Historical Range of Variability
and Current Landscape Condition Analysis:
South Central Highlands Section,
Southwestern Colorado & Northwestern New Mexico

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EXECUTIVE SUMMARY

The South Central Highlands Section encompasses 11,475,000 acres (46,000 km^2) of the southern Rocky Mountains in southwestern Colorado and northwestern New Mexico, and includes all or a portion of the San Juan, Rio Grande, Uncompahgre, Gunnison, Carson, and Santa Fe National Forests. The purposes of this study were to (1) characterize the range of variability in structure, composition, function, and dynamics of the ecosystems of the south central highlands section during an appropriate reference period, defined here as the period of indigenous settlement prior to major impacts of Euro-American settlement in the late 1800s, (2) assess the current state of these ecosystems in terms of the extent to which their structure, composition, function, and dynamics depart from that range of variability, (3) evaluate the ecological significance of specific departures from the reference range of variability, i.e., identify any risks associated with current conditions, and (4) suggest opportunities and challenges related to restoring the altered structure, composition, function, or dynamics to something more closely resembling the reference period, in those situations where this is a desired management objective.

Three important themes underlie all of what follows. First, the pronounced elevational and topographic variability of the South Central Highlands Section has produced a great variety of vegetation types, each with its own distinctive landscape structure and disturbance regime. Therefore, issues related to natural ecological patterns and processes, disturbance regimes, and the magnitude of human-induced changes, need to be evaluated individually for each vegetation type. There is considerable risk in extrapolating from one vegetation type to another, because some of the differences in ecology and history are profound.

A second major theme of this report is that we are not starting with a "blank slate" in terms of choices and options in management, but that we have several important legacies of past forest use and management that must be dealt with in developing future management strategies. Some of these legacies, such as over-cutting of mature ponderosa pine in the early 1900s, may reduce our management options today. In this paper we identify some of these major legacies, and suggest ways of compensating for what society may now regard as poor choices in the past. In some situations, this will entail restoration of ecological components and processes that have been lost.

The third major idea underlying this analysis is that our understanding of the ecology of the ecosystems in the southern Rocky Mountains is still inadequate to permit complete confidence in our management decisions today. We must acknowledge that we are going to make some mistakes. However, the seriousness of these mistakes can be reduced by placing forest management into a consciously experimental framework, by carefully observing the system's response to our well-intentioned efforts -- e.g., testing explicit ecological hypotheses with each timber sale, prescribed burn, or road closure -- and by modifying our actions appropriately as we learn more about the system (i.e., practicing adaptive management).

The organization of this report is structured around the three major themes outlined above. Following an overview of the South Central Highland Section's geography, climate, and vegetation, we describe and justify the concept of reference conditions, and then present a brief review of the principles of ecosystem management and adaptive management that are emerging from studies throughout North America. This introduction is then followed by a detailed analysis of each major vegetation type in the South Central Highlands Section, with respect to (i) vegetation structure, composition, and distribution, (ii) reference conditions, and (iii) legacies of past human land use; and we close the report with an overview of management challenges and opportunities for the region as a whole within the context of ecosystem management and adaptive management.

Ponderosa Pine Forests

Ponderosa pine forests are found at lower to middle elevations throughout the South Central Highlands Section. Ponderosa pine (*Pinus ponderosa* var. *scopulorum*) is the major canopy species, often with Gambel oak (*Quercus gambelii*) dominating the understory. Many ponderosa pine forests were composed historically of open stands of large trees, but denser stands also were present. Pre-1900 canopy densities and basal areas in open stands in the San Juan Mountains and Uncompahre Plateau were generally higher than in ponderosa pine forests of northern Arizona. Trees typically grew in clumps interspersed with open areas, and there was wide variation in the size of clumps and open areas.

The most important natural disturbance process in ponderosa pine forests prior to Euro-American settlement was fire; other natural disturbances included various tree-killing insects (notably bark beetles), pathogens, and parasites. Located geographically between the Southwest and the Colorado Front Range, historical fire regimes of ponderosa pine forests in the South Central Highlands Section were intermediate between the Southwestern model of frequent, low-severity fires and the variable-severity model of the Front Range which included a component of high-severity fire. Major fire years in the South Central Highlands Section, as in the Southwest, usually occurred when one to three very wet El Niño years -- in which abundant fine fuels were produced -- were followed by a very dry La Niña year -- when the accumulated fine fuels dried out and carried extensive fires. Approximately half of documented fires before 1900 in the San Juan Mountains burned in the summer season, and half burned in dormant seasons (spring or fall). However, in the years with very extensive fires (e.g., 1748, 1820, 1851, and 1879), fires apparently burned all summer long.

After about 1870 (earlier in some places), most of the ponderosa pine forests in the region were altered dramatically by livestock grazing, logging, and fire suppression by Euro-American settlers. By the mid-20th century, nearly all of the old growth ponderosa pine forests of this region had been liquidated. Consequently, unlogged, old-growth ponderosa pine stands today are very few in number. The most obvious legacy of the early logging of ponderosa pine forests is the general lack of large, old trees and snags

in these forests today; most stands are dominated by relatively small, young trees. Early livestock grazing was largely unregulated, and range conditions began deteriorating as early as the 1890s. As a result, some highly palatable species, such as the bunchgrasses *Muhlenbergia montana* and *Festuca arizona,* may have been locally extirpated from many stands, and the relative abundance of surviving species has been drastically altered. Heavy livestock grazing also disrupted the historical fire regime as the animals removed the grasses and herbs that formerly carried light fires through the forest. Lightning and humans still started fires, but they could not spread across bare ground. Grazing began to be regulated more effectively on federal lands beginning in the 1930s, but by this time the Forest Service was developing an effective fire suppression capability. Over most of the ponderosa pine zone in the South Central Highlands Section, there were no extensive fires throughout the 20th century. When fires occur today in ponderosa pine forests, they sometimes are more severe than before 1900, in part because of the greater fuels that have accumulated during 100+ years without fire. Earlier springs, earlier snow-melt, and longer fire seasons since the mid-1980s also have contributed to larger and more severe fires in ponderosa pine and other vegetation types throughout the West.

Spruce-Fir Forests

Spruce-fir forests, dominated by Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*), are the coldest and wettest forest type in the South Central Highlands Section, occurring at elevations of 2,720 – 3,580 m. The ground layer vegetation in these forests is highly variable but typically includes a rich mixture of mesophytic herbs.

The two most important and ubiquitous kinds of disturbance in high-elevation landscapes during the period of indigenous settlement were stand-destroying fire and bark beetle outbreaks. Stand-destroying fires generally initiated stand development and maintained a relatively coarse-grained mosaic of successional stages across the subalpine landscape. During the long intervals between fires, however, and especially as stands reached the later stages of development, the disturbance regime of individual stands was dominated by chronic, fine-scale processes involving insects, fungi, and wind, that killed individual trees or small groups of trees

Fires in high-elevation forests of the Rocky Mountains were infrequent and usually very small, because late-lying snowpacks and frequent summer rain showers kept fuels too wet to burn throughout most of the growing season. However, in the rare dry years when weather and fuel conditions were suitable for extensive burning, large portions of the subalpine landscape were burned in severe, stand-destroying fires. We determined that the fire turnover time (the average time required to burn an area equal in size to the entire study area, or the average interval between successive fires at a single site) in a spruce-fir dominated landscape of the western San Juan Mountains was ca. 300 years. Many individual stands escaped fire for many hundreds of years. For example, an ancient spruce-fir forest comprising 180 ha at the headwaters of Martinez Creek in the San Juan National Forest apparently has not burned for 600

years or longer, nor has it been affected by major windthrow events or severe insect outbreaks. Fires often were followed by rapid development of aspen forests with spruce and fir in the understory, especially in the lower-elevation portion of the spruce-fir zone. At the highest elevations, and in other places where aspen was not present, spruce and fir directly re-colonized burned sites; this was a very slow process and hundreds of years were required for a forest to again develop under these circumstances.

The spruce beetle (*Dendroctonus rufipennis*) is a native insect whose larvae feed on the phloem of large, living or dead, Engelmann spruce trees. Most of the time, the beetles persist in low-density, endemic populations that have little impact on forest structure. Periodically, however, populations explode into an outbreak, and the beetles may kill many or most of the large-diameter spruce trees over areas of thousands of hectares. However, small-diameter host trees and other tree species usually are not attacked by the beetles. Beetle outbreaks are followed not by establishment of new tree seedlings, but by accelerated growth of previously suppressed sub-canopy and understory individuals. Return intervals of extensive mountain pine beetle and spruce beetle outbreaks are shorter than the intervals between successive stand-destroying fires, but are still in the range of decades to centuries

In general, the effects of Euro-American activities have been less ubiquitous, or less intense, in spruce-fir forests than in lower-elevation forest types of the South Central highlands Section--although in localized areas the impacts have been just as great. Despite policies of fire exclusion throughout much of the 20th century, the fire regime in spruce-fir forests apparently has changed relatively little during the past century, primarily because large fires in subalpine forests are controlled primarily by regional weather conditions. Few large fires have occurred in spruce-fir forests of the South Central Highlands Section during the 20th century, but given the long fire-return intervals that naturally characterize these systems, the current fire-free intervals may not be far outside the range of variability in fire intervals that characterized the period of indigenous settlement. Certainly the time that has elapsed since the last fire in any individual stand is not exceptionally long, but larger landscape units (e.g., watersheds) may have gone for longer than usual without a large fire.

