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Cascading Effects of Fire Exclusion in Rocky Mountain Ecosystems:

A Literature Review

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Abstract

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The health of many Rocky Mountain ecosystems is in decline because of the policy of excluding fire in the management of these ecosystems. Fire exclusion has actually made it more difficult to fight fires, and this poses greater risks to the people who fight fires and for those who live in and around Rocky Mountain forests and rangelands. This paper discusses the extent of fire exclusion in the Rocky Mountains, then details the diverse and cascading effects of suppressing fires in the Rocky Mountain landscape by spatial scale, ecosystem characteristic, and vegetation type. Also discussed are the varied effects of fire exclusion on some important, keystone ecosystems and human concerns.

Keywords: wildland fire, fire exclusion, fire effects, landscape ecology

Research Summary

Since the early 1930s, fire suppression programs in the United States and Canada successfully reduced wildland fires in many Rocky Mountain ecosystems. This lack of fires has created forest and range landscapes with atypical accumulations of fuels that pose a hazard to many ecosystem characteristics. The health of many Rocky Mountain ecosystems is now in decline because of fire exclusion; fire exclusion has actually made it more difficult to fight fires, and this poses greater risks to the people who fight fires and for those who live in and around Rocky Mountain forests and rangelands. This paper discusses the extent of fire exclusion in the Rocky Mountains, then details the diverse and cascading effects of suppressing fires in the Rocky Mountain landscape by spatial scale, ecosystem characteristic, and vegetation type. A description of the effects of fire exclusion on some important, keystone ecosystems is also included. Effects of fire exclusion are detailed at the stand and landscape levels. Stand-level effects include increases in woody fuel loading, canopy cover, vertical fuel distribution, canopy stratum, and fuel continuity. Landscape-level effects include increases in landscape homogeneity, fuel contagion, and hydrology. Cross-scale exclusion effects concern increases in fire intensity, severity, and size as fuels increase and become more connected. Insect and disease epidemics are also likely to increase, and streamflows are likely to decrease. Restoration of some semblance of the native fire regimes seems a critical step toward improving the health of many Rocky Mountain ecosystems.

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Cascading Effects of Fire Exclusion in Rocky Mountain Ecosystems: A Literature Review

By

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Introduction

The extensive wildfire season of 1910 was a defining moment for the United States wildland fire management organization (Koch 1942). Although the primeval role of fire had already been altered in some areas of the Rocky Mountains since the mid-1800s by heavy livestock grazing, the General Land Office had established only a primitive fire control structure to suppress fires in the remote Rocky Mountains prior to 1910 (Benedict 1930; Koch 1942). Then, 1.5 million ha burned during that dry windy summer of 1910, and the Forest Service initiated a more aggressive fire suppression policy (Cohen and Miller 1978). The enactment of the Weeks Act in 1911 improved fire suppression coordination by providing funding to those states willing to adopt comprehensive fire suppression plans (Babbitt 1995). By 1929, this emergent fire suppression organization was fully functional with hundreds of fire towers built and thousands of men employed (Agee 1993; Koch 1942). Its effectiveness has accelerated in intensity and technology until present day. Similar advances in fire suppression organizations occurred after 1945 in the Canadian Rocky Mountains (Woodley 1995). However, Canadian National Parks policy was changed in 1979 to allow for natural ecosystem processes such as fire to occur under conditions dictated by park vegetation and fire management plans (Hawkes 1990).

This largely successful suppression program owes much of its success to strong governmental support and extensive advertisement campaigns. Smokey Bear's message was simple, direct, and effective—"Prevent wildfires"—but it was also shortsighted (Pyne 1982). In a perfect world, we should have known that there would be adverse consequences of this pervasive fire exclusion policy. But growth of Rocky Mountain forest and range vegetation and the subsequent accumulation of hazardous fuels are gradual processes, so it was difficult for one generation of forest and range biologists and scientists to observe and agree upon the adverse effects of excluding fire from Rocky Mountain ecosystems. Now we are faced with some critical ecological issues in the aftermath of our war on forest and range fires. The health of many Rocky Mountain ecosystems is now in decline because of fire exclusion. Moreover, fire exclusion has actually made it more difficult to fight fires, and this poses greater risks to the people who fight fires and for those who live in and around Rocky Mountain forests and rangelands. This report discusses the extent of fire exclusion in the Rocky Mountains and details the diverse and cascading effects of suppressing fires in the Rocky Mountain landscape by spatial scale, ecosystem characteristic, and vegetation type. We also describe the effects of fire exclusion on some important, keystone ecosystems.

It is well documented that most Rocky Mountain ecosystems evolved with fire (Arno 1980; Pyne 1982; Quigley and Arbelbide 1997; Swetnam and Baisan 1996). John Muir stated that fire, along with temperature and moisture, is one of the greatest factors that govern forest growth (Pinchot 1899). The importance of fire in maintaining ecosystem health and reducing fuel loads was also identified by Aldo Leopold (1924) in one of his essays on the subject (Flader and Callicott 1991). Even Gifford Pinchot (1899) recognized the critical role of fire in shaping North American forests. He noted, "the most remarkable regulative effects of forest fires relates to the composition of the forest," referring to the "qualities of resistance to fire the trees possess." Gruell (1985b) documents an extensive historical record of fire on the western landscape since the late 16th century. In Weaver's (1943) seminal paper, evidence shows that periodic fires operated to control the density, age, and composition of ponderosa pine stands. He further states that removal of fire would "threaten sound management and protection" of these forests. Benedict (1930) documented an "increasing hazard" over 21 years of fire protection. Yet despite these early warnings, fires continued to be suppressed on the majority of public lands because suppression was the more desirable land management policy (Mutch 1995). It wasn't until the late 1970s and early 1980s that wildland fires were

allowed to return to some National Parks and Wilderness Areas (Kilgore 1985; Kilgore and Heinselman 1990).

The ecological role of fire is to function as an extrinsic disturbance factor (Crutzen and Goldammer 1993). It is a "keystone" disturbance that (1) recycles nutrients, (2) regulates succession by selecting and regenerating plants, (3) maintains diversity, (4) reduces biomass, (5) controls insect and disease populations, (6) triggers and regulates interactions between vegetation and animals, and, most importantly, (7) maintains biological and biogeochemical processes (Agee 1993; Crutzen and Goldammer 1993; Mutch 1994). Fire is neither good nor bad; fire is an important ecological process that can produce variable effects. The value of these effects must be interpreted in the context of human desires and needs. One fact is known: the removal of fire from the fire-dominated ecosystems of the Rocky Mountains has caused a plethora of cascading effects that has permeated nearly every part of this rugged landscape (Allen and others 1998; Arno and Brown 1989; Bogan and others 1999; Ferry and others 1995; Mutch and others 1993). At first glance, the effects of fire exclusion may seem beneficial to society (for example, preservation of timber resources and watershed protection), but on closer scrutiny, there seems little doubt this policy has created many unhealthy features on Rocky Mountain landscapes.

It is important to clarify some terminology used in this chapter. First, "fire suppression" is the act of extinguishing or fighting fires, while "fire exclusion" is the defacto policy of trying to eliminate fires from the landscape using fire suppression techniques. A "fire regime" is a description of the long-term, cumulative fire characteristics of a landscape and is often described by frequency, extent, pattern, severity, and seasonality (Agee 1993; Malanson 1987; Martin and Sapsis 1992). Time periods commonly used in discussing historical changes in Rocky Mountain fire regimes include: (1) "Recent Native American Period," including the four or five centuries prior to permanent settlement by Euro-Americans (about 1850), (2) "Euro-American Settlement Period," which was a time of large uncontrolled resource exploration and utilization (about 1850 to 1920), and (3) "Fire Exclusion Period," during which government agencies, transportation facilities, and fire-control infrastructures have had a major impact on fire regimes. "Native fire regime" describes when fires are allowed to burn across the landscape, and eventually the character of the vegetation will reflect the character of the fires. It is often assumed that historical fire regimes prior to 1850 were native fire regimes. They may or may not have involved significant numbers of Indian-caused fires (Barrett and Arno 1982). "Fire severity" describes the impact of the fire on the biota and is quite different from fire intensity, which is the heat output from a fire. Three fire severity classes are commonly used—nonlethal (low intensity surface fires that do not kill larger individuals), mixed (patchy severity burns that create mosaics of severity), and stand replacement (lethal surface and crown fires that kill over 90 percent of trees) (Morgan and others 1996). "Ecological processes" are those factors that influence the flow of energy in an ecosystem and include transpiration, photosynthesis, and disturbances (Waring and Running 1998). "Keystone" refers to the presence of an important species or process that is crucial in maintaining the organization and diversity of an ecosystem (Mills and others 1993).

