Natural Heritage Baseline Survey
and
Fire Ecology/Effects Literature Review
for
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Compiled by:
Denise Culver, Assistant Wetland Ecologist

The Colorado Natural Heritage Program
General Services Building Room 254
Colorado State University
Fort Collins, Colorado 80523

Table of Contents

List of Tables

List of Figures

Executive Summary

The natural resource planners at the U.S. Air Force Academy (USAFA) in Colorado Springs, Colorado contracted the Colorado Natural Heritage Program (CNHP) to establish a monitoring study to investigate the effects of fire on plants, animals, and natural communities on USAFA lands. 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.

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 and continues to be an important process that contributes to the biodiversity of the flora on USAFA. Four globally rare natural communities occur on USAFA (big bluestem-little bluestem xeric tallgrass prairie, big bluestem-prairie dropseed herbaceous vegetation, montane grassland, and Great Plains mixed grass prairie). Additionally, two globally rare plants (Porter's feathergrass and southern Rocky Mountain cinquefoil), one federally listed threatened mammal (preblei subspecies of meadow jumping mouse), and several state rare plants and animals occur within the 18,455 acres of USAFA (BCD 1997).

This report presents the results of a thorough scientific literature review of fire effects and history for 4 major vegetation zones that occur on USAFA. Discussions of fire history, regimes, case studies, and biological diversity are included, as well as fire effects and ecology of 6 noxious plants and 7 common animals. A detailed section adapted from the U.S. Forest Service Fire Effects Information System (FEIS 1997) that discusses the fire effects and ecology for 33 dominant plants within the major vegetation zones is located in Appendix A.

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 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) and the desired ecological effects to management goals. 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). 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.

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. The 18 permanent plots established at Farish Memorial Recreation

Area provide the groundwork for a long-term study of fire effects and processes in the mixed coniferous forest vegetation type.

Prescribed Burning Guidelines

- **Jacks Valley and west of Rifle Range--prairie grasslands and oak shrubland**

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 factored in.

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 oak is seldom killed by fire, however control of Gambel 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.

- **Foothills and Farish Memorial Recreation Area--ponderosa pine and mixed conifers**

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 fire fuel moisture and fuel production.

Use moderate to high intensity fires to create patch mosaics that increase species diversity and improve the heterogenity 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 year. 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 forests is fuel moisture.

Prescribed Burning Guidelines (continued)

- **Monument Creek--riparian forests**

Low to moderate intensity fire 2-5 every 100 years will encourage root sprouting in cottonwoods and willows. Fire thins the overstory, allowing more light penetration thereby exposing the mineral soil so that seeds are able to establish if soil moisture is adequate.

- **Animals**

Prescribe burning in likely habitats (e.g., riparian areas) should be performed between October and late April when the preblei subspecies of meadow jumping mouse is in hibernation. Additional research is needed to determine specically the fire effects on this federally threatened 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 should 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. Butterfly nectar or host plants need to be factored in 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 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 33 native dominant plants, 4 vegetation types, 6 noxious weeds, and 7 common animals that occur on USAFA. A thorough literature search using current available information including the U.S. Forest Service Fire Effects Information System (FEIS 1997), scientific journals, and local fire experts is included. Data from 18 permanent plots established at Farish Memorial Recreation Area are located 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 changes 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 its 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, agreeds stating that historical fire frequencies cannont 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 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* sp.) 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 smooth brome (*Bromus inermis*), cheatgrass (*Bromus tectorum*), 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. USAFA is interested in improving wildlife habitat in the riparian locations.

The occurrence of the preblei subspecies of the meadow jumping mouse (*Zapus hudsonius preblei*), a federally listed threatened species is of concern in this area. Colorado Natural Heritage Program located the preblei subspecies of meadow jumping mouse in 1995 in this area (Corn et al. 1995). Monument Creek is the best known global occurrence for this subspecies of meadow jumping mouse (Biological Conservation Data System 1997). Little is known about the ecology of this subspecies in Colorado, and the effects of fire on the habitat of the meadow jumping mouse is unknown.