The most conspicuous anthropogenic disturbances in spruce-fir forests are clear-cut logging and road-building. These activities generally came later to the spruce-fir forest zone than to forests at lower elevations. For example, there was almost no logging and little road building in spruce-fir forests in a large portion of the central San Juan Mountains as late as 1950 Clear-cutting was used extensively during the 1950s - 1970s, but generally has been discontinued in spruce-fir forests because of problems with stand regeneration. Since the 1970s, silvicultural methods have emphasized partial cutting techniques rather than clear-cutting. Since most partial cutting involves selective removal of the larger trees, there is a gradual shift to smaller size classes. Large snags and fallen logs--key structural elements for many wildlife species and soil processes--gradually may disappear. Of all the novel kinds of disturbances that humans have introduced in high-elevation forest

of the South Central Highlands Section during the last century, roads may be the most ubiquitous and significant long-term legacy of our activities.

Aspen Forests

Aspen forests form an important vegetation type between about 2,000 and 3,300 m throughout the mountainous portions of Colorado, Utah, and northern New Mexico These forests grow on a wide variety of geologic substrates and soil types, although best growth usually is seen on deep, loamy soils having high nutrient availability. Aspen is a clonal species in which a single genetic individual may cover a large area (up to several hectare) and be represented by tens, hundreds, or even thousands of individual stems. Ranging from pure aspen stands ("stable aspen") to stands that are co-dominated by aspen and conifer species (seral aspen"), aspen forests generally are characterized by a luxuriant understory of deciduous shrubs and herbs and provide some of the most important wildlife habitats in the South Central Highlands Section.

In a survey of 100 stable and seral aspen forests in the western portion of the San Juan National Forest, stable aspen stands were strongly associated with lower elevations near the ponderosa pine zone, and weakly associated with shale substrates rather than sandstones or igneous rocks, whereas seral stands were found primarily at the higher elevations. This pattern suggests that stable aspen stands in the South Central Highlands Section may have developed in response to very short fire intervals in the past, but more research is needed to fully explain the distribution of stable vs. seral aspen in this region.

The most important natural agent of disturbance in aspen forests during the reference period was fire; other natural disturbances included windthrow, fungal diseases, tent caterpillars and other insects, snow damage, hail, lightning, and sunscald. Aspen stems are easily killed even by relatively low-intensity fire. However, the aspen root system usually is unharmed by fire, and it typically produces abundant root sprouts within the first growing season after the fire. A detailed study of fire history in the western portion of the San Juan National Forest revealed that there were extensive fires in the 1870s and 1860s, but no large fires after that time. Median aspen stand age in the 1880s was about 70 years. If half of the landscape had burned within the previous 70 years, then it would require about twice this length of time, or 140 years, for a cumulative area equal to the entire landscape to burn. This is our best estimate of the fire turnover time in this aspen-dominated landscape during the period of indigenous settlement.

Aspen in the Rocky Mountains is usually harvested by clear-cutting small blocks up to about 40 acres in size. Clear-cutting usually is followed by prolific sprouting of new stems from the root system, and often leads to increased understory production as well. However, some clear-cuts have failed to produce sufficient sprouts to regenerate the cut stand, especially on relatively flat sites having a high water table or heavy browsing by native or domestic ungulates

Extensive and seemingly precipitous mortality of mature aspen stems has been observed in many parts of Colorado and other western states since about 2000. The phenomenon has been labeled "Sudden Aspen Decline" or

"SAD." The cause of the mortality has not been determined.

Aspen cover has decreased in many areas since the early 1900s, leading to concerns about aspen decline. However, the area covered by aspen forests actually has remained stable or even increased throughout the past century in other areas. Assessing the magnitude and significance of aspen decline is complicated by the fact that investigators have used multiple, and often conflicting, concepts of "decline," and different definitions of aspen decline lead to very different conclusions. We evaluated two different ecological processes that have been interpreted as "aspen decline:" (i) replacement of aspen by conifers in the absence of fire or logging, and (ii) failure of aspen to regenerate because of chronic heavy browsing by native or domestic ungulates. We view the first process—a trend of increasing conifer dominance-- as a natural successional process that has always occurred during long periods without major stand disturbance and that is not a serious threat to the long-term persistence or ecological function of aspen. Future fires will reduce the conifer component and once again increase aspen dominance. The second process—failure of stands to regenerate because of excessive browsing—appears to be a more serious form of aspen decline that could lead to loss of aspen clones over a long period of time. Unfortunately, we do not have a good estimate of how much of the aspen in this region is declining as a result of excessive ungulate browsing; this is an important topic for future research. Nevertheless, aspen does not appear to be in imminent danger of disappearing from landscapes of the South Central Highlands Section.

Mixed Conifer Forests

The mixed conifer forest type occurs at elevations ranging from about 2270 – 3030 m (7,500 – 10,000 ft). Mixed conifer forests are perhaps the most variable and complex of any forest type in the southwestern mountains, in terms of species composition, stand structure, and dynamics. They also have received relatively little research attention Consequently, we have a relatively poor understanding of the long-term ecological dynamics and interactions that have shaped the mixed conifer landscape in the past, and that explain biotic responses to current management activities.

Environmental conditions, species composition, and disturbance regimes in the mixed conifer zone are intermediate between characteristics of the adjacent ponderosa pine zone at lower elevations and characteristics of the adjacent spruce-fir zone at higher elevations. Mixed conifer forests in the South Central Highlands Section can be divided into either two or three general sub-types or "phases." The two-phase classification recognizes a **warm-dry** phase, generally at lower elevations, and a **cool-moist** phase, generally at higher elevations. **Warm-dry** mixed conifer forests resemble ponderosa pine forests in general stand structure, but Douglas-fir and white fir are also important components of these forests. **Cool-moist** mixed conifer forests typically lack ponderosa pine, have a greater abundance of Douglas-fir and white fir, and, on some sites, include subalpine/corkbark fir and Engelmann spruce. The three-phase classification of mixed conifer forests retains the same **warm-dry** phase, but subdivides the

cool-moist phase into two units: a **cool-moist** phase with the same name as before but a different definition, and a new **cold-wet** phase. Ponderosa pine is the key indicator of the **warm-dry** phase, and subalpine/corkbark fir and Engelmann spruce are the key indicators of the **cold-wet** phase. Douglas-fir and white fir are the constant species occurring throughout the mixed conifer forest type in this three-phase classification, while ponderosa pine, Engelmann spruce, and subalpine/corkbark fir are absent or only minor components in the **cool-moist** mixed conifer phase of the three-phase classification.

During the period of indigenous settlement, many of the **warm-dry** mixed conifer forests of the South Central Highlands Section appear to have been composed of relatively open stands dominated by large ponderosa pine, Douglas-fir, and white fir trees. The understory contained saplings of white fir, Douglas-fir, aspen in some places, Gambel oak, and other shrubs. This kind of stand structure was maintained in part by recurrent fires of relatively low to moderate intensity, although patches of stand-replacing fire probably also were present, and relatively dense stands could be found in places. Fire intervals in **warm-dry** mixed conifer forests typically were measured in decades, and were longer than in ponderosa pine forests but shorter than in forest types at higher elevations. **Cool-moist** or **cold-wet** mixed conifer forests grow in areas that are too wet for the frequent fires that characterize the lower-elevation forest types. Late-lying snowpacks and more frequent summer rains keep fuels moist throughout most of the fire season in most years. Consequently, stands regularly may persist for many decades or centuries without fire. When fires finally do occur during prolonged dry periods, the fuel bed that has developed during the long fire-free period can support a high-intensity fire that kills most of the above-ground vegetation in the stand. Pre-1900 fires in **cool-moist** or **cold-wet** mixed conifer forests occurred at intervals of many decades to centuries, and typically burned at mixed severity, with patches of lethal fire interspersed with patches of low tree mortality. The result was a spatially complex and temporally variable landscape mosaic of stands having a spectrum of structural and compositional characteristics.

In addition to fire, other important agents of natural disturbance in all mixed conifer forests during the reference period included windthrow, fungal diseases, and outbreaks of various insect species. Although these disturbances were important in localized areas, they probably had far less impact on landscape-scale forest dynamics than fire – with the exception of spruce budworm outbreaks, which may have been as important as fire.