A discussion of fire exclusion effects must also include the role of burning by indigenous peoples on the Rocky Mountain landscape. There is substantial evidence that portions of the Rocky Mountain landscape were extensively humanized by the early 16th century (Denevan 1992). John Mullan (1866) recognized that these early inhabitants had a profound bearing on forest structure and composition resulting primarily from fires they set. They started fires for many reasons including land clearing, wildlife habitat improvement, cultivation, defense, signals, and hunting (Bahre 1991; Gruell 1985a; Kay 1995; Lewis 1985). However, there is great debate as to whether fire regimes maintained by Native Americans would have been similar if maintained by lightning fires alone (Barrett and Arno 1982; Fisher and others 1986; Silver 1990), and whether anthropogenic burning is considered part of the native fire regime (Arno 1985; Kilgore 1985). Fires set by Indians are often different from lightning fires in terms of seasonality, frequency, intensity, and ignition patterns (Frost 1998; Kay 1995). For this report, we assume anthropogenic ignitions are part of the Rocky Mountain historical fire regimes and, therefore, reflect the native or natural fire regimes (Arno 1985; Bahre 1991; Russell 1983). Separating anthropological fires from the historical fire record would be an impossible task with highly speculative results.

It is also important to recognize the critical role of livestock grazing on the decline in wildland fire in the Rocky Mountains. Extensive grazing of sheep, cattle, and horses from the early 1850s to the present removed an important layer of fine fuel (in other words, grass and forbs) from the landscape (Covington and Moore 1994). The reduction in grass cover not only removed fuel for fire spread, but it also limited the dry material available for fire ignition. Moreover, the elimination of grass competition allowed rapid conifer encroachment that further reduced grass cover by shading (Hansen and others 1995). Intensive grazing on Rocky Mountain landscapes has certainly exacerbated the impacts of modern fire suppression efforts (Bunting 1994; Gruell 1983; Shinn 1980; Swetnam and Baisan 1996).

Extent of the Problem

Impacts of fire exclusion are different from the effects of other management actions, such as logging, because the impacts occur gradually and are manifest in nearly every portion of the landscape rather than localized to small areas. It is difficult to comprehensively describe and quantify these effects across large regions because exclusion effects are tied to native fire regimes, which are extremely variable in time and space (Agee 1993; Arno 1980; Barrett and Arno 1993; Heinselman 1981; Heyerdahl and others 1995).

Moreover, not all fires can actually be suppressed on the landscape and, when these wildfires occur on fire-excluded landscapes, they are often different in severity and aerial extent from fires that occurred prior to the exclusion era (prior to 1900) (Mutch 1995; Swetnam and Baisan 1996). And last, there have been some major land use practices, such as agricultural development and urbanization, that have completely altered ecosystems so an historical comparison of fire effects would not be meaningful (Morgan and others 1998). As a result, there have been few regional assessments of fire exclusion effects for the Rockies, especially for both public and private lands (Ferry and others 1995).

It appears that only a small fraction of the pre-1900 annual average fire acreage is being burned today. Barrett and others (1997) estimated an average of 2.4 million ha burned annually in the interior northwestern United States prior to 1900. Even the biggest wildfire years of the 21st century burned less than half of this historical average. Approximately two-thirds of this annual historical burned area occurred in sagebrush and grassland vegetation that have mostly been converted to agriculture or dry pasture for livestock (Morgan and others 1998). Gruell (1985b) estimated from early journalist accounts of fire throughout the Rocky Mountain region that modern fires burn less than one-fourth of the land that burned historically. Leenhouts (1998) performed a comprehensive assessment of burning in the contiguous United States and estimated that approximately 3 to 6 times more area must be burned to restore historical fire regimes, thereby consuming 4 to 8 times more biomass and producing 6 to 9 times more emissions than present. Smoke emission production from prehistoric wildland fires in British Columbia was estimated to be 3 to 6 times larger than the average annual contemporary production because of the vast area burned prior to the 1900s (Taylor and Sherman 1996). A mapping of presettlement fire regimes for the United States revealed more than half the country experienced fires at intervals between 1 and 12 years (Frost 1998). Kilgore and Heinselman (1990) classified historical continental fire regimes and found the greatest detrimental impacts of fire exclusion were in frequently occurring (less than 25 years), low-intensity fire regimes (for example, grasslands and ponderosa pine) that encompass a large part of the Rocky Mountains.

A comparison of current and historical fire regimes for the Interior Columbia River Basin (ICRB) using comprehensive digital maps developed by Morgan and others (1996, 1998) provides another spatial assessment of the extent of fire exclusion. Generally, they found that recent fires tended to be less frequent and more severe than those that occurred prior to 1900. The greatest change in fire regimes were in the shrublands, grasslands, dry forests, and woodlands, which accounted for over 40 percent of ICRB area (88 million ha). Fire regime changes were especially dramatic in areas converted to agriculture and pastures. As expected, short fire return interval ecosystems were most affected by fire suppression. For example, ponderosa pine cover types decreased by 23 percent, while Douglas-fir cover types increased by 40 percent in aerial extent.

Not all ecosystems or all Rocky Mountain landscapes have experienced the impacts of fire exclusion as yet. In some wilderness areas, where in recent decades natural fires have been allowed to burn, there have not been major shifts in vegetation composition and structure (Brown and others 1994). In some alpine and desert-scrub ecosystems, fire was never an important ecological factor. In some upper subalpine ecosystems, fires were important, but their rate of occurrence was too low to have been significantly altered by the relatively short period of fire suppression. For example, the last 70 to 80 years of fire suppression have not had much influence on subalpine landscapes with fire intervals of 200 to several hundred years (Romme and Despain; Veblen and others 1994; White 1985, 1989). White (1985) mentions fire suppression was not effective enough to reduce subalpine burned area in Bannff National Park in Canada, but Rogeau (1996) found recent shifts in forest stand ages to older age classes. Consequently, it is unlikely that fire exclusion has yet to significantly alter stand conditions and forest health in these subalpine ecosystems. Yet, Barrett and others (1991) recognized that fire exclusion effects in long fire interval fire regimes, such as those in lodgepole pine and spruce fir, are not yet manifest at the stand level, but are detectable at the landscape level. For example, they mentioned young age classes are often missing from subalpine landscapes where fires have been excluded. Thus, the well substantiated relationship of reduced forest health due to fire exclusion in ecosystems characterized by high fire return intervals (for example, low-elevation ponderosa pine woodlands) **cannot be applied to all** mesic subalpine ecosystems with long fire return intervals. But despite these exceptions, the Rocky Mountain landscape, taken as a whole, is not burning at the pre-1900 rate (Covington and others 1994; Frost 1998; Mutch 1995).

Stand-Level Effects

Stand Composition and Structure

Perhaps the most documented and studied effect of fire exclusion is the change in stand composition and structure. In general**, forest composition has gone from early seral, shade-intolerant tree species to late seral, shade-tolerant species, while stand structure has gone from single-layer canopies to multiple-layer canopies with fire exclusion** (Mutch and others 1993; Quigley and Arbelbilde 1997; Steele 1994; Veblen and Lorenz 1991). This phenomenon is repeated over and over again for most fire-dominated ecosystems of the United States and Canada (Chang 1996; Ferry and others 1995; Frost 1998; Kolb and others 1998; Quigley and Arbelbide 1997; Taylor 1998). It is well illustrated by comparing historical and current photographs for identical landscapes, such as those shown in figure 1, as compiled from a wide variety of vegetation change studies (Gruell 1980, 1983). **It is this fundamental**

Figure 1—Historical and current photographs illustrating the changes in vegetation structure and composition due to fire exclusion (figures 1a through 1h; photos and text from Gruell 1993).