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 an area targeted for burning in 1997. The purpose of reintroducing fire to this area is to reduce fuels and restore the ecological function of fire 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 Ranger District and would be on a large scale, approaching 400 acres or more. The purpose of burning oak scrub is to improve bighorn sheep habitat, creation of corridors, and increase patch dynamics.

Fire, Landscape Pattern, and Biological Diversity

Fire is a major force in structuring landscape 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 (colonizing species may increase, biological diversity is reduced). 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.

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. In a study of grassland communities in the Serengeti National Park, Joy (1992) states that species diversity increased when after burning.

Biodiversity has been variously defined over the past 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.". It is important to point out that diversity is a mathematical measure, while the notion of biodiversity is more holistic and inclusive (Alington 1996).

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. The incorporation of the biodiversity of plants, animals, and natural communities located on USAFA is integral to CNHP's mission. There are four globally rare natural communities occurring on USAFA (big bluestem-little bluestem xeric tallgrass prairie, big bluestem-prairie dropseed herbaceous vegetation, montane grassland, and Great Plains mixed grass prairie). Additionally, there are two globally rare plants (Porter's feathergrass and southern Rocky Mountain cinquefoil), one federally listed threatened mammal (preblei subspecies of meadow jumping mouse), and several state rare plants and animals are known to occur within the 18,455 acres of USAFA lands (Biological Conservation Data System 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 type occurs broadly in various areas of North America, but on USAFA, they occur in close proximity to one another, creating a unique mosaic of plant species.

Ripley (1994) and Pat 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 gracile*), prairie sand reed (*Calamovilfa longifolia*), Parry oatgrass (*Danthonia parryi*), hairy grama grass (*Bouteloua hirsutum*), purple pinegrass (*Calamagrostis purpurascens*), Japanese brome (*Bromus japonicus*), smooth brome (*Bromus inermis*) and cheatgrass (*Bromus tectorum*).

Oak shrubland
Dominated by Gambel oak (*Quercus gambelii*), pinyon 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 engelmanii*), quaking aspen (*Populus tremuloides*), white fir (*Abies concolor*), bush oceanspray (*Holodiscus dumosus*), and common snowberry (*Symphoricarpos albus*).

Riparian
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 significant natural communities, plants, vertebrates, and invertebrates for USAFA. (See Tables 7 and 8 for a complete description of natural heritage, federal, and state ranks).

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

Common name	Scientific name	Global rank	State rank	Fed status	State status	Fed sens
xeric tallgrass prairie	*Andropogon gerardii-Schizachyrium scoparium*	G2	S2			
xeric tallgrass prairie	*Andropogon gerardii-Sporobolus heterolepis*	G2	S2?			
montane grassland	*Danthonia parryi*	G2?	S2?			
Great Plains mixed grass prairie	*Stipa comata*	G2	S2			

Table 2. 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 3. 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 4. State rare 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 5. Globally rare mammal occurring on U.S. Air Force Academy.

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

Table 6. 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 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

Colorado Natural Heritage Methodology

The Natural Heritage Methodology operates at several different levels. First, **elements of natural diversity** are ranked according to their rarity and/or degree of imperilment. These **elements** consist of rare or imperiled species, subspecies and significant natural communities. The relative rarity of the various elements is based upon the scientific biological information and population locations known currently. As new information is acquired, element ranks can be modified.

The second level of the Natural Heritage Methodology is the ranking of the populations or **occurrences** of a particular element. Since it is frequently impossible to protect all populations of a particular species, subspecies, or natural community, attempts are made to evaluate the relative quality of various occurrences of these elements so that conservation efforts can be focused on the best representatives of the elements and the healthiest, most viable populations.