The major Euro-American impacts on **warm-dry** mixed conifer forests are generally similar to impacts in ponderosa pine forests. Extensive and unregulated livestock grazing, which began in the late 1800s, removed fine herbaceous fuels that formerly carried low-intensity surface fires, and resulted in a relatively sudden and dramatic drop in fire frequency. Today, many **warm-dry** mixed conifer stands have not burned in over one hundred years, and many have greater tree densities than was typical of the reference period. Euro-American impacts on **cool-moist** or **cold-wet** mixed conifer forests are

generally similar to impacts in spruce-fir forests. Fire regimes of **cool-moist** or **cold-wet** mixed conifer were not so strongly affected by early grazing and fire suppression. Fire frequency and behavior in these and other high elevation forests were controlled primarily by weather conditions, not fuels, and most fires that covered large areas were severe, stand-replacing fires. There were few large fires during the 20th century in **cool-wet** mixed conifer and other high elevation forests of the South Central Highlands Section, but this may be due as much to wet weather conditions and limited ignitions as to active fire suppression efforts. Many **cool-moist** or **cold-wet** mixed conifer forests have high tree densities today, but this does not represent a major departure from historical conditions in these forests.

Extensive logging occurred throughout the lower elevation forests of the South Central Highlands Section, beginning in the late 1800s and especially during the first half of the 20th century. In nearly all of the accessible stands of ponderosa pine and **warm-dry** mixed conifer forest, the large trees were selectively removed. The combination of fire exclusion, selective logging, and favorable climatic conditions for young tree establishment in the early 20th century, has created an unusual stand structure in many **warm-dry** mixed conifer forests today. The large, old ponderosa pine and Douglas-fir trees that formerly dominated the canopy are gone, and the stands now are dominated by smaller, young individuals of pine, Douglas-fir, and white fir. White fir especially has increased in density during the long fire-free period of the 20th century, and establishment of

new ponderosa pine and Douglas-fir individuals has tapered off or stopped in many stands, probably because of the dense stand conditions.

In contrast to lower-elevation forests, the **cool-moist** or **cold-wet** mixed conifer forests and other high elevation forest types were not subjected to extensive logging until the middle of the 20th century in many portions of the South Central Highlands Section. Nearly all of the timber harvests in mixed conifer forests have involved road building and some form of partial cutting; little clear-cutting has occurred in this forest type. Selective removal of the high-value canopy trees, with little disturbance to the smaller trees or the species of lower economic value, has created some unusual stand structures that do not resemble the kinds of structures resulting from natural disturbances. Some of these stands are now dominated by a dense growth of small to moderate sized firs and shrubs, with little or no Douglas-fir regeneration. Douglas-fir may not become re-established in such stands for centuries, unless fire occurs or an active restoration treatment is applied.

Other Vegetation Types

This report also gives a brief treatment of seven other vegetation types that are of relatively limited extent in the South Central Highlands Section. However, these types are ecologically important where they occur. They include piñon-juniper woodlands, mountain grasslands, riparian vegetation and wetlands, alpine ecosystems, mountain shrublands, lodgepole pine forests, and bristlecone pine forests.

Many of the ecosystems of the South Central Highlands Section today are substantially different from the ecosystems that existed during the reference period, and many of these changes are regarded as undesirable. In this chapter we first review some of the major challenges facing land managers in the South Central Highlands Section, and then explore some of the opportunities for mitigation and restoration of damaged or degraded components of the ecosystems in the region.

A. **Challenges: Forest Fragmentation by Roads & Logging**: One of the major global challenges to biodiversity and ecological integrity is fragmentation of wildland habitats by roads, logging, agriculture, home building, and other human developments. Roads are unprecedented features in ecological history, and have greater impacts on plants, wildlife, and natural ecological processes than almost any other feature of the human-influenced landscape. Road densities are surprisingly high throughout the South Central Highlands Section. For example ponderosa pine forests in the San Juan National Forest in southwestern Colorado contain 2.60 miles of road per square mile.

To evaluate quantitatively the effects of road-building and logging on high-elevation forests, we reconstructed logging history in the Pagosa District of the San Juan National Forest. Although logging began in the late 1800s in most of the ponderosa pine forests of the region, very little logging occurred in high-elevation forests of this district prior to 1950. The number of acres affected by logging each year increased through the 1950s and 1960s, reached a peak in the 1970s, and then declined through the 1980s and 1990s. This half-century of logging has directly affected only a small proportion (ca. 5 %) of the total spruce-fir forest acreage on the Pagosa District. However, nearly all (85%) of the suitable timber base in spruce-fir has been entered at least once for timber removal. We also measured changes in several metrics of landscape structure between 1950 and 1992 using the program FRAGSTATS. For all 45 landscape metrics reported by FRAGSTATS, the average percentage change from 1950 - 1993 was 15 - 25 % when roads were included, but only 3 - 8 % when roads were excluded. The number of distinct patches has increased, average patch size has decreased, edge density (meters of edge between two different cover types) has increased, and mean core area (patch area >50 m from an edge) has decreased

B. **Challenges: Outdoor Recreation, Roads, and Exurban Development**: Outdoor recreation is now the principal use of most forests in the southern Rocky Mountains. Although often regarded as a benign activity that has little or no adverse impact on ecological conditions or processes, outdoor recreation actually poses a major threat to biodiversity and ecological integrity. Of all species federally listed as endangered or threatened, outdoor recreation affects more of these species than any other activity except water development. Outdoor recreation affects plant and wildlife species via direct harvest (hunting, fishing, and collecting), disturbance (either accidental or intentional), habitat modification, and pollution.

The human population in the Rocky Mountain region is one of the fastest growing in the nation, and much of the new development is occurring in formerly rural or undeveloped areas. Exurban development poses several important ecological and social challenges in the South Central Highlands Section. First, the houses and access roads fragment natural habitats and displace sensitive wildlife. Second, exurban development often occurs in ponderosa pine forests and other vegetation types where fire was formerly an important ecological process. Not only are these houses vulnerable to loss in wildfire, but restoration of a natural fire regime is severely constrained by the presence of vulnerable structures that must be protected from fire. Finally, the costs to local governments of maintaining roads and providing services (fire protection, police, and schools) to these scattered residents usually exceed the revenues generated by property taxes; in effect, residents in town are subsidizing the exurban life style. The ecological and social costs of exurban development can be reduced in a number of ways, e.g., by clustering developments and protecting agricultural land through the use of conservation easements.

C. Challenges: Climate Change: The earth's climate is changing, global temperatures are increasing, shifts in precipitation patterns are likely to become pronounced in the next century, and weather related disturbances (e.g., hurricanes and droughts) are likely become more frequent and severe. The potential for large, severe fire also is increasing. An upsurge in the frequency of large fires began in the mid-1980s and is expected to continue. Climate change complicates all of our efforts in land management.

D. Opportunities: Principles of Ecological Restoration: One of the major land management activities in the twenty-first century likely will be restoration of ecosystems that were degraded during the past century. *Passive restoration* involves removal of the stressors that have degraded an ecosystem, e.g., regulating livestock grazing and recreational activity, and allowing lightning-ignited fires to burn without interference. *Active restoration* involves conscious modification of an ecosystem to achieve a desired result, e.g., selective logging and manager-ignited prescribed fire. Active restoration is not needed (nor is it feasible) in every location throughout the South Central Highlands Section. An important component of all restoration programs is protecting and maintaining those areas that have been little altered, to serve as reference areas for restoration treatments in degraded areas.

E. Opportunities: Passive Restoration: Two major stressors in the South Central Highlands Section are excessive fire suppression and poorly conducted livestock grazing. Land management agencies now have opportunities to allow fires or portions of fires to burn naturally with minimal human interference (2009 Implementation Guidelines for Fire Management on Federal Lands). Innovative new grazing management practices, such as those being conducted by the Quivira Coalition, headquartered in Santa Fe, New Mexico, are available to help manage and passively restore these grasslands for long-term viability and sustainability.

F. Opportunities: Active Restoration of Ponderosa Pine Forests: Perhaps the clearest opportunities and methods for active ecological restoration in the South Central Highlands Section are seen in ponderosa pine forests. The basic approach involves mechanical thinning to reduce tree density and basal area and to restore a clumped pattern of tree dispersion in the stand, followed by a low-severity prescribed fire. In conjunction with the prescribed burning program, it is necessary to protect recently treated stands from excessive livestock grazing and to mitigate invasive, non-native plants. Monitoring is implemented before treatment, repeated just after treatment, and then continued for several years to detect short-term and long-term effects. It is important to emphasize that restoration of an open stand structure should be applied only to some, not all, ponderosa pine forests throughout the South Central Highlands Section.

G. Opportunities: Emulating Natural Disturbance Processes in High-Elevation Forest Landscapes:

Where the management emphasis is not on ecological restoration per se, but on sustainably harvesting an economic product in such a way that ecosystem services are not impaired or degraded, innovative new approaches to silviculture are being developed, based on mimicking the kinds of natural disturbances that have shaped these forests for hundreds or thousands of years. Because the native biota are already adapted to these kinds of disturbances, anthropogenic disturbances of similar kind and intensity are less likely to produce unexpected or undesirable ecological surprises than are human disturbances that do not resemble the natural disturbance regime. Possible strategies for making anthropogenic disturbances mimic more closely the natural disturbance regime in high-elevation landscapes involve: (i) revising the size, shape, and spatial arrangement of logging units, (ii) providing "biological legacies" of the former stand in logged areas by means of variable retention harvest systems and retaining coarse wood, and (iii) reducing the extent and impact of roads.

