area should be expected. Damage to needle-and-thread grass can be minimized by preventing the buildup of high fuel loads prior to burning (Wright et al. 1979).

Riparian Shrub/Tree

Populus angustifolia--narrowleaf cottonwood

Fire ecology or adaptations
Plant adaptations to fire
Root sprouting occurs on all *Populus* species (Schier and Campbell 1976). Narrowleaf cottonwood will sprout after light to moderate-intensity fires (Hansen et al. 1989). The morphologically similar eastern cottonwood and balsam poplar develop fire-resistant bark after 15 to 20 years of age (Collingwood 1937; Fowells et al. 1987) it is likely that narrowleaf cottonwood does the same. However, young balsam poplars are susceptible to fire (Haeussler and Coates 1986), and young narrowleaf cottonwoods are probably susceptible also.

Seedling regeneration is favored following disturbances such as fire (Fowells 1965; Gruell 1980a). Fire thins the overstory, allowing more light penetration, and exposes the mineral soil such that seeds are able to establish if soil moisture is adequate (Fowells 1965).

Fire effects
Immediate fire effect on plant
Although the bark of mature narrowleaf cottonwood trees may resist fire, older trees are commonly killed by even relatively cool fires which wound trees and facilitate the onset of heartwood decay (Myers and Buchman 1984). Young narrowleaf cottonwoods are probably killed by fire.

Plant response to fire
Narrowleaf cottonwood probably resprouts following light and moderate-intensity fires (Hansen et al. 1988), although it is the least vigorous sprouter of all the *Populus* species (Schier and Campbell 1976). When fire removes competing conifers in the stand and mineral soil is exposed, cottonwood seed germination is enhanced. If fire does not occur, cottonwoods may be lost to conifer succession (Gruell 1980a; Gruell 1980b)

Fire management considerations
Fire should not be used as a management tool on sites where the maintenance of bottomland hardwoods is a management objective. Ground fires will kill narrowleaf cottonwood saplings and seedlings and facilitate the onset of heartwood decay in older trees (Myers and Buchman 1984). Long-term maintenance of cottonwood stands may require fire use where spruce.

Populus deltoides var. *monilifera* (var. *occidentalis)*-plains cottonwood

Fire ecology or adaptations
Plant adaptations to fire
The bark of older cottonwoods can be up to 4 inches (10 cm) thick at the base, affording fire protection (Fowells 1965). Trees less than 20 years old are susceptible to fire (Collingwood 1937) but may resprout. Plains cottonwood (var. *occidentalis*) is able to produce sprouts from the rootcrown and the stump after fire (Dickman and Stuart 1983; Severson and Boldt 1977).

Fire effects
Immediate fire effect on plant
Seedlings and saplings of plains cottonwood are killed by ground fires that only scar older trees (Maisenhelder 1951; Myers and Buchman 1984). Trees 6 to 10 inches (15-25 cm) in d.b.h. may be killed by a single moderate severity fire (Maisenhelder 1951). Trees are susceptible to fire until they are 15 to 20 years old and have developed bark thick enough to be fire resistant. Mature trees may be wounded by fire; these wounds may act as entry points, thus facilitating the onset of heartwood decay (Myers and Buchman 1984).

Plant response to fire
Plains cottonwood may sprout following fire (Severson and Boldt 1977). Nearly all poplars readily sprout following harvesting or when killed by fire (Dickman and Stuart 1983).

Fire management considerations
Prescribed fire is not recommended on bottomland forest sites where wood production is a primary management objective. Seedlings and young trees are easily killed by ground fires, while mature trees are often wounded (Bull and Muntz 1943). Wounded trees often contract heartrot, leading to substantial cull and volume loss (Bull and Muntz 1943). On the other hand, prescribed burning may stimulate sprouting and improve food supplies for wildlife species such as moose and beaver (Severson and Boldt 1977).

Salix amygdaloides-peachleaf willow

Fire ecology or adaptations
Plant adaptations to fire
Peachleaf willow sprouts from its roots following fire (Hansen et al. 1988). The high soil and fuel moisture content characteristic of its streamside habitat reduces the chance of fire ignition and spread. Its numerous wind-dispersed seeds are also important in revegetating areas following fire (Zasada 1986).