The third level of the Natural Heritage Methodology is the delineation of potential conservation sites and the ranking of these sites. This ranking is based on the rarity and quality of the element occurrences contained within the sites. This enables conservation efforts to focus on assemblages of rare elements as well as on the elements themselves. A comprehensive, scientific approach to protecting species results when these three levels of Natural Heritage Methodology are applied.

Element Ranking

CNHP uses an element ranking system emphasizing the number of occurrences at distinct localities as an index of known biological rarity. The primary criterion for ranking elements is the number of occurrences because an element found in one place is more imperiled than an element found in twenty places. Also of importance is the size of the geographic range, the number of individuals, trends in both population and distribution, identifiable threats, and the number of already protected occurrences. Each element is assigned a rank that indicates its relative degree of imperilment on a five point scale:

1 = critically imperiled because of extreme rarity; five or fewer occurrences;

2 = imperiled because of rarity; 6 to 20 occurrences;

3 = very rare or vulnerable; generally found in a restricted range; 21-100 occurrences;

4 = apparently secure but may be declining; and

5 = demonstrably secure.

Element imperilment ranks are assigned in terms of imperilment within Colorado, the state rank, and the element's imperilment over its entire range, the global rank. The global rank, or G-rank, sets the overall priorities. The state rank, or S-rank, is used in discerning local, regional, and state priorities. For example, an element with a rank of G3/S2 will receive higher priority than an element with a rank of G5/S1 due to its global rank. Together these two ranks provide an instant picture of an element's degree of imperilment or rarity. It should be noted that an element can never be more common within a state than it is globally. Therefore, the element's S-rank will always be as rare as the global ranking, i.e., G3/S2 not G2/S3.

Elements that receive a rank of S1, S2 and S3 are used to set species protection priorities. Elements with a ranking of S3S4 are "watchlisted"; data is collected and periodically analyzed to determine if more active tracking is warranted. Any element more common than a "watchlisted" element, with an S-rank of S4 or S5, is not monitored. Accepted subspecies are also included on the CNHP list (with associated trinomial ranks, or T-ranks), but they receive less priority than an equivalently ranked or imperiled species.

This single ranking system identifies all imperiled elements except those that are migratory. When ranking migratory elements it is necessary to distinguish between breeding, non-breeding, and resident species. A rank followed by a "B", e.g., S1B, indicates that the rank applies only to the status of breeding occurrences. Ranking followed by an "N", e.g., S1N, refers to non-breeding status, typically during migration and winter. Elements without this notation are believed to be year-round residents within the state. A complete description of each of the Natural Heritage global and state ranks is provided in Table 7.

Colorado Natural Heritage Program

The Colorado Natural Heritage Program is building on a solid base of biodiversity information. In 1992, after 14 years of operation with the Division of Parks and Outdoor Recreation, CNHP was relocated in the University of Colorado Museum. Quickly outgrowing available space, CNHP transferred its offices to Colorado State University's College of Natural Resources in September 1994. CNHP has established itself as a statewide repository for information on rare or imperiled species and significant natural communities in Colorado. The multi-disciplinary team of scientists gather information and information managers continually incorporate these data into CNHP databases. CNHP is part of an international network of conservation data centers that use the Biological and Conservation Data System (BCD, developed by The Nature Conservancy). Concentrating on site-specific data for each element of natural diversity, the accurate status of each element is known. Maps of the data contained in BCD illustrate sites that are important to the conservation of Colorado's natural heritage. By using the element ranks and the quality of each occurrence, priorities can be established for the protection of the most sensitive sites. This updated locational database and priority-setting system provides CNHP with effective, proactive land-planning tools.

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 was their ability to die down to underground roots, exposing only dead tops aboveground. 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 83-680 degrees C (181-1256 degrees F) (Wright 1974b; Rice and Parenti 1978). However, soil is a good insulator, thus there is little penetration of heat more than a centimeter below the soil surface (Anderson 1982).