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CHAPTER 1: INTRODUCTION AND OVERVIEW

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A. OBJECTIVES AND ORGANIZATION OF THIS REPORT

The South Central Highlands Section encompasses 11,475,000 acres (46,000 km^2) of the southern Rocky Mountains in southwestern Colorado and northwestern New Mexico, and includes all or a portion of the San Juan, Rio Grande, Uncompahgre, Gunnison, Carson, and Santa Fe National Forests (Figure I-1). The purposes of this study were to (1) characterize the range of variability in structure, composition, function, and dynamics of the ecosystems of the south central highlands section during an appropriate reference period, defined here as the period of indigenous settlement prior to major impacts of Euro-American settlement in the late 1800s, (2) assess the current state of these ecosystems in terms of the extent to which their structure, composition, function, and dynamics depart from that range of variability, (3) evaluate the ecological significance of specific departures from the reference range of variability, i.e., identify any risks associated with current conditions, and (4) suggest opportunities and challenges related to restoring the altered structure, composition, function, or dynamics to something more closely resembling the reference period, in those situations where this is a desired management objective.

Three important themes underlie all of what follows. First, the pronounced elevational and topographic variability of the South Central Highlands Section has produced a great variety of vegetation types, each with its own distinctive landscape structure and disturbance regime. Therefore, issues related to natural ecological patterns and processes, disturbance regimes, and the magnitude of human-induced changes, need to be evaluated individually for each vegetation type. There is considerable risk in extrapolating from one vegetation type to another, because some of the differences in ecology and history are profound.

A second major theme of this report is that we are not starting with a "blank slate" in terms of choices and options in management, but that we have several important legacies of past forest use and management that must be dealt with in developing future management strategies. Some of these legacies, such as over-cutting of mature ponderosa pine in the early 1900s, may reduce our management options today. In this paper we identify some of these major legacies, and suggest ways of compensating for what society may now regard as poor choices in the past. In some situations, this will entail restoration of ecological components and processes that have been lost.

The third major idea underlying this analysis is that our understanding of the ecology of the ecosystems in the southern Rocky Mountains is still inadequate to permit complete confidence in our management decisions today. We must acknowledge that we are going to make some mistakes. However, the seriousness of these mistakes can be reduced by placing

Figure I-1. Major features of the South Central Highlands Section in southwestern Colorado and northwestern New Mexico.

forest management into a consciously experimental framework, by carefully observing the system's response to our well-intentioned efforts -- e.g., testing explicit ecological hypotheses with each timber sale, prescribed burn, or road closure -- and by modifying our actions appropriately as we learn more about the system (i.e., practicing adaptive management).

The organization of this report is structured around the three major themes outlined above. Following an overview of the South Central Highland Section's geography, climate, and vegetation, we describe and justify the concept of reference conditions, and then present a brief review of the principles of ecosystem management and adaptive management that are emerging from studies throughout North America. This introduction is then followed by a detailed analysis of each major vegetation type in the South Central Highlands Section, with respect to (i) vegetation structure, composition, and distribution, (ii) reference conditions, and (iii) legacies of past human land use; and we close the report with an overview of management challenges and opportunities for the region as a whole within the context of ecosystem management and adaptive management.

The analyses reported here are based primarily on recent ecological investigations conducted in the San Juan National Forest by the authors, databases available for the San Juan National Forest, and the ecological literature. As a result, our analysis and examples emphasize the San Juan National Forest portion of the South Central Highlands Section. However, extrapolation to the remainder of the Section is made wherever warranted or possible, and major gaps in data and analysis for other parts of the Section are identified. A companion report (Romme and McGarigal 2006) provides more detail on the plant communities and vegetation dynamics of the Uncompahgre Plateau, located in the northwestern corner of the South Central Highlands Section. In addition, the historical range of variability of ecosystems in the San Juan Mountains and Uncompahgre Plateau has been described by means of a dynamic landscape simulation model (RMLANDS), which provides a somewhat different but complementary picture of the natural ecological dynamics of this region based on a different kind of methodology (McGarigal and Romme 2005).

B. OVERVIEW OF THE REGION'S PHYSICAL GEOGRAPHY AND VEGETATION

The South Central Highlands Section is centered on the massive San Juan Mountains system, which extends from west to east across southwestern Colorado, and the South San Juan Mountains which run southward from the east end of the San Juan Mountains into northwestern New Mexico. Associated with the central San Juan system are several smaller but nevertheless impressive mountain ranges and high plateaus, including the Uncompaghre Plateau, the Mesa Verde cuesta, the La Plata and La Garita Mountains in Colorado, and the Nacimiento and Jemez Mountains in New Mexico (Figure I-1). Most of the area lies within the southern Rocky

Mountains province but it also includes a portion of the Colorado Plateau (Blair 1996a). Elevations range from approximately 1,500 m (5,000 ft) in the basins and valleys of northwestern New Mexico to over 4,200 m (14,000 ft) in the central San Juan Mountains of Colorado. The continental divide runs along the crest of the South San Juan Mountains, and several important rivers rise in the high mountains of the South Central Highlands Section, including the Chama, Conejos, Rio Grande, San Juan, Dolores, and major tributaries of the Gunnison.

The geology of the South Central Highlands Section is complex. The mountains and basins are composed of material dating from ancient Precambrian rocks to recent alluvial deposits (Ellingson 1996a, Campbell 1996, Campbell and Brew 1996). The landforms we see today were created by a variety of geological processes, including plate tectonics, volcanism, glaciation, and erosion (Brew 1996, Ellingson 1996b, Blair 1996a).

The vegetation, flora, and fauna of the South Central Highlands Section are rich and complex, as would be expected in such a topographically and climatically diverse region. Povilitis (1993, 2000) analyzed the Greater San Juan Mountain Area, which encompasses most of the South Central Highlands Section plus some adjacent areas. The Greater San Juan Mountain Area supports some 507 native vertebrate species, as well as biotic communities representative of eleven potential natural vegetation types (sensu Kuchler 1964). Thirty-two species of vascular plants and 40 species of vertebrates are officially recognized as endangered, threatened, or of special concern at the state or federal level.

Little is known about the invertebrate fauna, but there is a rare, endemic butterfly (the Uncompahgre fritillary) known from only two sites in the San Juan Mountains. There also are several species of moss that are found primarily in northern Canada and Alaska but exist also as small disjunct populations in cool, wet habitats of the San Juan Mountains (D. W. Jamieson, personal communication 1996).

A great variety of vegetation types is found in the region, each having a unique ecological setting and history (Romme et al. 1992, Spencer and Romme 1996, Floyd-Hanna et al. 1996, Jamieson et al. 1996, Somers et al. 1996), as well as distinctive human impacts and changes since Euro-American settlement (Table I-1). At the lowest elevations the vegetation is dominated by pinon-juniper woodlands and various kinds of semi-arid grasslands and shrublands. Moving into the foothills of the mountains, and on the tops of broad plateaus and mesas adjacent to the mountains, we find extensive forests of ponderosa pine interspersed with tall shrub-dominated stands (Petran chaparral). The middle slopes of the mountains are covered by a mosaic of mixed conifer and aspen stands, broken by occasional meadows and grasslands. At the highest elevations are extensive spruce-fir forests, subalpine meadows, and treeless alpine communities on the highest peaks. Running through all of these vegetation types are riparian woodlands and meadows along the borders of perennial rivers and streams.

For the purposes of this analysis, we focus on four major vegetation types that are of particular ecological and economic significance throughout the South Central Highlands Section:

ponderosa pine forests, spruce-fir forests, aspen forests, and mixed conifer forests. Ponderosa pine and spruce-fir forests grow at the lowest and highest elevations, respectively, and differ the most among forest types in historical structure and dynamics and in the type and magnitude of change during the past century. Aspen and mixed conifer forests grow at intermediate elevations and share many characteristics with the forests at higher and lower elevations. Therefore, we first cover ponderosa pine and spruce-fir forests, and then examine aspen and mixed conifer forests. We also provide a brief review and assessment of six other vegetation types that are less widespread within the South Central Highlands Section but also important: piñon-juniper woodlands, mountain grasslands, riparian areas and wetlands, alpine ecosystems, mountain shrublands, lodgepole pine forests, and bristlecone pine forests.