Figure 1(A)—1909. Fire Group 4: warm-dry Douglas-fir. Elevation 4,400 ft (1,341 m). A northwesterly view showing cleanup operations on the Lick Creek timber sale, Bitterroot National Forest, near Como Lake. The number of stumps and slash piles suggests that this was an open ponderosa pine stand, a condition typical of the bitterroot Valley where stands had been subjected to frequent ground fires. Fire scar samples showed a mean fire interval of 7 years between 1600 and 1900. The understory appears to have a high incidence of lupine, but few shrubs are evident. Forest Service "lumberman" C. H. Gregory stands in foreground (USDA Forest Service photograph 86476 by W. J. Lubken).

Figure 1(B)—September 1979—70 years later. Camera point replicated original position. Soil disturbance during logging and exclusion of wildfire allowed ponderosa pine and Douglas-fir seedlings to become established and develop into a dense understory. The large ponderosa pine in center foreground in the 1909 view, as well as others trees, were cut during shelterwood and selection harvests in 1952 and 1962 (photograph by W. J. Reich).

Figure 1C—1871. Fire Group 5: cool-dry Douglas-fir. Elevation 6,200 ft (1,890 m). High on the slopes above the Yellowstone River 10 miles southeast of Livingston, the view is east toward the head of Suce Creek. Snags and age of young limber pine on near slope indicate locality was swept by wildfire several decades earlier. Based on present plant composition, ground cover was apparently dominated by Idaho fescue and bluebunch wheatgrass. Fire mosaics are evident in the distance (W. H. Jackson photograph 57-HS-1213, courtesy of National Archives).

Figure 1(D)—July 27, 1981—110 years later. A dense stand of Douglas-fir, limber pine, and Rocky Mountain juniper now occupies near slope shown in original view. The grass cover on this slope has been largely eliminated by tree canopy closure. Camera was moved approximately 100 yards down slope to allow unobstructed view of distant slopes. Tree cover on distant slopes has increased dramatically (photograph by R. F. Wall).

Figure 1(E)—1909. Fire Group 6: moist Douglas-fir. Elevation 6,100 ft (1,860 m). Looking north-northwest up Blake Creek at a point 1 mile above forest boundary on south side of Big Snowy Mountains, Lewis and Clark National Forest. Scene shows effects of wildfire in the late 1800s that burned both sides of drainage. Scattered ponderosa pine and Douglas-fir occupy near slope and canyon bottom. Herbs and shrubs comprise early successional vegetation in burned areas. On right, fire created a mosaic of burned and unburned timber. Note rock outcrop in burned stand (U.S. Geological Survey photograph 114 by W. R. Calvert).

Figure 1(F)—August 20, 1980—71 years later. Camera was moved left of original point to avoid trees that screened early view. Regeneration of Douglas-fir has resulted in a landscape dominated by conifers. The rock outcrop visible in 1909 is now almost totally obscured by tee growth. Conifer competition has a largely eliminated early successional understory species (photograph by W. J. Reich).

Figure 1(G)—1900. Fire Group 6: moist Douglas-fir. Elevation 5,300 ft (1,616 m). From the ridge about 5 miles west of Haystack Butte, the view is southwest across Smith Creek toward Crown Mountain on east front of Rocky Mountains, Lewis and Clark National Forest. Near slopes are in early succession following wildfire in latter 1900s that removed conifers and stimulated production of aspen, willow, chokecherry, mountain maple, and other deciduous vegetation. Stumps resulting from timber cutting and snags indicate that the pre-1900 conifer stands were less dense than current stands (U.S. Geological Survey photograph 665 by C. D. Walcott).

Figure 1(H)—September 16, 1981—81 years later. Slopes below camera point and adjacent terrain as well as near slope are now densely covered by Douglas-fir. View was obtained by cutting screening fir and climbing one of the larger Douglas-fir about 50 yards from original camera position at top of ridge. Canopy closure has resulted in a decline in condition of deciduous species (photograph by G. E. Gruell).

Table 1—Summary of the documented effects of fire exclusion by organizational level and ecosystem characteristic. References for each effect are detailed in the chapter.

Scale	Ecosystem attribute	Fire exclusion effect
Stand	Composition	Increased number of shade-tolerant species, decreased number of fire-tolerant species, decreased forage quality, decreased plant vigor, and decreased biodiversity in plant and animals.
	Structure	Increased vertical stand structure, multistoried canopies, increased canopy closure, increased vertical fuel ladders and continuity, greater biomass, higher surface fuel loads, and greater duff and litter depths.
	Ecosystem processes	Slowed nutrient cycling, greater fire intensities and severities, increased chance of crown fires, increased insect and disease epidemics, short-term increase in stand productivity, decrease in individual plant vigor, and decreased decomposition. Increased leaf area; increased evapotranspiration, rainfall interception, autotrophic and heterotrophic respiration; increased snow ablation.
	Soil dynamics	Decreased nutrient (N,P,S) availability; increased pore space, water-holding capacity; lower soil temperatures; increased hydrophobic soils; and increased seasonal drought.
	Wildlife	Increased hiding and thermal cover, increased coarse woody debris, lower forage quality and quantity, increased insect and disease, and decreased biodiversity.
	Resources	Decrease in aesthetics, increased timber production, decreased visitation, increased risk to human life and property, increased fire fighting efforts, and improved air quality.
Landscape	Composition	Decrease in early seral communities, increased landscape homogeneity, increase in dominance of one patch type, and decreased patch diversity.
	Structure	Increase in patch evenness, patch size, patch dominance, and contagion.
	Disturbance	Larger and more severe fires, increase in crown fires, increased insect and disease epidemics, and increased contagion resulting in more severe insect and disease epidemics.
	Carbon and water cycles	Increased water use, increase in drought, lower streamflows, higher emissions of carbon dioxide from respiration, increased water quality, and decreased stream sediment.
	Resources	Decreased visitation, visual quality, and viewing distance.

change in vegetation composition and structure that cascades downward to impact a myriad of other ecosystem characteristics. Table 1 summarizes the effects of fire exclusion at the stand and landscape levels described in detail in the following sections.

The compositional and structural shifts are consistent with the current successional theory that characterizes how vegetation will change without disturbance (Arno and others 1985; Drury and Nisbet 1973; Horn 1974). Plant species adapted to the early stages of succession, such as those that best survive or regenerate after a fire, are replaced by species that are better able to compete for growing resources in the absence of fire (Bazzaz 1979; Noble and Slatyer 1980). Species adapted to the latter stages of succession tend to possess ecophysiological and morphological characteristics that dramatically alter the dynamics of stands and landscapes as succession progresses (Bazzaz 1979, 1990; Drury and Nisbet 1973; Grime 1966, 1974; Noble and Slatyer 1980; Wallace 1991). For instance, early seral species tend to have high photosynthetic rates, low tolerance for shade, rapid height and diameter growth, frequent cone crops, long lifespans, and short crown lengths, while most late seral species generally have the opposite characteristics (Bazzaz 1990; Grime 1979; Horn 1974; Van Hulst 1978). Since late seral species are shade tolerant and able to photosynthesize under low light conditions, forests composed of late seral species commonly have higher plant densities (plants per unit area) with many individuals of different size classes (Oliver and Larson 1990). Especially abundant are the young shade-tolerant individuals, which bide their time in the understory waiting for a gap to open in the overstory canopy. Shade-tolerant trees also tend to have denser crowns that

often extend to the ground (Brown 1978). These thick crowns coupled with high densities and many size classes create multilayered stand structures with sparse undergrowth (Frost 1998) (fig. 1b).

The increase in **density, biomass, and number of woody species** also seems to be a common theme in most fire-altered ecosystems (fig. 1). Invasion of shrubs and trees into grasslands and shrublands due to the lack of fire and heavy grazing is evident in many areas of the Rocky Mountains (fig. 1c,d). In southern Canada, Taylor (1998) reported that fire suppression has become increasingly effective during the last 40 years, and ecosystems that historically experienced surface fire regimes are now experiencing a reduction of grassland and open forests and an increase in shade-tolerant species and dense forests (Taylor and others 1998). Deep-rooted shrubs increased dramatically in shrub-steppe ecosystems (Link and others 1990). Inland Douglas-fir invasion into western grasslands of Montana has been well documented (Arno and Gruell 1986; Hansen and others 1995), as has ponderosa pine into the Colorado Front Range grasslands (Veblen and Lorenz 1991). Limber pine is encroaching prairie grasslands in many parts of its range due to the removal of fire (Gruell 1983). Extensive conifer invasion into ancient montane grasslands is occurring in the Southern Rocky Mountains (Allen and others 1998; Bogan and others 1999). Lodgepole pine invades sagebrush communities when fire is removed from Yellowstone National Park uplands (Patten 1969). Ogle and DuMond (1997) documented the increase in woody material due to increases in tree density for many parts of the Intermountain region across many forest types and fire regimes. Habeck (1994) and Arno and others (1995) documented a three- to fivefold increase in density of shade-tolerant conifers in ponderosa pine forests (fig. 1a,b). Covington and Moore (1994) recorded a tenfold increase in ponderosa pine density since European settlement.