Fire effects
Immediate fire effect on plant
Most fires kill only above ground plant parts. However, severe fires can completely remove soil organic layers, leaving willow roots exposed and charred, and thus eliminating basal sprouting (Kovalchik 1987; Rowe and Scotter 1973; Zasada 1986).
Plant response to fire

Generally peachleaf willow will resprout from its roots following fire (Hansen et al. 1988). Peachleaf willow is a prolific seeder, and off-site plants are important seed sources for revegetating burned areas (Zasada 1986).

Fire management considerations
Due to their proximity to water or high water tables, peachleaf willow communities may serve as natural fire breaks (Hansen et al. 1988).

Salix exigua-coyote willow

Fire ecology or adaptations
Plant adaptations to fire
Sandbar willow sprouts from its roots after fire (Conrad 1987; Rowe and Scotter 1973; Zasada 1986). Its numerous wind-dispersed seeds are also important in revegetating burned areas (Rowe and Scotter 1973). The high soil and fuel moisture content characteristic of its streamside habitat reduces the chance of fire ignition and spread.

Fire effects
Immediate fire effect on plant
Most fires kill only above ground plant parts. However, severe fires can completely remove organic soil layers, leaving willow roots exposed and charred, thus eliminating basal sprouting (Bellah and Hulbert 1974; Bernard and Brown 1977; Boyd et al. 1986).

Plant response to fire
Generally sandbar willow will sprout from its roots following fire (Rowe and Scotter 1973; Zasada 1986). Irwin (1985) found that sandbar willow was common in the boreal forest of northeastern Minnesota 2 years after a May wildfire. Sandbar willow is a prolific seeder; off-site plants are important as a seed source for revegetating burned areas (Zasada 1986).

Fire management considerations
Due to its proximity to water or high water tables, sandbar willow communities may serve as natural fire breaks (Arno and Hammerly 1977).

Salix lutea-yellow willow

Fire ecology or adaptations
Generally yellow willow has the ability to sprout from its roots or stem base following fire (Hansen et al. 1988; Zasada 1986). When found along a streamside, the high soil and fuel moisture content characteristic of this habitat reduces the chance of fire ignition and spread. Its numerous wind dispersed seeds are important in revegetating areas following fire (Zasada 1986).

Fire effects

Immediate fire effect on plant

Severe fires can completely remove organic soil layers leaving willow roots exposed and charred, and thus eliminating basal sprouting. However, most fires kill only above ground plant parts (Kovalchik 1987; Rowe and Scotter 1973; Zasada 1986).

Plant response to fire

Generally yellow willow will sprout from its roots or stem base following fire (Hansen et al. 1988; Rowe and Scotter 1973; Zasada 1986). It shows better recovery from quick hot fires, as slow burns are more damaging to plants (apparently burning down into the roots) (Haessler and Coates 1986; Hansen et al. 1988; Kovalchik 1987). A prolific seeder, off-site plants are important in revegetating burned areas through the dispersal of numerous wind and water transported seeds (Zasada 1986).

Fire management considerations

Quick hot fires may be used to rejuvenate decadent willows, thus producing abundant browse for big game animals (Haessler and Coates 1986), however it may take 5 or more years for willows to regain stem height and diameters resistant to browsing (Kovalchik 1987).

Riparian Grass/Forb

Agrostis scabra-redtop or ticklegrass

Fire ecology or adaptations

Wind-dispersed ticklegrass seeds readily colonize bare mineral soil on recently burned sites (Carroll and Bliss 1981; Johnson 1975; Smith 1970). Seeds may also be stored for short durations in the soil, allowing for early establishment of areas burned in the spring (Fyles 1989).

Fire effects

Immediate fire effect on plant

Fire generally top-kills ticklegrass. Specific effects on ticklegrass mortality, however, are not well documented.