In general, the vegetation at lowest elevations has had the longest and most intensive impacts from Euro-American settlement, whereas at the highest elevations the impacts have been somewhat lighter and more recent. For example, grazing and fire exclusion began to significantly change the structure of ponderosa pine forests as early as the mid-1800s in the southern portion of the South Central Highlands Section. In contrast, extensive logging and road-building in spruce-fir forests of the central portion of the Section did not begin until 1950. There are important exceptions to this generalization. For example, fire exclusion has profoundly altered some lower and mid-elevation ponderosa pine, aspen, and mixed conifer forests, but has had relatively little impact on the broad-scale structure and function of piñon-juniper woodlands and spruce-fir forests -- which represent both the uppermost and lowermost forest zones. All of these topics are developed more fully in the chapters that follow.

C. CLIMATIC VARIABILITY IN SPACE AND TIME

1. Geographic Patterns in Climate

The climate of the South Central Highlands Section is extremely variable because of the tremendous range in elevation and topography. Generally, the higher elevations are cooler and wetter than the lower elevations, and north-facing aspects are cooler and wetter than south-facing aspects. The eastern and northeastern portions of the South Central Highlands Section, in the lee of the main crest of the San Juan range, lie within a rain shadow, since most major winter storm systems move into the region from the west or southwest; these eastern and northeastern areas generally receive less precipitation than do areas at comparable elevations in the western and southern portions of the Section. Overall, the region is arid, and precipitation exceeds evaporative demands only at the highest elevations (Spencer and Romme 1996).

Climatological stations are few in number, but the range of climatic variability within the region is illustrated by the following examples (Keen 1996;125). In Cortez, Colorado, located in a broad basin near the foot of the mountains at 1,880 m (6,200 ft), temperatures range from an average high of 31 ^{o} C (88 ^{o} F) in July to an average low of -11 ^{o} C (12 ^{o} F) in January. In

Table I-1. Overview of major vegetation types in the South Central Highlands Section.
Environmental data provided by Jeffery S. Redders, San Juan National Forest.

Vegetation Type	Environment	Dominant Species
Piñon -juniper woodlands & Petran chaparral	lower elevations (1500-2600 m), warm, dry	Colorado piñon, Utah juniper, Rocky Mountain juniper, Gambel oak, Utah serviceberry, bitterbrush, muttongrass
Ponderosa pine forests	lower elevations (2100-2600 m), warm (90-110 frost-free days/yr), dry (50-65 cm/yr)	Ponderosa pine, Gambel oak, snowberry, Arizona fescue, muttongrass, elksedge
Mixed conifer forests	middle elevations (2400-2900 m), cool (75-90 frost-free days/yr), moist (60-75 cm/yr)	Ponderosa pine, Douglas-fir, white fir, blue spruce, southwestern white pine, trembling aspen, Gambel oak, snowberry, elksedge
Aspen forests	middle elevations (2400-3400 m), cool (55-90 frost-free days/yr), moist (60-90 cm/yr)	Trembling aspen, snowberry, Douglas-fir, white fir, subalpine fir, blue spruce, Engelmann spruce, snowberry, geranium, heartleaf arnica, meadow-rue
Spruce-fir forests	high elevations (2700-3600 m), cold (45-75 frost-free days/yr), wet (75-100 cm/yr)	Engelmann spruce, subalpine fir, whortleberry, geranium, strawberry, heartleaf arnica, daisy
Grasslands & meadows	all elevations and climatic zones; may be on drier sites than nearby forest vegetation	Big sagebrush, Arizona fescue, Thurber fescue, mountain muhly, Parry's oatgrass, other grasses & forbs
Riparian vegetation	permanently moist sites in all elevation and climatic zones	Cottonwoods, willows, thin-leaf alder, sedges
Lodgepole pine forests	high elevations; mostly on the northeastern flank of the San Juan & La Garita Mountains	Lodgepole pine, other species of spruce-fir and mixed conifer forests
Bristlecone pine woodlands	high elevations; mostly on the eastern sides of the San Juan, South San Juan, and La Garita Mountains	Bristlecone pine, some Engelmann spruce, xerophytic shrubs and forbs of spruce-fir forests

Silverton, Colorado, located at 2,800 m (9,300 ft) within a deep valley receiving cold air drainage from surrounding high peaks in the central San Juan Mountains, temperatures range from an average high of 23 ^{o} C (73 ^{o} F) in July to an average low of -18 ^{o} C (- 1 ^{o} F) in January. Mean annual precipitation may exceed 148 cm (60 in) per year on some of the highest peaks in Colorado, but is less than 25 cm (10 in) per year in portions of the San Juan Basin in northwestern New Mexico (Keen 1996;123). Seasonal distribution of precipitation also is uneven. Highest precipitation usually falls in late summer (July and August) and winter (January through March); May and June are usually relatively dry (Keen 1996). Even within one geographic area, there may be significant variation among sites, as illustrated by Figure I-2 which depicts monthly precipitation and temperature at six climatological stations on the southern and western flanks of the San Juans.

2. Long-Term Variability in Climate

The climate in the South Central Highlands Section is influenced not only by local effects of latitude and topography, but also by atmospheric processes occurring thousands of miles away in the Pacific and Atlantic Oceans. These teleconnections have received increasing attention during the past 20 years, and our understanding of how these far-away phenomena produce long-term variability in our local climate has improved greatly. The explanation below draws heavily from the summary provided by Kulakowski and Veblen (2006) for the Grand Mesa National Forest, to which the reader is referred for additional details and documentation.

One of the most important broad-scale influences on the climate of the southwestern U.S. is the El Niño – Southern Oscillation (ENSO), which results from anomalous water temperatures in the tropical Pacific Ocean. ENSO produces alternating periods of wet (El Niño) vs. dry (la Niña) winters and springs in the Southwest, with a periodicity of 2–6 years (Diaz and Kiladis 1992, Michaelsen and Thompson 1992, Woodhouse 1993, Swetnam and Betancourt 1998, Veblen et al. 2000). The Pacific Decadal Oscillation (PDO) and the Atlantic Multidecadal Oscillation (AMO) reflect variation in water temperatures in the northern Pacific and northern Atlantic Oceans, respectively, and both of these phenomena alternate between warm and cold phases with a periodicity of 20-30 years (PDO) or 60-80 years (AMO). The warm phase of PDO is associated with increased precipitation in the central and southern Rocky Mounains (Gray et al. 2003), but the warm phase of AMO is associated with drought in the western U.S. (McCabe et al. 2004). Severe drought in the Southwest occurs when the cold phase of PDO coincides with the warm phase of AMO, as occurred in the 1950s. Although the correlations between PDO/AMO phases and climate in the South Central Highlands Section are now well documented, the mechanisms involved are not yet well understood. Nevertheless, the correlations are important: for example, fire occurrence tends to increase throughout the region during drought periods associated with ENSO, PDO, and AMO.
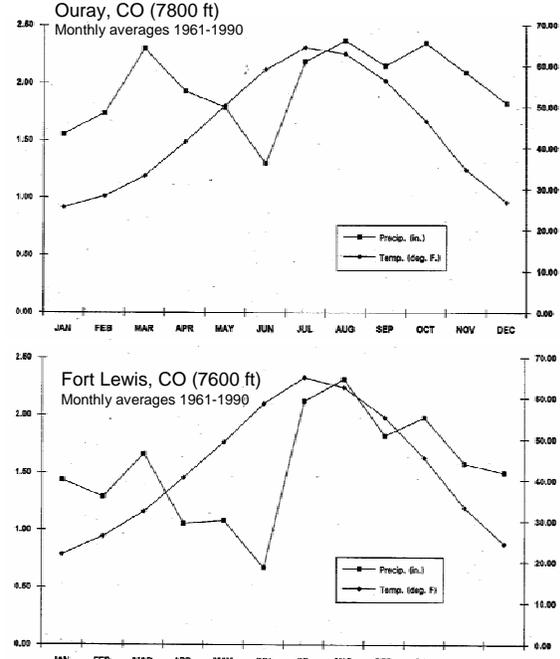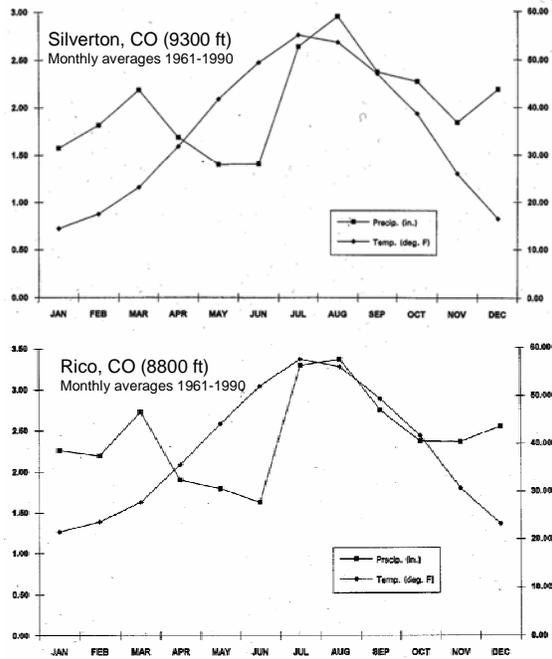
Figure I-2. Monthly temperature and precipitation at two high-elevation sites in the San Juan Mountains, CO.