Fire can act as a stabilizing or destabilizing factor in vegetation, depending upon fuel availability and quality 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 anytime 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 semidesert 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 of 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 aboveground 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 if burned 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 (*Muhlenbergi montana*) increase significantly in number and yield following a spring fire; prairie junegrass (*Koeleria macrantha*), sand dropseed (*Sporobolus cryptandrus*), blue gramma (*Bouteloua gracile*), and hairy gramma (*Bouteloua hirsutum*) show lesser increase; and buffalograss (*Buchloe dactyloides*) is unaffected (Pyne et al. 1996). Annuals (e.g., cheatgrass) 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 assure 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, however, which initiate growth earlier in the spring than native species and continue growing later in the fall (cool season grasses), 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 presettlement 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, woodlots, 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).

<u>*Case Studies*</u> (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 scabrell, 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 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 cristata*), needle and thread grasses (*Stipa spartea, S. comata*), rough fescue (*Festuca scabrella*), blue grama (*Bouteloua gracile*), and bluegrass (*Poa secunda*). Researchers found that on ungrazed burned sites, spring burning decrease plant yield by 50% the year of the burn but that yield returned to pre-burn levels within two years. Fall burning did not decrease yield as drastically in the fires growing season following a burn but pre-burn production levels also returned 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.

Gruell et al. (1986) present a reference to aid managers in improving productivity of bunchgrass in areas that have been invaded by shrubs and trees. General guidelines to improve range and wildlife habitat with prescribed burning are provided for six different habitat types: 1). Douglas-fir seedling stage in Douglas-fir/Idaho fescue, 2). Douglas-fir sapling stage in Douglas-fir/Idaho fescue, 3). Douglas-fir pole stage in Douglas-fir/rough fescue, 4). Douglas-fir sapling/pole stage in curlleaf mountain mahogany on Douglas-fir/Idaho fescue, 5). Douglas-fir pole stage in aspen in Douglas-fir/rough fescue, and 6). Douglas-fir sapling/pole stage in Douglas-fir/pinegrass. For each situation, information is provided on site characteristics, vegetation characteristics, vegetation trend, response potential. Prescription considerations are also included and resource and fire objectives for the burn, type of fire desirable for the site, and dealings with fuels are given.

Oak Shrubland

<u>*Fire History*</u>

Gambel 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 oak. Gambel 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 codominates with Gambel oak on USAFA lands. In open, dry habitats where mountain-mahogany is likely to occur, fires in presettlement 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

Gambel oak is seldom killed by fire. Gambel oak generally sprout 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). Fire promotes root sprouting and the formation of buds on rhizomes (McKell 1950; Reynolds et al. 1970). In a Colorado study, Gambel 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 oak can readily reoccupy a site through seed protected in buried caches of rodents (Neuenschwander n.d.).

Gambel oak often grows in very dense thickets on steep slopes creating heavy leaf litter accumulation resulting 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 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 oak. 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.

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 in a management objective. Kufeld (1983) recommended that "prescribed burning rather than spraying or chaining be used to manage Gambel oakbrush rangelands for elk, deer, and cattle".

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 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 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.

Mixed Coniferous Forests

<u>*Fire History of Ponderosa Pine*</u>
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 have 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 (Sweetnam 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 9). Veblen articulated in a recent telephone interview (April 3, 1997) that the mean fire interval for Great Plains/ponderosa pine ecotone in Boulder County is 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 (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.).

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

Mean Fire Interval (years)	
Roosevelt National Park, 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, with the result that new age groups of trees and considerable amounts of dead material exist in stands that otherwise might have had the classic, park-like appearance with numerous grassy openings characteristic of presettlement 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 increase 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 forests; too early in the fuel cycle and there will be insufficient fuel to carry an effective fire; too late and fires may be intense with high tree mortality resulting. 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 increases 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, is 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 ranges 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 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 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-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 insect, such as the mountain pine beetle (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).