Biodiversity

Diversity of plants, animals, and ecological processes are enhanced by fire in many ecosystems. Martin and Sapsis (1992) mentioned that the control of fire reduces biodiversity, and it is the variability of fire regimes in time and space that creates the most diverse complexes of species. Higgins and others (1991) detailed the adverse effects of fire exclusion on grass diversity, quality, and vigor in rangelands. Vogl (1979) related that the healthiest and most diverse grasslands are composed of a complex mosaic of burning histories. Landscapes having fires with high variability in timing, intensity, pattern, and frequency tend to have the greatest diversity in ecosystem components (Brown and others 1994; Romme 1982; Sieg 1997; Swanson and others 1990). Long-term health of below-ground fauna depends on the variability of fires, whereas the diversity of soil organisms can be reduced by the severe fires that can occur in stands where fire has been removed for long periods (Borchers and Perry 1990). Biondini and others (1989) documented the complex effect of the season and frequency of fire on increased herbaceous diversity in composition and structure in grasslands. In Canada's Banff National Park, Achuff and others (1996) modeled future vegetation succession for 95 years without disturbance and found an overall loss of biodiversity caused by a loss of 19 of 29 vegetation types.

Successional floristics in most Rocky Mountain ecosystems are best described by the Initial Floristics Model of Egler (1954), which states that the majority of plant species present before the fire will be there after it burns the stand (Anderson and Romme 1991; Veblen and Lorenz 1986). However, plant diversity tends to decrease with advancing succession because there are higher numbers of species adapted to colonize postfire settings from highly dispersed or dormant propagules (Gill and Bradstock 1995; Stickney 1990). In addition, the density, cover, height, and vigor of undergrowth species tend to decrease as the overstory becomes dense and tree leaf area increases because of dominance by shade-tolerant species (fig. 1a,b) (Gruell 1986; Stickney 1985). Therefore, undergrowth vascular plant species richness and density tend to be low in the late successional communities commonly found on fire-excluded landscapes.

Many rare and threatened species have declined with the reduction of fire (Greenlee 1997). Hessl and Spackman (1995) found that 135 of the 146 threatened, endangered, and rare plant species in the United States benefit from wildland fire or are found in fire-adapted ecosystems. Although local species extinction can occur with fires that occur too frequently or fires that burn too severely, it is generally accepted that the locally rare plants have greater chances of thriving on those landscapes that have diverse vegetation communities and structures created by diverse disturbance histories and fire regimes (Gill and Bradstock 1995). Sheppard and Farnsworth (1997) point out that while historical fires were beneficial ecological agents of change, today's fire can be a destructive force that endangers rare plant species because these fires have a greater probability of being large, severe, and stand replacing (see later sections).

Crown and Surface Fuels

One important stand characteristic that changes with advancing succession is the increase in amount of dead and live biomass, which the fire community calls "fuels" (Peet 1992). Fuel loadings (mass per unit area) generally increase in the absence of fire because of a myriad of ecological factors (fig. 1a,b) (Brown 1985). First and most important, long fire return intervals mean live fuels have longer times to grow and dead fuels have longer periods to accumulate on the ground. Next, crown fuels (aboveground foliar biomass) increase because late seral, shade-tolerant species tend to have more biomass in the forest canopy due to their high leaf areas, and the biomass tends to be well distributed over the height of the trees (Brown 1978; Minore 1979; Waring

and Running 1998). Stand leaf area generally increases over successional time because shade-tolerant species generally have longer needle retention times, higher leaf area:sapwood ratios, and more leaf mass in the crown (Brown 1978; Callaway and others 1998; Keane and others 1996; Peterson and others 1989; White and others 1990). Higher leaf areas usually require additional conducting tissue for support, which means the tree may need to produce more branch and twig wood along greater portions of its stem (Landsberg and Gower 1997). And because late seral species are shade tolerant, there are many smaller seedlings and saplings present in the understory to take advantage of any gaps in the canopy. So, the greater crown leaf and branch biomass distributed along greater parts of the stem, coupled with high seedling and sapling densities, can create "ladder" fuels that allow flames from surface fires to climb into the forest canopy and result in "crown" fires.

Surface fuel loadings increase as fire is eliminated because the greater crown biomass ultimately results in increased leaf and woody material accumulating on the forest floor fuel because the recycling process of fire is absent (Brown and Bevins 1986; Covington and Moore 1994; Fulé and others 1997). Dense crowns also reduce solar radiation attenuated to the forest floor which may lower soil temperatures resulting in decreased decomposition rates and still higher branch and litter accumulations (Borchers and Perry 1990; Brown and Bevins 1986). Duff and litter depths generally increase proportionate to the crown closure and leaf area because of the additional needlefall and reduced decomposition (DeBano 1991). The rate and maximum amount of fuel buildup depend on many biophysical factors that control decomposition, including site productivity, rainfall, and climate (Olsen 1981). Habeck (1985) found large surface fuel accumulations on those moist sites with long fire return intervals such as subalpine fir and western red cedar forests. Van Wagtendonk (1985) simulated a decrease in fine fuels but a large increase in large woody fuels when fires are suppressed in short fire interval ecosystems. However, Keane and others (1990, 1997) simulated two- to threefold increases in live and dead biomass as fires were removed from several ecosystems in the Northern Rockies. Thick litter and duff layers with high woody fuel loadings are commonly found in fire-excluded stands of ponderosa pine (Covington and Moore 1994; Habeck 1994; Keane and others 1990; Steele and others 1986), lodgepole (Arno and others 1993; Brown 1973), whitebark pine (Keane and others 1994), and aspen (DeByle 1985).

Soils

Soil properties change as fires are reduced and succession advances in an ecosystem. Organic matter generally increases with decreased fire frequency, and this improves pore space, water-holding capacity, and aggregation. However, when soils with thick organic horizons are burned, some of the volatilized organic matter moves downward along a steep temperature gradient and condenses to form a water repellent layer that impedes infiltration and can cause massive erosion (DeBano 1991). Endo- and ectomycorrhizae are particularly sensitive to soil heating by fire because they are concentrated in the organic and upper mineral soil layers (Borchers and Perry 1990; Hungerford and others 1991). While historical fires often killed some of these microorganisms, severe fires resulting from the abnormally high fuel loadings after fire exclusion can severely reduce their populations (DeBano 1991). Consumption of the thick soil organic matter accumulations (in other words, duff and litter) from fire exclusion will result in deep soil heating that can kill more plant propagules and microorganisms (Hungerford and others 1991). Most fires generally increase soil pH due to ash accretion, which directly affects availability of many nutrients (Higgins and others 1991). Temperatures of burned soils rise earlier in the season, which may stimulate some decomposing bacteria and early season grasses and forbs (Higgins and others 1991). Upper subalpine soils are often classified as Cryochrepts or Cryoboralfs during whitebark pine dominance, but convert to Cryoborolls under subalpine fir canopies (Hansen-Bristow and others 1990).

Fire exclusion effects on nutrient cycling are more complex and confounding (Grier 1975; Klopatek and others 1990). While there is abundant nitrogen in the large amounts of organic matter accumulated as a result of fire exclusion, only a small portion of this nitrogen is made available to plants each year from decomposition by soil organisms (Waring and Running 1998). Fire's combustion process releases some of the nitrogen sequestered in the fuels and makes it available as ammonium-N to the plants as it condenses on lower soil layers, even though a large portion of nitrogen is volatilized and lost to the atmosphere depending on amount of combusted organic matter (Grier 1975; Little and Ohmann 1988; Pehl and others 1986; Schoch and Binkley 1986). Nitrogen fertilization is also increased by the actions of decomposing bacteria that are stimulated with fire (Sharrow and Wright 1977). White (1994, 1996) found that ground fires consumed volatile organic compounds that could inhibit decomposition. Klopatek and others (1990) found significant losses of N from the forest floor of a pinyon-juniper woodland after fire, while White (1996) found nitrogen mineralization and nitrification rates were higher in recently burned stands, and Ryan and Covington (1986) found 80 times more ammonium-nitrogen in burned ponderosa pine stands. Only 60 percent of phosphorous is lost during fuel consumption, so substantial amounts of highly available P are found in ash and at soil surface after fires. Sulphur availability is also increased by fires (DeBano 1991). Higgins and others (1991) documented several cases where grassland production and yield increased after burning in the Great Plains due to recycled nutrients. White and others (1990) found high nitrogen and magnesium levels in whitebark pine litter, while subalpine fir and Engelmann spruce litter had high calcium and high lignin contents. In summary, it seems intact fire regimes ensure continued nutrient cycling and soil health in the drought-frequent Rocky Mountains.