Figure I-2 (cont.). Monthly temperature and precipitation at two mid-elevation sites in the San Juan Mountains, CO.
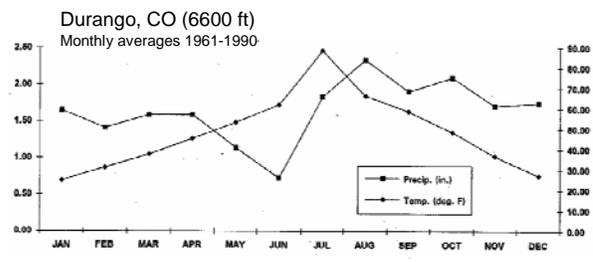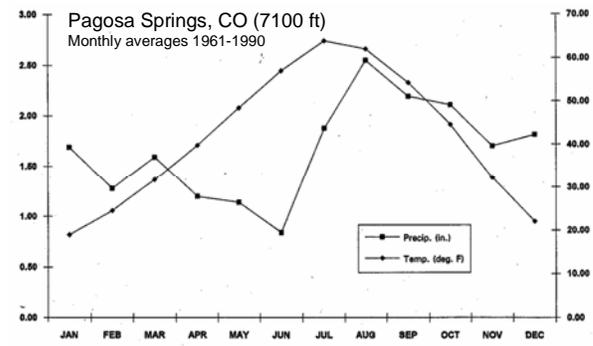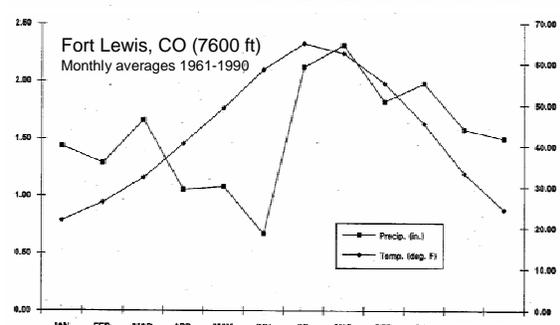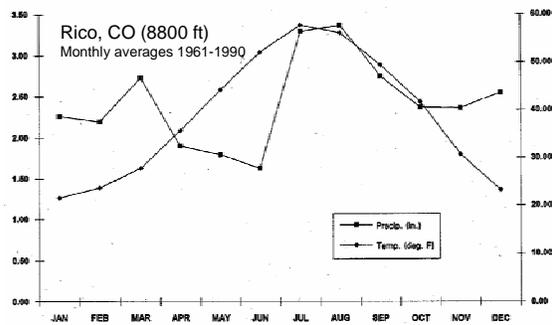
Figure I-2 (cont.). Monthly temperature and precipitation at two low-elevation sites in the San Juan Mountain country, CO.

Climatic fluctuation during the 20th century, associated in part with variation in ENSO, PDO, and AMO, can be seen in the record for Grand Junction, Colorado (Fig. I-3).

In addition to decadal-scale fluctuations, climatic variation occurs over centuries-long time scales. The "Little Ice Age," which occurred roughly from 1550 – 1850 A.D., was a generally colder and drier period in much of the world including the southern Rocky Mountains. (Note that Hughes and Funkhouser (1998) prefer to call this the "1600-1850 period of anomalous climate" because climatic changes were neither consistent nor synchronous from region to region.) The colder temperatures of the Little Ice Age were reflected in the advance of small mountain glaciers in northern Colorado

(Benedict 1973, Grove 1988), changes in fossil pollen preserved in ponds of southwestern Colorado (Fall 1988, Petersen 1994), and narrow tree-rings in many parts of the Southwest (Grissino-Mayer 1995, Figure I-4). In contrast, much of the 20th century has been characterized by warmer and wetter conditions. For example, the period from 1976-1995 was one of the wettest in the Southwest in the last thousand years (Grissino-Mayer 1995). In interpreting recent ecological changes in the South Central Highlands Section, it is important to remember that Euro-American settlement of this region coincided with the transition from colder, drier Little Ice Age conditions to the warmer, wetter climate of the 20th century.

D. REFERENCE CONDITIONS: CONCEPT AND APPLICATION

1. Historical Range of Variability
    Concept

To evaluate current ecological conditions in the South Central Highlands Section, we need a benchmark or reference period of relatively stability and ecological health to compare with the present (Leopold 1966). The "historical range of variability" (HRV) provides such a benchmark. HRV can be defined in a number of ways that all express the same basic idea, e.g.,
-- "the ecological conditions, and the spatial and temporal variation in these conditions, that are relatively unaffected by people, within a period of time and geographical area appropriate to an expressed goal" (Landres et al. 1999) …

-- "the estimated range of ecological conditions and processes that occurred in the past, often expressed as a probability distribution of likely states" … "a dynamic set of boundaries between which most native biodiversity variables have persisted–with fluctuations-through time and across space" (Binkley et al. 2007)

A number of synonyms for HRV express this same basic idea that we are concerned not with a static description of a single time in the past, but with a range of historical conditions in which the current biota developed. These synonyms include "range of natural variability," "natural range of variability," "reference variability," and "natural variability" (from Landres et al. 1999). For this report we use the term "historical range of variability" or HRV.
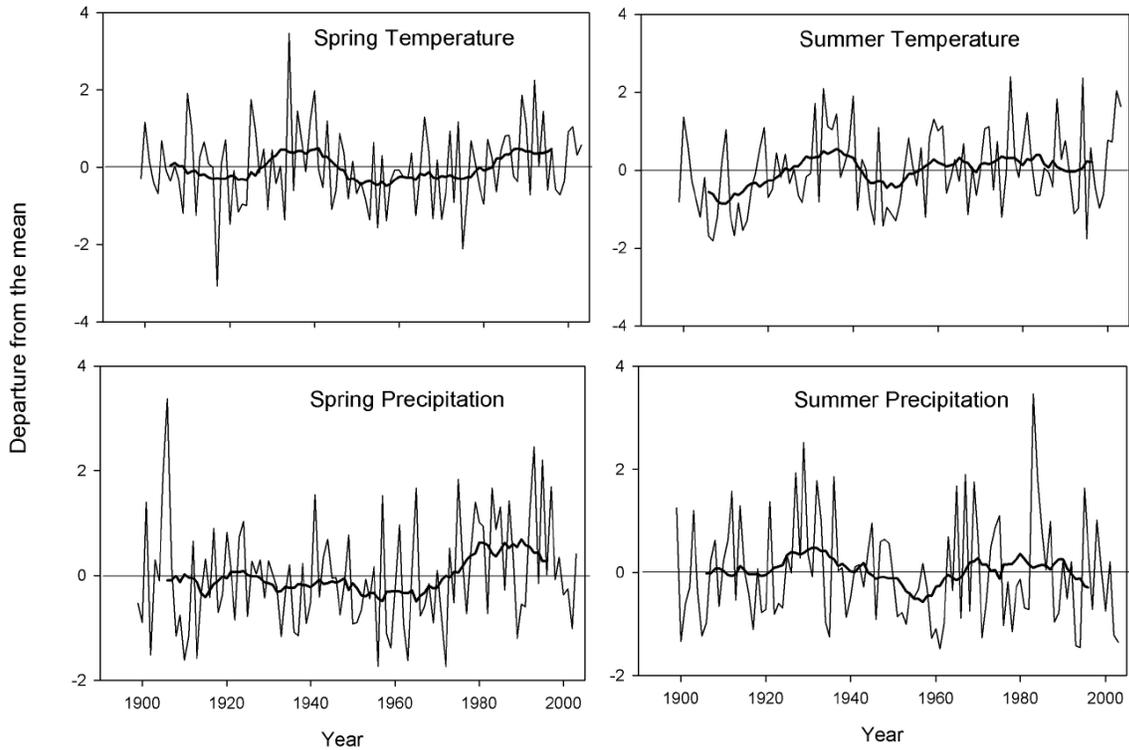
Figure I-3. Twentieth-century trends in spring and summer temperature and precipitation at Grand Junction, Colorado (copied from Kulakowski and Veblen 2006).
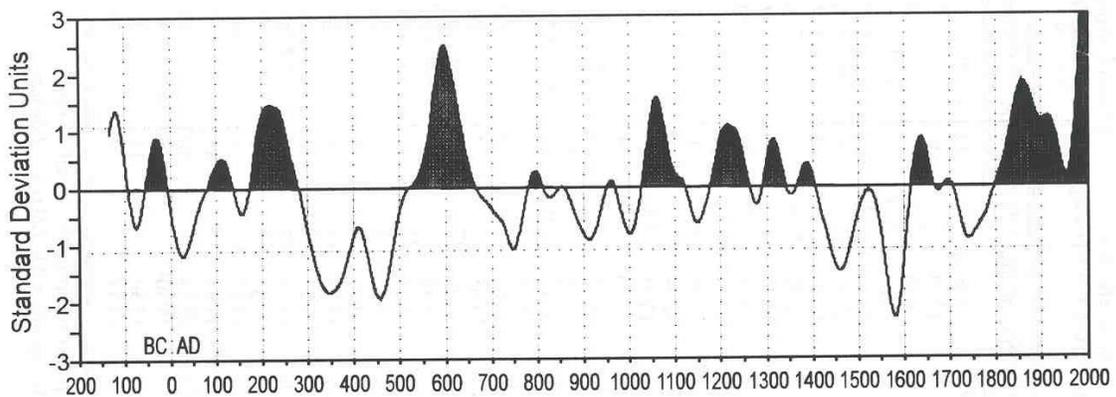
Figure I-4. Long-term variability in total winter precipitation as re-constructed from tree-rings in El Malpais National Monument, western New Mexico (H.D. Grissino-Mayer, personal communication 1995). Time periods depicted in black above the zero line were relatively wet (and the higher the line the wetter the climate), while time periods below the line were relatively dry (and the lower the line the drier the climate).