<u>*Fire Regime of Douglas-fir, white fir, Colorado blue spruce, and aspen*</u>
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, because of greater shade tolerance of the dominant tree species, the snow pack persists longer into spring. Moreover, the shaded conditions limit the development 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 fires and equally or slightly less fire resistant than ponderosa pine. In the southwest, frequent surface fires in dry Douglas-fir habitats would maintain seral stands of ponderosa pine and/or southwestern white pine (Moir and Ludwig 1979). Exclusion of fire will lead to the establishment of dense Douglas-fir sapling thickets (Wright and Bailey 1982).

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 spruce to fire damage is mitigated somewhat by the moist and cool sites where it grows (Crane and Fischer 1986). Pockets of 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, stimulate 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 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).

Many believe that the majority of aspen communities probably only burned naturally under fairly extreme fire condiitons (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 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 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 mountain maple) are being replaced by dense understories and thickets of shad-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). 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). 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).

Cottonwood seedling regeneration is favored following disturbances such as fire and flood (Fowells 1965). Fire thins the overstory, allowing more light penetration, and exposes the mineral soil so that seeds are able to establish if soil moisture is adequate.

Willows
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. Peachleaf willow 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 unpredictabel and often uncontrollable nature of fire. Few studies (e.g., Rassman 1993) are planned carefully around controlled burns.

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 above, 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) (Table 10).

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 has 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 USAFA.

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)	Great Plains
		1 every 5-10 yrs (Wright et al. 1978)	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. 1985)	Colorado Unknown
Mixed Coniferous			
Ponderosa pine	low	1 every 2-10 yrs (Swetnam 1990)	southwest
		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)	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
Colorado blue spruce (wet site)	moderate	1 every 350-400 yrs (Crane 1982)	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

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).

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 pratense*) (Anderson 1965).

Whisenant (1990) suggested that Japanese brome post-fire response is best explained as a function of litter level reduction by fire and the amount of fall precipitation following fire. Since litter accumulations are more critical for seed germination and seedling establishment in dry years, populations are reduced when burning is followed by below-average precipitation. During wet years, fire has little impact on subsequent generations. 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). Gartner et al. (1978) recommended burning Japanese brome in the ripe seed stage in order to maximize kill of seeds in panicles.

Smooth brome (*Bromus inermis***)**

Smooth brome is fairly tolerant to fire when dormant and in 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). The chances of survival are increased when the seeds are covered by soil (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). The success of cheatgrass after fire has often been attributed to its capability to rapidly occupy the 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 number 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 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 The 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, preserve manager, April 1997 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). Melgoza and Nowak (1991) concluded that cheatgrass reduces root length densities of native species (e.g. rabbitbrush and needle and thread grass).

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. 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). However, as few as 43 seeds per square meter are required to reduce establishment of crested wheatgrass (*Agropyron cristatum*) and 633 seeds per square meter may prevent the establishment of perennials (Evans 1961).

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 aboveground 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). It 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 probably resists low-severity fire because of its stout taproot. Spotted knapweed probably colonizes after fire from seeds buried in soil or from off-site sources. Low-severity fire probably top-kills spotted knapweed. Buried seeds probably remain undamaged by most fires. Spotted knapweed shows moderate increases after fire (Noste 1982). Established plants may regrow and/or buried seed may germinate after fire.

On a western Montana site prescribed burned in the fall, prefire spotted knapweed herbaceous volume was 238 cubic feet per acre. There were 0.57 to 1.21 tons per acre of fine (0-0.25 inch class) fuel. The fire had a rate of spread of 2,640 to 3,393 feet (805 to 1,126 meters per hour), and flame height of 8 to 10 feet (2.4-3.1 m). Knapweed was not present in post-fire year 1, but was present at 476 cubic feet per acre herbaceous volume in post-fire year 2 (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 prefire 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 immediate effects of fire upon Russian-thistle were not found in the literature. 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).

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.

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*). They utilize the riparian communities dominated by cotonwoods 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 the Prairie Falcon (*Falco mexicanus*).