Carbon and Water Cycles

Some interesting and complex changes in major ecophysiological processes result as species and structure changes occur during the prolonged successional cycle resulting from the absence of fire (table 1). Increased leaf area at the species and stand level triggers complex physiological responses in the ecosystem dynamics of the stand. Transpiration, snow ablation, and canopy interception generally increase with higher leaf areas, and this can result in periodic seasonal depletion of soil water, increased canopy evaporation, and decreased streamflow (Hann 1990; Kaufmann and others 1987; Skidmore and others 1994; Troendle and Kaufmann 1987; Waring and Running 1998). Increased water use often results in seasonal droughts that reduce water availability to individual trees (Kolb and others 1998). High leaf areas also diminish radiation attenuated to the forest floor, which when coupled with lower soil moistures, can slow decomposition rates, limit snow accumulation, and delay soil thaw (Bazzaz 1979; Kaufmann and others 1987; White and others 1990). Reduced decomposition can then result in delayed nutrient cycling, and most often, high woody fuel and duff accumulations, which when burned, can cause severe wildfires with deep soil heating and high plant mortality (Kolb and others 1998). Hungerford and others (1991) identified the role of fire as the primary control of decomposition and nutrient cycling in fire-prone ecosystems.

Fauna

The high canopy cover and multistoried stand structure found in late stages of succession certainly improves big game thermal and security cover (Gruell 1980). However, the dense canopies also shade out early seral shrubs and grasses that usually have high forage value for many ungulates. Production of palatable shrub forage in old, fire-excluded stands may be less than 1 percent of that found in young postfire communities. Moreover, ungulates may find dense late seral stands difficult to traverse because of the abundance of downed logs and thick understory (fig. 1b, f) (Gruell 1979; Lonner and Pac 1990). In Canada, prime wood bison habitat consists of early successional mesic prairie that depends on frequent fire to prevent conifer encroachment and organic matter buildup (Gates and others 1998). In contrast, Gruell (1986) hypothesized that mule deer irruptions from 1930 to 1970 were primarily a result of rangeland succession from grasses to shrubs due to reduction in fire size and frequency. Bighorn sheep can benefit from fire by reduced lungworm infections, improved forage, and reduced tree cover (Peek and others 1985). Freedman and Habeck (1985) noted that fire exclusion reduced winter range and forage quantity and quality eventually reducing deer populations in Montana. Reduction of aspen forests from the absence of fire has significantly affected the diets of ungulates that highly prize this valuable forage species (DeByle 1985). The decline of whitebark pine from fire exclusion and blister rust in the Northern Rockies can adversely affect summer range for elk and deer (Lonner and Pac 1990). Carrying capacity for elk can be diminished by removing fire from the ecosystem due to reduction in quality browse plant species (Gruell 1979). Drew and others (1985) noticed a significant reduction in winter ticks, parasites on large mammals, with spring prairie burning in Alberta.

The plant species that define many fire-adapted ecosystems (for example, aspen, ponderosa pine, whitebark pine) are often keystone species that are critical for the survival of many other animals in that ecosystem (deMaynadier and Hunter 1997). For instance, Hutchins (1994) has chronicled over 110 animals that consume whitebark pine seeds in high-elevation ecosystems. DeByle (1985) documented over 134 birds and 55 species of mammals that regularly utilized aspen forests. Kendall and Arno (1990) mention that whitebark pine dominance benefits many wildlife species because the typically open canopy promotes undergrowth forage quality and production (for example, increased berry production). Higgins and others (1991) detailed the significantly higher number of insects, birds, waterfowl, and small mammals on prairie landscapes with healthy fire regimes.

Perhaps the largest impacts of fire exclusion may be felt by nongame wildlife species. Hutto (1995) noted the importance of fire, especially stand-replacement fire, for creating habitat for many Rocky Mountain bird species. Around 15 species were solely associated with postburn communities, and over 87 species were found in burned stands. Hejl (1992) identified the importance of fire-dominated heterogeneous landscapes to bird diversity. Landscapes with intact fire regimes have high variability in patch size, shape, and type, which is extremely beneficial for the existence of many avian species. This can also be said for many insect and rodent species (Higgins and others 1991). Finch and others (1997) mention that fire exclusion in Southwestern forests tends to favor generalist bird species that can utilize all stages of succession rather than specialist bird species found primarily on heterogeneous landscapes, open forests, burns, snags, or a combination of all. Small mammal populations may increase with the number of down logs as fuels accumulate during succession, but many mice, shrews, and gophers are found mostly in those early seral communities that directly follow fire. Moreover, the diverse mosaic of stand structures and composition created by an intact fire regime greatly correlate with higher numbers of small mammal individuals and species (Ream and Gruell 1980). Covington and others (1994) noted that the increased erosion from large-scale fires on landscapes with altered fire regimes degrades riparian habitat and adversely impacts aquatic organisms because of high sediment loadings.

Exotics

The introduction of exotics into Rocky Mountain ecosystems has complicated and, in some cases, intensified fire exclusion effects. Some exotic plant species tend to efficiently colonize following disturbances so restoration of fire

regimes may increase exotic dominance and reduce diversity (Covington and others 1994). However, landscapes will burn regardless of fire control measures, and the severely burned areas created by wildfires on fire-excluded landscapes might accelerate exotic invasions. Low-severity burns may tend to favor native plants adapted to survive fire. The invasion of annual grass exotics such as cheatgrass (*Bromus tectorum*) into sagebrush-steppe vegetation types has actually increased fire frequency because of the presence of abundant fine fuels (in other words, grasses) resulting in the elimination of sagebrush and a permanent conversion to annual grasslands (Whisenant 1990). The use of fire for control of exotic weed species has had mixed results.

Some exotic diseases and pests have accelerated the successional cycle resulting in mid-seral stands having compositions and structures similar to old-growth stands. For example, white pine blister rust (*Cronartium ribicola*) has killed many mature whitebark pine in northern Montana and Idaho rapidly converting stands to subalpine fir (Keane and others 1994; Kendall and Arno 1990). Blister rust has also speeded succession in western white pine forests to grand fir, western red cedar, and western hemlock. This has severely altered the fire regime and successional process, making restoration difficult but not impossible.

Cultural and Natural Resources

People have changed the way they used fire-dominated ecosystems as fire was removed from the landscape. Gruell (1990) noted that the absence of fire has had profound implications on natural resource management. Livestock forage resources in the West have been depleted in some areas because of conifer shading (Gruell 1979). Livestock and big game carrying capacities have decreased because of conifer and tall shrub encroachment. And, although fire is often blamed for destroying visual quality, it can enhance long-term visual quality and viewing opportunities by reducing tree densities. The dense tree growth found in forests without fire restricts viewing and detracts from the outdoor experience. Open-grown, parklike stands of ponderosa pine created by frequent surface fires have a high aesthetic quality and are preferred by today's outdoor enthusiasts (Warskow 1978). A decrease in recreation activities and visitation can occur as tree cover and density increase with fire exclusion.

Fire exclusion will heighten fire hazards to forest homes as people continue to develop and settle lands along the urban-wildland interface (Fischer and Arno 1988). The loss of homes and human life can escalate as the surrounding forest advances in succession because of the buildup of canopy and surface fuels (Freedman and Fischer 1980). Moreover, multilayered canopies and dense crowns will increase the chance of crown fires that are difficult to control, especially in an urban setting (Alexander 1988). This could increase the risk of harm to the people who own the property and the firefighters who try to protect it. This is reflected in figure 2 where the amount of area burned in the Western United States has actually increased even though we are currently using better fire suppression technology and are spending more money to fight fires.