Plant response to fire

In general, ticklegrass increases in abundance in response to fire. Seedlings immediately colonize recently burned areas, provided a favorable seedbed has been established (Lutz 1956; Stickney 1985). Annual spring burns in aspen stands in Alberta caused an increase in ticklegrass inflorescence production. In unburned areas, there was an average of one flower head per square foot (10/sq m), but on burned sites 10 flower heads per square foot (110/sq.m) were produced (Anderson 1959). In interior Alaska, seedlings were not found in burned plots where the organic layer had not been completely removed, although a seed source was nearby. Seedlings were, however, abundant on adjacent firelines (Viereck 1982).

Eleocharis palustris-spikerush

Fire ecology or adaptations
Common spikerush is adapted to fire because of its rhizomes (Millar 1973; Young 1986).

Fire effects
Immediate fire effect on plant
Common spikerush is usually top-killed by fire (Young 1986).

Discussion of fire effect
Prescribed burns will reduce the litter layer in wetlands but often do not affect plant species composition unless the organic layer is burned (Kovalchik 1987).

Discussion of plant response
Common spikerush sprouted following prescribed burning in Saskatchewan (Millar 1973). Following prescribed burning in October in Oregon, common spikerush's above ground standing crop was nearly twice that of unburned plots (Young 1986). One hundred percent of the shoots produced flowers in both the fall-burned and unburned plots.

Fire management considerations
Young (1986) cautions that wetlands and their respective upland communities should not be managed separately because of the combined importance of both for wildlife habitat. Thus, the effects of fire on one community should be considered in conjunction with the effects on the other community. He also recommends defining the soil-peat moisture conditions before burning wetlands.

Juncus balticus-baltic rush

Fire ecology or adaptations
Baltic rush can survive fire by sprouting from its extensive rhizomes (Parker 1975).

Fire effects
Immediate fire effect on plant
Fire in wetland communities often only top-kills plants, leaving rhizomes in moist soil unharmed (Young 1986).

Plant response to fire
Baltic rush response to fire can vary. A September prescribed fire in a wetland meadow resulted in no significant change in ground cover of Baltic rush 4 years following the fire (Hargis and McCarthy 1986). However, following a May prescribed fire in sub-irrigated rangeland dominated by Baltic rush and Kentucky bluegrass, cover of Baltic rush increased from 35 percent to 60 percent by the fifth post-fire year (Bailey and Anderson 1979).

Fire management considerations

Above ground biomass has been measured for Baltic rush at 572.8 g m-2 dry weight in August in Ontario (Glooschenko and Harper 1982). This was the highest biomass of all salt marsh species in this community.

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APPENDIX A

Methods for Baseline Survey

Vegetation Sampling Methodology (Claire DeLeo)

A modification of the cover-frequency transect method from the Rangeland Analysis and Management Training Guide (USDA Forest Service 1994) was used for the study. Woody vegetation was quantitatively sampled for percent cover using three 25 meter line intercept transects measured to the nearest centimeter. Overhead tree cover was measured along the transect using a clinometer to find the vertical intercept of the tree cover. Three transects were located per site with the random placement of transects.

For the Farish study, the location of the three transects were defined as a plot. Each plot was randomly located by overlaying a grid on a map of the area to be burned and random numbers generated for coordinates on the grid. At each pair of coordinates chosen, a plot was placed by running a 50 meter tape in a northernly direction from the randomly selected point. Placing a transect every 5 meters starting at the 5 meter mark, nine possible transects exist along the 50 meter tape, excluding the 50 meter mark. Three 25 meter transects were randomly chosen. These transects were placed perpendicular to the first 50 meter line in a westerly direction (Figure 1a and 1b). Plots or sites were marked with a T- post so that they may be located for post-burn monitoring.

Herbaceous vegetation was sampled using 0.1 m^2 (20 cm x 50 cm) Daubenmire quadrats. Percent cover was estimated to the nearest percent by herbaceous species within the Daubenmire quadrats. Ten quadrats were located along each transect for a total of thirty quadrats per site. One quadrat was placed every 2.5 meters along the transect, starting at the 1 meter mark, and 1 meter to the side, alternating sides (Figure 2). Plot photos were taken in a northwesterly direction before and after burning and while data were being collected at the 0 meter and 25 meter marks on the 50 meter transect.