2. The Reference Period for this Analysis

There is no single "correct" reference period for an assessment of HRV. Different questions may require insights from different time periods in the past. It is also important to recognize that the San Juan Mountain country has never been static; there have been continual changes in the climate, geomorphology, vegetation, and human influences at least since the end of the Pleistocene ice age some 14,000 years ago. Thus, selection of a benchmark period of relative stability against which we can compare present ecological conditions is somewhat problematic (Lertzman and Fall 1998). Nevertheless, against the background of continual change, we can recognize several long periods characterized by relatively small-scale changes that were punctuated by brief but intense periods of rapid change in climate, human influences, or other factors.

Rapid and substantial global warming around 14,000 years ago brought about the end of the Pleistocene ice age and ushered in the Holocene period that has continued to the present. This huge climatic change resulted in drastic readjustments in the distribution and abundance of the biota. Cold-adapted plants and animals became extinct or migrated to the highest elevations while more warmth-tolerant species expanded out of local warm sites in the foothills or migrated into the region from outside (Pielou 1991). Since the end of the Pleistocene, however, the regional climate has been comparatively stable, though there have been important periods of smaller-scale variability -- such as the warmer and drier conditions of the altithermal (ca. 4000 years ago), the prolonged droughts that coincided with ancestral Pueblo people's abandonment of the region around 1300 AD, and the Little Ice Age that lasted from about 1600 AD until the early 1800s. Nevertheless, general vegetational zonation and species composition apparently have remained roughly the same throughout the latter half of the Holocene period (Petersen 1981, Betancourt 1990).

Humans have been present in the country now called the South Central Highlands for at least 11,000 years, and have influenced ecosystems throughout the time of their occupancy (Duke 1995, 1996; Paulson and Baker 2006). The extent and magnitude of human influences on local ecosystems during the Paleo-Indian and Archaic periods (prior to ca. 1 A.D.) is poorly understood, but people probably had substantially less ecological impact during that early period of apparently low population densities than they had in later periods. There probably was a dramatic change in the extent, magnitude, and kinds of human impacts during the Ancestral Pueblo (formerly called Anasazi) period in southwestern Colorado and northwestern New Mexico (ca 1 A.D. to 1300 A.D.), when human population densities reached levels comparable to those of today and agricultural and subsistence activities affected large extents of land area. Following abandonment of the region by ancestral Pueblo people around 1300 A.D., human populations apparently fell to relatively low levels in southwestern Colorado and northwestern New Mexico, and remained comparatively low until the arrival of large numbers of

Euro-American settlers in the middle to late 1800s.

Two periods of relative stability of climate, vegetation, and human influences potentially could provide a general reference period for this analysis of current ecological conditions in the South Central Highlands section: (i) the period between the end of the Ice Age and the rise of ancestral Pueblo culture, and (ii) the period from ancestral Pueblo abandonment to Euro-American settlement. The first of these periods is difficult to work with because of the paucity of specific ecological information available (the "fading record;" Swetnam et al. 1999). Therefore, we selected the second period (from about 1500 until the mid to late 1800s) as our principal reference, or benchmark, for evaluating the current ecological state of the South Central Highlands. We refer to this time as the *period of indigenous settlement*, in contrast to the period of Euro-American settlement that began in the mid to late 1800s.

It cannot be emphasized too strongly that the period of indigenous settlement was *not* a time of stasis, climatically, ecologically, or culturally (Paulson and Baker 2006). For example, the Little Ice Age occurred during this time, and was associated with small shifts in the position of the upper timberline and in the elevational breadth of the forest zone on the middle slopes of the mountains (Petersen 1981, Millar and Woolfenden 1999). Local human inhabitants obtained horses and new technology and were affected by disease and displacement of other tribes brought about by European colonization farther to the east (Whitney 1994). Nevertheless, compared with some other periods in history, the period from about 1500 - 1870 AD was a time of relatively consistent environmental and cultural conditions in this region -- and a time for which we have a reasonable amount of specific information.

It also is important to emphasize that our goal today should not be to attempt to recreate all of the ecological conditions of the 1500s through 1800s. Complete achievement of such a goal would be impossible, given the climatic, cultural, and ecological changes that have occurred in the last century. It also would be unacceptable socially, economically, and politically. Nor do we suggest that the reference period was completely "natural" or preferable in all ways to today's landscape. However, the period of indigenous settlement does provide a good benchmark for evaluating current conditions, because it appears to have been a time when the ecosystems of the South Central Highlands Section were intact and functioning well. These ecosystems apparently supported rich biodiversity, conserved soils and nutrients, and ran sustainably on solar energy (Kaufmann et al. 1994). Some of the ecological characteristics of the period of indigenous settlement have been lost to a greater or lesser degree in some portions of the South Central Highlands section, but there are opportunities to restore those ecological structures and processes that have been lost or altered -- should we decide to do so.

3. <u>Human Residents and Influences During the Reference Period</u>

During the period from the 1500s through the late 1800s, the major inhabitants of the San Juan Mountain country in southwestern Colorado apparently were the Ute people

(Callaway et al. 1986, Ellis 1996, Paulson and Baker 2006). The Navajo also may have used the area occasionally (Duke 1995). Little is known about the prehistory of the Utes; it is not even certain when they moved into the region (Duke 1995). However, the information that is available suggests that they were relatively few in number and scattered over an enormous area. Rockwell (1956) estimates that when Europeans arrived in Colorado there were only about 1,000 Utes living in southwestern Colorado, and that all seven bands of Utes throughout Colorado never comprised more than 10,000 people. Moreover, when Dominguez and Escalante traveled through southwestern Colorado in August of 1776, they never saw an Indian, which suggests that there were not many people living in the area although they simply may have all been up in the high country at that time (Bolton 1950). It is possible that Ute numbers were much higher before the late 1700s, but that they were decimated by disease as with many other North American tribes (Whitney 1994). However, there is no documented evidence either supporting or refuting this hypothesis.

The Ute people during this period were generalized hunters and gatherers who moved about frequently in response to changing resource availability (Callaway et al. 1986, Fowler 1986, Harper 1986, Duke 1995, Ellis 1996, Paulson and Baker 2006). Loosely organized bands tended to gather in the lower river valleys during the winter, but in the summer the population became widely scattered as individual family groups moved into the mountains to hunt, fish, and gather (Rockwell 1956, Aikens and Madsen 1986). Oral histories indicate that the Utes set fires on occasion (Marilyn Colyer, Mesa Verde National Park, personal communication 1996), but almost nothing is documented about specific Ute cultural practices with fire. Some tribes in the Great Basin region are known to have ignited fires to favor wild tobacco and seed crops (Harper 1986), and indigenous people elsewhere in North America burned forests and grasslands to improve forage for game animals (Barrett and Arno 1982, Whitney 1994, Vale 2002). However, some early settlers in southwestern Colorado reported that the Utes did not use fire to as great an extent as some other tribes (Anonymous 1971;122). In short, we simply do not know enough about the Utes' former hunting, gathering, and burning practices to allow any definitive conclusions about their impact on local ecosystems. Given a low population, widely dispersed over a huge area, it seems likely that Ute subsistence would have had relatively little influence on plant and animal populations and fire regimes, except locally in areas near popular encampments.

The magnitude and extent of indigenous peoples' influences on the ecosystems of the South Central Highlands Section during the reference period is one of the big questions about this region that requires more research. In a pioneering assessment of this general issue, Whitney (1994) summarized the available information on effects of indigenous peoples in eastern North America prior to European settlement. He concluded that people significantly altered ecosystems in certain areas (e.g., coastal regions, major river valleys, around the Great Lakes), but that there were extensive areas where human impact was minimal (e.g.,

northern New England, portions of the Allegheny Plateau). Baker (2002) and Vale (2002) recently suggested a similar interpretation for western North America, and Paulson and Baker (2006) concluded the same for southwestern Colorado. We expect that the situation was generally similar throughout the South Central Highlands Section -- local sites of intense human effects and extensive areas with little human influence -- but we emphasize that our current understanding of the human occupants of this region during the 1500 - 1800 period is inadequate.