The amount and availability of wood biomass will definitely increase with the continued ingrowth of woody material resulting from fire exclusion. Accretion in Western United States conifer forests was estimated at 1.7 times the historical average (Covington and others 1994). Keane and others (1990) simulated a tripling of stand basal area as fire was excluded in ponderosa pine-Douglas-fir stands. Parker (1988) found nearly double the basal area in stands where fire was excluded as species diameter-class distributions went from even aged to uneven aged in mountain hemlock and red fir stands. Most literature sources contend that productivity will tend to decline after a stand reaches maturity as old-growth stands photosynthesize only enough carbon to meet respirative demands (O'Laughlin 1995). However, this assumes the same tree species is present throughout stand development. Callaway and others (1998) found upper subalpine sites dominated by seral whitebark pine reached peak productivities at around 150 years. But, productivity decreased only slightly as subalpine fir trees replaced whitebark pine in the stand because the fir's thin sapwood layer reduced stemwood respiration and allowed the stand to maintain high productivities into the latter stages of succession. Nevertheless, tree vigor will tend to decrease as stands of shade-tolerant and shade-intolerant species become dense and stagnated as resources become limiting (O'Laughlin 1998). Moreover, thinning by periodic fire tended to concentrate productivity in fewer, but larger individuals, thereby resulting in larger stems for a wider variety of wood products and a higher percent utilization (Harvey and others 1989; O'Laughlin 1995).

Air quality has improved, somewhat, at the expense of ecosystem sustainability with fire exclusion policies (Brown and Bradshaw 1994; Covington and others 1994). Less smoke from summer fires lowers atmospheric particulate levels, improves visibility, and decreases natural pollution, but this may only be a short-term advantage. However,

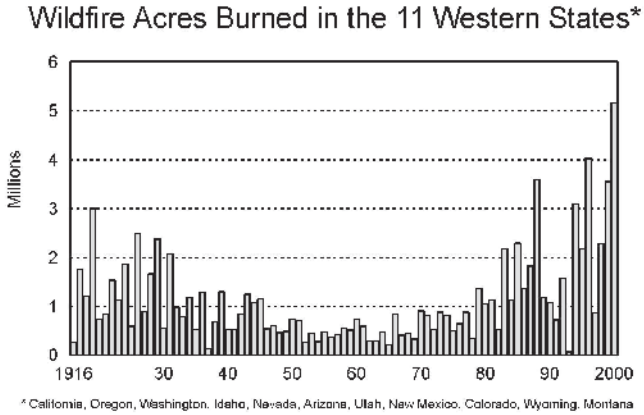

Figure 2—Annual area burned is increasing despite recent advances and increases in fire suppression technology and resources.

Brown and Bradshaw (1994) found similar smoke pollution in the era of total fire exclusion (1930 to 1970) to that generated during the prescribed natural fire period (1971 to 1992) for the Selway-Bitterroot Wilderness Area. Mutch (1994) and Leenhouts (1998) noted that smoke produced from uncontrolled wildfires occurring on fire-excluded landscapes may greatly exceed historical levels, and these high smoke emissions may be more harmful to people than smoke released from a prescribed burning program. The worst smoke pollution in recent times was from unusually severe fires burning on Columbia River Basin fire-excluded landscapes that historically experienced lower severity fire regimes (Ottmar and others 1996). Moreover, less smoke does not mean less carbon dioxide is released into the atmosphere. Computer simulations by Keane and others (1997) showed an increase in atmospheric CO_2 inputs from fire-excluded landscapes due to the increased autotrophic and heterotrophic respiration with advancing succession. At a continental scale, the cumulative effects of this high atmospheric CO_2 input from respiration could have profound effects in global climate warming.

Landscape-Level Effects

Composition and Structure

Historical fire regimes created shifting mosaics of patches, processes, and habitats on Rocky Mountain landscapes (Agee 1993; Romme 1982; Swanson and others 1990). These landscapes tend to become more homogeneous as fire is removed because succession will eventually advance all stands to similar communities dominated by shade-tolerant species (fig. 1e,f,g,h) (Keane and others 1996, 1997; Marsden 1983; Turner and others 1994). Even though late seral species may differ across a landscape depending on site, the multilayer structures of these late seral stands are nearly identical across most biophysical settings (Oliver and Larson 1990). Habeck (1970) noted that fire control on Glacier National Park landscapes resulted in shifts of young and intermediate-aged forests to older forests where communities less than 10 years old were rare. Arno and others (1993) and Hartwell (1997) measured declines of whitebark pine and young lodgepole pine stands and increases in subalpine fir after 91 years on a fire-excluded Northern Rocky Mountain subalpine landscape. Rogeau (1996) reported shifts to older age classes in Banff National Park. McKenzie and others (1996) constructed transition pathways from the current landscape with fire exclusion to a future landscape with increased fire intensity for the conterminous United States and found that increased fire activity could actually decrease broad-scale landscape diversity by increasing the extent of grassland and shrubland types.

Landscape structure (spatial distribution of patches) also changes with fire exclusion as landscapes generally become less fragmented, have lower patch density, and evolve decreased patch diversity, which often results in more contagion, corridors, and large patches (Hann 1990; Hessburg and others 1999; Keane and others 1998; Li and others 1996) (fig. 1g,h). Romme (1982) found that fire control policies tended to reduce landscape richness and patchiness, and to increase evenness and dominance in Yellowstone National Park, but there were situations where the exclusion of fire actually increased landscape diversity. Murray (1996) found that the lack of fire created high-elevation landscapes with low diversity, high mean patch size, and high fractal dimension indices. This creates an interesting situation because larger patches and high homogeneity tend to foster more continuous crown and surface fuels, which can then burn in large fires that create still larger patches and so on in this downward "fire-exclusion" spiral. Baker (1992) found lower fractal dimension, mean shape, Shannon diversity, and patch richness on simulated landscapes with suppression of all fires as compared to landscapes with intact historical fire regimes.

The effect of fire exclusion on patch dynamics depends on the biophysical complexity of the landscape. Often, terrain and landforms, rather than disturbance history, will be the primary factor determining patch dynamics in heavily dissected landscapes (Kushla and Ripple 1997; Li and others1996; Swanson and others 1990). This is because complex landforms create unique and diverse environments that partially control the structure and composition of the potential and existing vegetation (Daubenmire 1966; Pfister and others 1977). But more importantly, fire behavior and growth are heavily influenced by slope, elevation, and aspect, so fire patterns tend to follow topographic landforms (Rothermel 1972, 1991). As a result, it is difficult to generalize about changes in landscape structure with fire exclusion without knowing the degree of topographic complexity.

Hydrology

Landscape hydrologic cycles can be altered as late seral communities progressively dominate landscapes without fires (Covington and others 1994). Higher leaf areas from increased woody biomass will increase evapotranspiration and interception (table 1), resulting in lower streamflows and the drying of springs (Gruell 1979; Romme 1982; Troendle and Kaufmann 1987). This would limit the amount of water available for irrigation and community water supplies, especially during the late summer and early autumn.

Fire exclusion generally reduces soil water and overland flow thereby reducing surface erosion, mass movement, and sediment yield, depending on geomorphic landforms (Swanson 1981). Warskow (1978) demonstrated the critical role of fire in regulating surface and ground-water production in southwestern ponderosa pine forests. Link and others (1990) found range fires removed deep-rooted, woody shrub species from a shrub-steppe ecosystems thereby eliminating the ability of the vegetation to access deeply stored soil water, making more available for human use. Pielke and others (1997) simulated accelerated thunderstorm development and turbulence with increasing landscape heterogeneity indicating vastly different rainfall patterns on

fire-dominated landscapes (Pielke and Avissar 1990). Clark (1989) measured more positive water balances on landscapes with fire.

Fire-excluded landscapes are especially vulnerable to adverse changes in the hydrology when stand-replacement wildfires inevitably occur. Severe fires that burn in heavily grazed forested stands that are outside historical fire frequencies may cause excessive erosion that degrades water quality and aquatic habitat (Covington and others 1994; Tiedemann and others 1979). Snowmelt may be faster from the larger patches created by modern wildfires resulting in earlier and higher spring runoffs. Peak flows usually increase severalfold after large, intense wildfires (Dennis 1989; Tiedemann and others 1979), which would presumably increase surface and mass erosion (Covington and others 1994). The increased vegetation cover near streams on fire-excluded landscapes would probably decrease stream water temperatures, increase long-term inputs of coarse woody debris to streams, and delay and reduce peak runoffs (Dennis 1989). However, when wildfires eventually occur on these protected watercourses, their high severity can reduce shading, increase erosion, and increase water temperatures by 3 to 10 °C depending on streamflow (Amaranthus and others 1989).