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, chipmunks 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 killled 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 creating mosaics (Oates 1995).

Fire Effects on USAFA's Common Fauna

Mule deer (*Odocoileus hemionus*)

Although uncommon, mule deer can be trapped and killed by fast-moving fires (Cowan 1956; Hines 1973). In general, fires that create mosaics of forage and cover are beneficial. 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.

Burning in grassland communities reduces litter that otherwise inhibits new growth of grasses. 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 pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) 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 more after 15 years (McCulloch 1969; Klebenow 1985). Klebenow (1985) reported that the beneficial effects of fire for mule deer in pinyon-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 pinyon-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). 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. A mosaic of seral stages is best for mule deer (Gruell 1986).

Prescribed burns in oak woodlands can create access through to the understory forage of the oak woodlands (Klinger et al. 1989). Biswell (1989) recommended burning oak woodlands every 30 years to create a mosaic of young stands. Late summer or early fall burning promotes the highest seed crop for most species in these plant communities.

Fire can control pinyon-juniper woodlands by maintaining them in a subclimax state (Blackburn et al. 1975). 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). To maintain forage in bunchgrass communities, burning at 4- to 6-year intervals in winter or early spring is recommended (Johnson 1989).

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 oak in autumn during or immediately following leaf fall and building fire breaks 26 feet wide (8 m) around the areas to be burned. Because Gambel oak recovers quickly following fire, particularly at low elevations where mule deer winter, its growth must be monitored and retarded to improve mule deer habitat (Kunzler and Harper 1980).

Elk *(Cervis elaphus)*

Young calves can be trapped and killed by fire, although losses are probably not significant (Kramp et al. 1983). Following fire, the most preferred elk forage species are enhanced by an increase in nutrients (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). No information is available regarding the direct effects of fire on bighorn sheep.

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). Prescribed burning has been used to establish and maintain subalpine bighorn sheep range in British Columbia. According to Bentz and Woodard (1988) burning provides an economical method of converting subalpine forests, which are of low value to sheep, to earlier seral plant communities. In this range, bighorn sheep used burned sites more than adjacent unburned sites.

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 altered by reducing bighorn sheep concentrations; however, 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)*

Coyotes are very mobile and can probably escape most fires. There are no reports of direct coyote mortality due to fire (Nichols and Menke 1984). 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 et al. 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. Red-tailed Hawks are more abundant in recently burned chaparral areas than in unburned areas due to greater visibility and less cover for prey (Nichols and Menke 1984). Additionally, Red-tailed Hawks habitat is favored by fires that open up or clear pinyon-juniper woodlands (Mason 1981). Raptors associated with pinyon-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. Regular burning helps to keep habitats in a suitable condition for many prey species of the Golden Eagle and increases hunting efficiency (Landers 1987). In forested areas of the East, Golden Eagles forage on burns, though they may prefer bogs (Palmer 1988). 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 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 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).