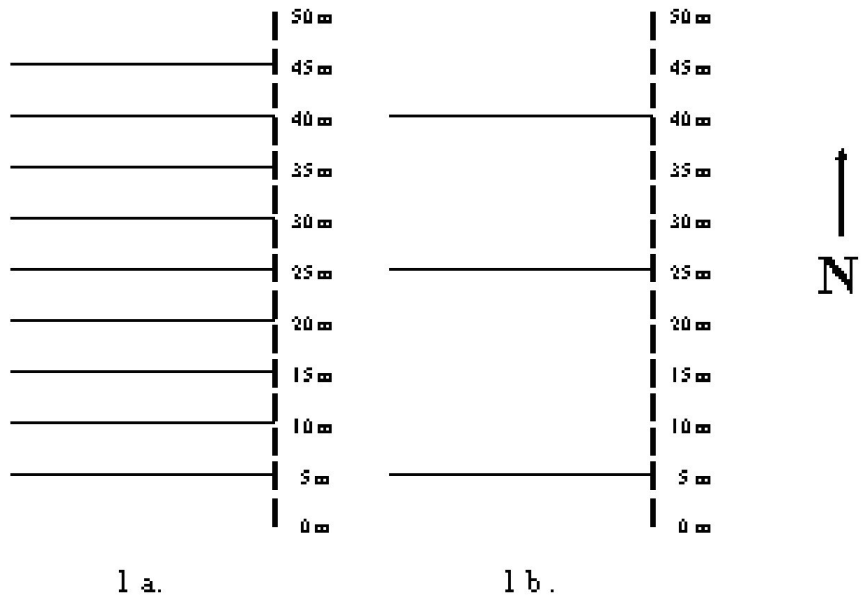

Figure 1 a. and b. The dashed lines represent the 50 meter lines. The solid lines represent the 25 meter transects to be sampled. Figure 1a. on the left represents all the possible transects on a 50 meter line within a plot. Figure 1b. on the right demonstrates a random sampling of three transects.

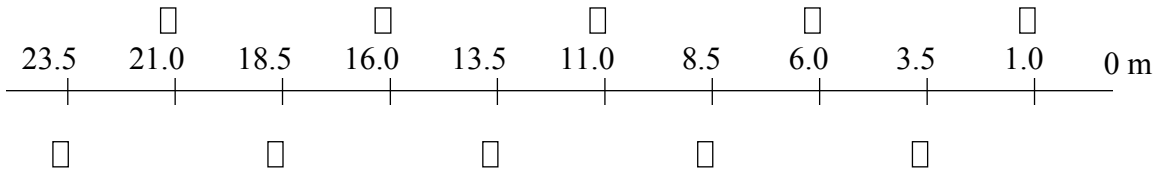

Figure 2. Location of Daubenmire quadrats along a transect. Numbers represents meter marks. Rectangles represent Daubenmire quadrats.

Sampling Design

Control and Treatment Plots

To better determine cause and effect, control plots were established in addition to fire treatment plots. A plot would be an area that is sampled by the proposed methodology above. Ideally, fire treatment plots should be burned separately for statistical analysis of the data. A paired t-test could then be used to statistically analyze the data, with appropriate transformations for percent data.

For the Farish study, the meadow north of Schubarth Road was burned in the fall of 1996 and the meadow south of Schubarth Road will be burned in the spring of 1997. Six plots were established for the fall 1996 burn and six plots for the spring 1997 burn. A total of six control plots were established, three in the north meadow and three in the south meadow. Fire was kept out of the control plots by applying fire resistent foam around the plots.

Monitoring Plots

Plots could be monitored before and after burning, without control plots. This study would be able to detect a change between before burning and after burning, as with the experimental design above. The difference between the monitoring study and the experimental study is that only the experimental study can infer the *cause* of a change (USDI Bureau of Land Management 1996). Monitoring is less time consuming than the experimental study, and more sites could be sampled if desired.