Cross-Scale Disturbance Effects

Fire

Perhaps the most important ecosystem process altered by fire exclusion is the native fire regime (Arno and Brown 1991; Covington and others 1994; Morgan and others 1998; Mutch and others 1993). Fires generally become less frequent and more severe with active suppression on the landscape (Arno and others 1993; Loope and Gruell 1973; Steele and others 1986). Modern wildfires on late seral landscapes tend to be larger, more intense, and more severe because of high biomass loadings, multilayer stand structures, and the high connectivity of the biomass at the stand and landscape level (Arno and Brown 1991; Keane and others 1997; Knight 1987). Moreover, only the unusually severe wildfires escape our suppression efforts in our present de facto exclusion policy (Arno and Brown 1989; Brown 1995). Covington and others (1994) state "the end result of fire exclusion in fire-prone forests is increasingly synchronous landscapes dominated by large, catastrophic disturbance regimes." There is a close inverse relationship between available fuel and mean fire frequency, so fire return intervals tend to increase with increasing fuels, and with increasing fuels comes increasing fire severities (Olsen 1981). Romme and Despain (1989) mentioned that the principal effect of 30 years of fire exclusion in Yellowstone National Park was to delay the onset of a major fire event, which was inevitable.

Fires on fire-altered landscapes may burn more area in fewer years, meaning that rare fire years, like 1910, may be especially high in fire activity (Bessie and Johnson 1995). And the increasing numbers of large, severe fires in 1 fire year will make suppression and control increasingly difficult further risking human life and property. This is again illustrated in figure 2 where the area burned has recently been increasing despite continually higher suppression efforts and technology. Few fire years will also tend to create less diversity in patch age and size because large areas tend to burn in 1 year, as demonstrated by the Yellowstone fires of 1988 (Baker 1989; Romme and Despain 1989).

High surface fuel loads and complex vertical stand structures increase the chance that modern surface fires will become crown fires and burn overstory trees through torching and crowning (Brown and Bevins 1986; Kolb and others 1998; Steele 1994). Early seral tree crowns tend to be heat porous and high off the ground while late seral trees have dense crowns extending nearly the entire length of the stem (Minore 1979). Higher flame lengths due to more surface fuels, coupled with lower and thicker shade-tolerant tree crowns, hasten the transition of a surface fire to a crown fire. Once a crown fire has started, the high leaf areas and high crown bulk densities typical of late seral forests favor propagation of fire throughout the crowns in a stand (Brown and others 1994; Rothermel 1991). Furthermore, these crown fires are likely to be propagated across the homogenous landscapes because high contagion between multilayered stands ensures high connectivity in crown fuels. Taylor and others (1998) estimated fire behavior potential changes using the frequency of three crown fire severity and six fire intensity classes on a southeastern British Columbia landscape during a climatologically normal fire season. The proportion of the landscape susceptible to a fire with more than 50 percent crown consumption increased from 7 to 14 percent from 1952 to 1992, and projected that proportion to increase to 29 percent by 2032. Therefore, the long-term consequences of fire exclusion in Rocky Mountain ecosystems is the conversion of historically low- to moderate-severity fire regimes to a high-severity, stand-replacement fire regime (Mutch 1994; Morgan and others 1998).

Land use changes on the Rocky Mountain landscape have also altered the ignition and spread patterns of historical fires. Barrett and others (1997) noted the majority of historical area burned occurred in sagebrush-grasslands that have now been altered and interrupted by agriculture, grazing, and land development. Fires burning through these rangelands often gained access to adjacent forest lands prior to European settlement and livestock grazing. Now, the continuity of fine fuels across these nonforest rangelands has been reduced or eliminated because of human land use activities such as development, agriculture, and grazing.

Insects and Disease

Insect and disease processes are also affected by the shift in host tree species across a landscape as fires are suppressed. Increases in insect and disease activity are attributed mostly

to increased stress and reduced vigor of the early seral, fire-dependent tree species (Heinrichs 1988; Hessburg and others 1994; Kolb and others 1998). This plant stress is a direct result of the increased competition from rising stand biomass and ballooning plant density (Harvey 1994, 1998; O'Laughlin 1998). Stressed plants and dense canopies are usually a recipe for severe insect and disease infestations (Heinrichs 1988). Harvey (1994) recognized that ecosystems with intact fire regimes have lower levels of plant stress, which reduces insect and disease infestations.

There are many examples of heightened insect and disease activity with fire suppression (Veblen and others 1994). Dwarf mistletoe (*Arceuthobium* spp.) has proliferated on landscapes with more older age classes resulting from fire exclusion especially in lodgepole pine and ponderosa pine (Alexander and Hawksworth 1976; Wilson and Tkacz 1996; Zimmerman and Laven 1984). The absence of fire is implicated in chronic spruce budworm (*Choristoneura occidentalis*) epidemics in many Douglas-fir and true fir stands in the Rockies (Carlson and others 1983; Hadley and Veblen 1993; Holland 198; Swetnam and Lynch 1993). Persistent defoliation by budworm outbreak can predispose host trees to Douglas-fir beetle (*Dendroctonus pseudotsugae*) and root rots. Bark and pine beetle and blister rust epidemics are replaced by root rot and fir decline diseases as the landscape converts from whitebark pine to subalpine fir and spruce cover types (Arno and Hoff 1989; Alexander and others 1990). Mountain pine beetle (*Dendroctonus ponderosae*), bark beetles, and dwarf mistletoe outbreaks are more common in southwestern ponderosa pine forests because tree densities increased due to lack of fire (Covington and others 1994). In spruce-fir forests of Colorado, spruce beetle (*Dendroctonus rufipennis*) outbreaks do not affect young (less than 80 years) postfire stands (Veblen and others 1994), which implies that long-term fire exclusion in the subalpine zone eventually would result in increased beetle activity as a larger portion of the landscape enters old-growth stages. However, it is debatable if fire suppression can actually prevent the infrequent but widespread fires of the subalpine zone that are associated with unusual weather events (Romme and Despain 1989).

Increased patch contagion from lack of fire may amplify the severity of insect and pathogen outbreaks. Contagion is generally described as the probability that similar patches are adjacent to each other. Landscapes dominated by one cover type seem to have the greatest potential for epidemic infestations of insects and disease because host species patches are near and migration distances are small. Conversely, patchy landscapes under native fire regimes often had greater probabilities of nonhost patches being barriers to pathogen dispersal. Research by Hessburg and others (1994) showed increases in budworm and tussock moth as the host tree species become more continuous across the landscape. Increased bark beetle outbreaks have been reported in many pine ecosystems where fires have been excluded (Wilson and Tkacz 1996).

Affected Rocky Mountain Ecosystems

The complex effects of fire exclusion are best understood when illustrated by examples. We selected several keystone Rocky Mountain ecosystems where fires were historically common but now have experienced several decades of fire exclusion (deMaynadier and Hunter 1997; Ferry and others 1995). Not all forest and range types are discussed because of lack of space, but it is estimated the ecosystems presented here comprise over 60 percent of Rocky Mountain lands (Ferry and others 1995).

The absence of fire on many Rocky Mountain grassland ecosystems usually resulted in the invasion of woody species (Arno and Gruell 1986). Historical grassland fire return intervals ranged anywhere from 2 to 27 years depending on topography and native American settlement, and have increased to over 27 years in many places because of development and fire suppression (Gruell 1986; Higgins and others1991; Seig 1997; Wright and Bailey 1982). Arno and Gruell (1986) found frequently occurring fires prior to 1890 tended to favor grasslands and confined tree establishment to rocky areas or topographically moist sites, but grazing and fire exclusion has allowed extensive areas of inland Douglas-fir (*Pseudotsuga menziesii* v. *glauca*) invasion into mountain grasslands. Fisher and others (1986) found an expansion of closed ponderosa pine forests at the expense of pine savannas and grasslands after 50 years of fire exclusion. Shinn (1980) documented how the association of fire with the deterioration of range resources by the public led to fire exclusion policies that resulted in juniper invasion and herbland decline in grass and shrublands of the Central Rockies. Conifer encroachment into montane sagebrush grasslands has resulted from delayed fire activity (Patten 1969).