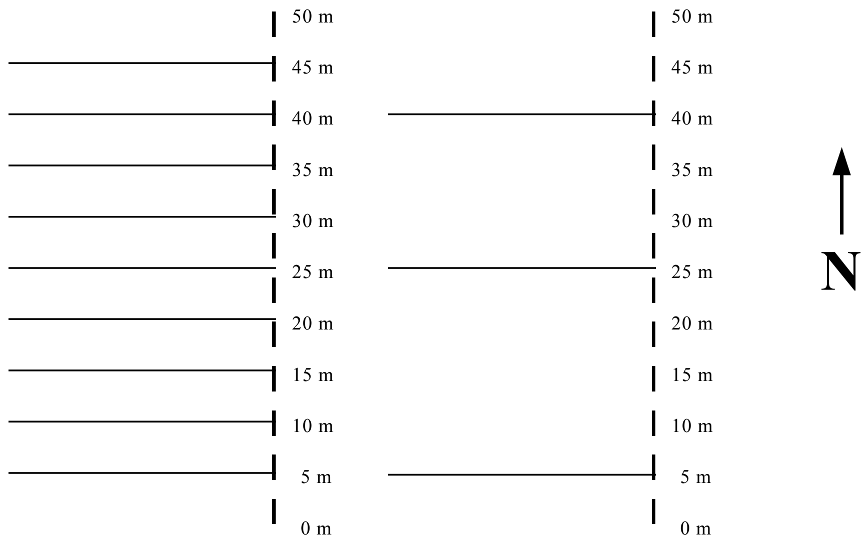
Methods

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. In the riparian areas, transects were located parallel to the stream channel and in the uplands, transects will be located perpendicular to the slope. 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 1 a 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.



1 a. 1 b.

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.

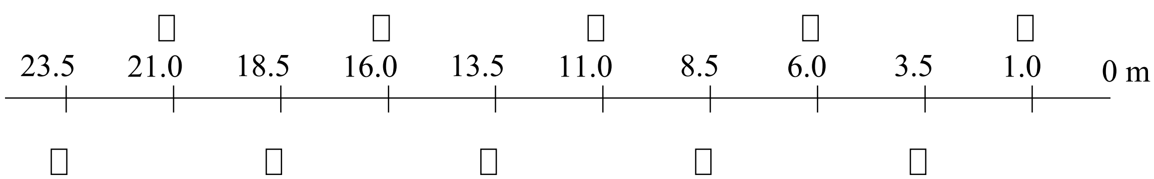


Figure 2. Location of Daubenmire quadrats along a transect. Numbers represents meter marks. Rectangles represent Daubenmire quadrats.

Sampling Design

<u>*Control and Treatment Plots*</u>

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.

Collection of vegetation and environmental data

Within stands of relatively undisturbed vegetation the following data was collected:

- Percent canopy cover by vascular plant species to the nearest 10% in the following cover classes: 5-15%, 16-25%, 26-35%, 36-45%, 46-55%, 56-65%, 66-75%, 76-85%, 86-95%, and >95%.   Plant cover 5% or less was estimated into two categories, <1% and 1-5%

- Total canopy cover by life-form (trees, shrubs, graminoids, and forbs). Overhead tree cover was measured along the transect using a clinometer to find the vertical intercept of the tree canopy

- Ground cover of bare soil, litter, wood, gravel, rock, bryophyte, and non-vascular plants

- Signs of wildlife or domestic livestock utilization

- Signs of disturbance (flooding, fire, wind throw, logging, etc.)

- Successional relationships where trends could be inferred

- Reference site and plot 35 mm color slides

- Size of occurrence mapped on 7.5 min. USGS topographic maps elevation (from 7.5 min. topographic maps)

- aspect

- history of use when obtainable

Each site was ranked A (highest) through D (poorest) for quality, condition, defensibility, and viability, using the following criteria:

quality--overall size, connectedness to surrounding natural ecosystems, degree of alteration.
condition--abundance of non-native plant species, degree of soil compaction, amount of species composition change by livestock grazing, degree of human disturbance, appropriateness of management for riparian ecosystem health.
viability--extrinsic factors: are natural hydrological processes in place, will site improve, or remain in current condition with current management?
defensibility--extrinsic and intrinsic factors affecting the long term existence of the ecosystem, define any known threats, site specific problems, and adjacent land use compatibility.

All plants not identified in the field, particularly of difficult genera such as *Salix, Carex*, and *Juncus*, were collected, pressed, and identified (to species level when possible) at Colorado State University herbarium. Voucher specimens were deposited at the Colorado State University herbarium.

Zoology Component

An extensive zoological inventory for the Air Force Academy was completed last year by the Colorado Natural Heritage Program (Corn et al. 1995), and will not be a part of this fire ecology study. If the Monument Creek area is burned, trapping will be conducted to monitor the preblei subspecies of meadow jumping mouse.