Sagebrush-steppe ecosystems, encompassing some 45 million ha in the Western United States, typically burned at 60- to 110-year intervals prior to European settlement. However, fire frequency has increased in many areas due to the invasion of cheatgrass (*Bromus tectorum*) and medusahead *Taeniatherum caput-medusae*) (West and Hassan 1985; Whisenant 1990). Cheatgrass could expand from currently 6.8 million ha to over 25 million ha in the Great Basin (Ferry and others 1995). This increase in fire frequency would exert strong selective pressure against many native plants and animals.

Juniper and pinyon pine woodlands have been increasing in density and distribution since the early 1900s due to climate change, grazing, and most importantly, lack of fire (Gottfried and others 1995). The expansion is mostly into grass and shrublands (Ferry and others 1995). Bunting (1994) mentioned that the advance of juniper woodlands would have been curbed if the historical fire regime with 50-year fire-return intervals had not been altered by grazing, climate change, and wildfire suppression.

Aspen (*Populus tremuloidies*) is a short-lived, broadleaf tree species (100 to 125 years) that is seral to conifers in much of its nearly 3 million ha range in the Western United

States (DeByle and Winokur 1985). It is maintained by periodic mixed- to high-severity fires that kill most trees and allow aspen to regenerate from root suckers. The lack of fire has allowed the encroachment and dominance of conifers in many aspen stands. Restoration of fire is complicated by the fact that, today, aspen is becoming scarce and has poor vigor. On some landscapes, aspen is so rare that ungulates will concentrate in regenerating aspen stands to eat the high-quality aspen suckers, thereby preventing suckers from becoming trees.

Ponderosa pine forests occur on approximately 16 million ha in the Western United States. These low- and mid-elevation forests were historically maintained as grassy, parklike stands by frequent (2- to 40-year average fire return intervals), low-severity surface fires that killed many seedlings of its competitors, namely inland Douglas-fir and other true firs (fig. 1a) (Allen and others 1998; Arno and others 1995; Bogan and others 1999; Covington and Moore 1994). Removal of fire in these forests has created multiple-storied, dense forests that have a greater potential for stand-replacement fires (fig. 1b) (Steele and others 1986). Crown fires were historically rare in these forests, yet several Idaho fires in the 1980s and 1990s were active crown fires that burned large areas (USDA Forest Service 1993). Ponderosa pine forests in the Southern Rockies have been severely altered by the combined effects of heavy grazing, logging, and fire exclusion (Covington and Moore 1994; Fulé and others 1997). Historically, rare crown fires are now common in today's southwestern ponderosa pine forests, and these fires tend to be larger, more severe, and less common (Covington and Moore 1994; Swetnam and Baisan 1996). Tree density has experienced an almost tenfold increase, and basal areas have nearly doubled from 1883 to 1994 (Fulé and others 1997). Litter and duff depths have thickened by 200 percent, and woody fuels have also increased.

Although the effects of fire exclusion are not readily evident on lodgepole pine (*Pinus contorta*) landscapes, this forest type deserves mention because of its large range (4 million ha in the Rockies) and management implications in most of the Rocky Mountains (Romme 1982). Franklin and Laven (1990) recognized two types of fires occur in lodgepole pine forests: surface and crown fires. Crown fires killed all species but gave colonization advantage to the serotinous lodgepole pine. Surface fires only scarred lodgepole but killed most subalpine fir and spruce thereby delaying succession (Brown 1973). However, Brown and others (1995) found the majority of stand-replacement fire in lodgepole pine was actually severe surface fires. Fire exclusion has converted some forests from lodgepole pine to fir and spruce. In addition, some stand structures have gone from single-age or diameter-class dominance (even aged) to multiple age and diameter classes (unevenaged). As a result, these lodgepole pine stands are currently experiencing heavy infestations of mountain pine beetle, dwarf mistletoe, and root diseases (Alexander and Hawksworth 1976; Romme 1982; Zimmerman and Laven 1984).

Fire regimes in high-elevation whitebark pine forests, which occupy around 1 million ha in the Rocky Mountains, are typically described as mixed severity occurring at 80- to 500-year intervals (Arno 1980; Keane and others 1994). Although whitebark pine is long lived (more than 400 years), it is eventually replaced by subalpine fir and spruce without fire. Encroachment of subalpine fir into seral whitebark pine stands creates multilayered canopies with low crown base heights and high crown bulk densities, increasing the chance of stand-replacement fires (Murray and others 1995). The long-term consequence of fire exclusion in whitebark pine ecosystems is the conversion of a mixed-severity fire regime to a stand-replacement fire regime (Arno and others 1993; Loope and Gruell 1973; Hartwell 1997). Fires in this new regime will tend to be larger and more intense (Keane and others 1997). Effects of fire exclusion have recently been accelerated by the introduction of the exotic disease white pine blister rust (*Cronartium ribicola*), which has devastated many whitebark pine stands in the Northern Rockies, resulting in landscapes with abnormally high coverages of subalpine fir types (Keane and others 1994).

What's Next?

Restoration of some semblance of the native fire regimes seems a critical step toward improving the health of many Rocky Mountain ecosystems. However, there are problems. First, the immensity of any Rockies restoration effort is somewhat daunting considering projected future fires would need to burn from 3 to 7 times more than present. The 70 years of fire suppression have caused usually high live and dead fuel accumulations in many stands that, when ignited, would create a fire that would be abnormally severe and kill most of the trees (Mutch 1994). So, reintroduction of fire must be done carefully to prevent further damage to the stressed old-growth trees and other ecosystem components (Arno and others 1995). Developing a fire prescription (in other words, a set of weather conditions in which to burn) to minimize fire intensity but still accomplish restoration objectives is problematic because the high fuel loadings may preclude the implementation of a low-severity burn (Covington and Moore 1994). In addition, land management agencies are limited in conducting the extensive restoration treatments that are needed because of competing governmental regulations (for example, smoke, Endangered Species Act) and the high cost of implementation from environmental assessments to executing treatments. Despite these challenges, a functional restoration program is possible and necessary (Arno and Brown 1991; Babbit 1995; Brown 1995; Hardy and Arno 1996; Harvey 1998; Parsons and Landres 1998).

Many believe silvicultural cuttings are the only feasible method to remove some combustible biomass and thereby reducing fire intensities so fire severity will be similar to historical events (Arno and others 1995; Covington and Moore 1994). Baker (1992) felt that landscapes altered by settlement and fire suppression cannot be restored using only

the traditional methods of prescribed burning. Moreover, fire restoration cannot be done with just one or two prescribed burns or silvicultural treatments. Some field and simulation studies have shown that it may take as many as 50 to 75 years or at least two and as many as seven fire treatments or rotations to restore native fire regimes to stands and landscapes where fire has been excluded (Baker 1993, 1994; Keane and others 1997). Additionally, it may take more than one treatment to accomplish the objectives for one prescribed burn. For example, it may take two low-severity prescribed burns to achieve 90-percent mortality in shade-tolerant species for a stand. Site-specific studies and careful monitoring of the consequences of prescribed burning are essential to obtain goals related to ecosystem restoration. Even for ponderosa pine ecosystems, the role of surface versus stand-replacing fires in the pre-1900 fire regime is sometimes a contentious issue (Shinneman and Baker 1997).

The role of fire will continue to change in the Rocky Mountains as we continue to exclude fires from landscapes. It is not a question of "if" a landscape will burn, but rather, when it burns, how severe that fire will be. There are profound consequences of altered fire regimes as summarized in table 1. Fires occurring on fire-excluded landscapes will generate significantly different effects compared with effects of pre-1900 historical fires. Modern fires will be large and severe, killing more plants and altering many ecosystem processes. Bessie and Johnson (1995) point out that fires occurring in subalpine forests during severe weather years burn the most land area because wind coupled with excessive drought, and not fuels, drive these fires. Extreme fire years, such as 1910, 1987, 1994, and 1996, will tend to burn most plant communities regardless of fuels or ecosystem health, but the severity of these burns at the stand- and landscape-level will be dictated by the fuel loadings. One can only wonder what would happen if the extreme weather conditions of 1910 occurred today on our fire-excluded landscapes of the Rocky Mountains.

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