Given the probably low population densities of the Native American inhabitants of the South Central Highlands Section during the period of indigenous settlement, and especially considering the paucity of specific information regarding their

effects on ecosystems of the region, this report describes reference conditions primarily in terms of climatic and biological processes and influences. It is important to recognize, however, that the human history and cultures in the southernmost portion of the South Central Highlands Section were different in important ways from those of the northern portion of the Section during the period of indigenous settlement. Notably, the Jemez and Zia Pueblos, as well as several other pueblos along the Rio Grande, were permanent settlements throughout this period, and their occupants made use of adjacent highland areas. The interactions of these more southerly inhabitants with their local physical and biological systems are not treated in any detail in this report, but see Scurlock (1998) and Allen (2002) for more on this topic.

E. OVERVIEW OF INTEGRATED ECOSYSTEM MANAGEMENT

The general approach and specific analyses of this report were undertaken partly in response to recent changes in the direction of U. S. Forest Service management across the nation (e.g., Kessler et al. 1992, Kaufmann et al. 1994, Haynes et al. 1996, Quigley et al. 1996, Kaufmann et al. 1998).

As Vogt et al. (1997;1) put it:
-- "The management of natural resources has entered a period of unprecedented change and uncertainty. Our former paradigm, which stressed land allocation measures, maximum sustained yield principles, and multiple-use objectives, is being rapidly replaced by a new paradigm, which emphasizes *sustainable ecosystems* rather than *sustainable yield*.

This new paradigm, termed *ecosystem management*, focuses on management of "whole systems" for a variety of purposes, rather than simply focusing on commodity production for a single resource."

According to Kaufmann et al. (1994;3):
-- "Our guiding premise for sustaining ecosystems and protecting biodiversity now and into the future is to manage ecosystems such that structure, composition, and function of all elements, including their frequency, distribution, and natural extinction, are conserved."

Christensen et al. (1996;668-669) defined ecosystem management as

-- "... management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem structure and function."

Table I-2 provides several other characterizations of ecosystem management.

Table I-2. Characterizations of ecosystem management.

A. Some elements of ecosystem management (Christensen et al. 1996;669-670):
     (1) Long-term sustainability as a fundamental value.
     (2) Clear operational goals.
     (3) Sound ecological models and understanding.
     (4) Understanding complexity and interconnectedness.
     (5) Recognition of the dynamic character of ecosystems.
     (6) Attention to context and scale.
     (7) Acknowledgment of humans as ecosystem components.
     (8) Commitment to adaptability and accountability.

B. Some principles of effective or "good" ecosystem management (Vogt et al. 1997;101-102):
     (1) Spatial and temporal scales are critical.
     (2) Limits to system capacity are considered.
     (3) Ecosystem managers look for the root of the problem, rather than treating one symptom at a time.
     (4) Social constraints are considered.
     (5) Science is used to inform decisions.
     (6) System function as well as structure is investigated.

C. Ten guiding principles for sustaining biodiversity, ecosystem function, and ecological resilience while producing commodities from natural landscapes (Fischer et al. 2006):
     (1) Maintain and create large, structurally complex patches of native vegetation.
     (2) Maintain structural complexity throughout the landscape.
     (3) Create buffers around sensitive areas.
     (4) Maintain or create corridors and stepping stones.
     (5) Maintain landscape heterogeneity and capture environmental gradients.
     (6) Maintain key species interactions and functional diversity.
     (7) Apply appropriate disturbance regimes.
     (8) Control aggressive, over-abundant, and invasive species.
     (9) Minimize threatening ecosystem-specific processes.
     (10) Maintain species of particular concern.

It also is instructive to review some of the things that ecosystem management is *not*. It is not focused on maximizing the abundance or production of a single species or a single commodity (Franklin 1989). Rather, ecosystem management seeks to sustain a variety of "ecosystem goods and services "(Christensen et al. 1996;667), as illustrated in Table I-3. However, ecosystem management is not merely the old policy of multiple use with a new name. Ecosystem management explicitly acknowledges that not everything can be maximized on one piece of ground, because of inherent limits within ecosystems and the presence of strong feedbacks and interactions among the components and processes of all ecosystems.

Table I-3. Examples of ecosystem goods and services (from Christensen et al. 1996;667). "Goods" are things that have monetary value in the marketplace, whereas "services" are valued by society but rarely bought and sold.

Ecosystem goods:

    -- food
    -- construction materials
    -- medicinal plants
    -- wild genes for domestic plants and animals
    -- tourism and recreation

Ecosystem services:

    -- maintaining hydrological cycles
    -- regulating climate
    -- cleansing water and air
    -- maintaining the gaseous composition of the atmosphere
    -- pollinating crops and other important plants
    -- generating and maintaining soils
    -- storing and cycling essential nutrients
    -- absorbing and detoxifying pollutants
    -- providing beauty, inspiration, and research

Whereas the old multiple use idea suggested that everyone's desired goods and services could be maximized by effective management, ecosystem management recognizes that tradeoffs are inevitable and that some sacrifices will have to be made (Vogt et al. 1997;369). Finally, ecosystem management emphasizes neither a purely anthropocentric nor a purely biocentric approach to managing public lands. Human values and choices are central to ecosystem management, but so are the inherent limits of the ecosystems being

managed. Ecosystem management "acknowledges the importance of human needs while at the same time confronting the reality that the capacity of our world to meet those needs in perpetuity has limits and depends on the functioning of ecosystems" (Christensen et al. 1996;666).

Additional descriptions of some of the new approaches to forest management being carried out in the spirit of ecosystem management, adaptive management, and emulating natural disturbance processes, are provided in books by Kohm and Franklin (1997), Perera et al. (2004, and Stanturf and Madsen (2005).

Closely related to the idea of ecosystem management is rapidly evolving practice of ecological restoration. The term "restoration" is often used in a loose sense to refer to any desirable change or management activity on a landscape, but in fact it has a precise meaning as defined by the Society for Ecological Restoration (http://www.ser.org/):

-- "Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed."

-- "Ecological restoration is an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability."

-- "Restoration attempts to return an ecosystem to its historic trajectory."

-- "Restoration represents an indefinitely long-term commitment of lands and resources, and a proposal to restore an ecosystem requires thoughtful deliberation."

Recent essays by Veblen (2003) and by Brown (2004) have clarified the distinctions between genuine ecological restoration and other management objectives such as fuel reduction and wildfire mitigation. Great advances have been made in the past decade in restoration of southwestern ponderosa pine forests, as summarized in the recent book by Friederici (2003), which also contains a case study of forest restoration on the San Juan National Forest (Romme et al. 2003).

This report constitutes only a portion of the larger ecosystem management approach, i.e., we provide an assessment of current ecological conditions in relation to the historic bounds of variability in the ecosystems and landscapes of the South Central Highlands Section. Although our assessment is based on a considerable body of research and analysis, we must acknowledge at the outset that there still is substantial uncertainty about many of the ecological characteristics and processes that we address. However, uncertainty is the rule, not the exception, in efforts to understand and deal with complex systems of this kind (Walters 1986, Christensen et al. 1996;676). Therefore, it is essential that managers embrace this uncertainty, and have procedures and policies in place to deal with unexpected events and new information -- i.e., to practice adaptive management (e.g., Walters 1986).

There can never be enough scientific research to allow certain prediction of the consequences of management actions before those actions occur, although often we can make reasonable predictions about the probability of certain possible outcomes (e.g., Bayesian analysis; Walters 1986, Vogt et al. 1997). However, management itself can be viewed as one of our best sources of new scientific understanding, since each management

action (e.g., logging a forest or closing a road) is in effect an experiment from which we can learn if we ask the right questions in advance and do the right kinds of monitoring (Walters 1986, Vogt et al. 1997). Even if management experiments turn out badly—i.e., if the results of an action have unexpectedly negative consequences--we still can learn from the experience if the action was carried out in an explicitly experimental framework. Writing about ecosystem restoration, Ewel (1987;31) made a comment that applies equally well to our attempts at ecosystem management:

-- "Ecosystem restoration is an activity at which everyone wins: when successful, we are rewarded by having returned a fragment of the earth's surface to its former state; when we fail, we learn an immense amount about how ecosystems work, provided we are able to determine why the failure occurred ... The ecologist capable of creating [a restored] ecosystem that [meets certain rigorous criteria] earns high marks; the one who fails is sure to gain new insights into ecosystem structure and function."

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CHAPTER II: PONDEROSA PINE FORESTS

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A. VEGETATION STRUCTURE AND COMPOSITION

Ponderosa pine forests are found at lower to middle elevations throughout the South Central Highlands Section (Figure I-1), just above the piñon-juniper zone, in the foothills of the major mountain ranges, and on broad mesas and plateaus such as the Uncompahgre Plateau and the Glade area north of Dolores, Colorado. They also cover a large area on the extensive lava and ash flows of the Jemez Mountains. The ponderosa pine forests in the South Central Highlands Section are characterized by cold winters and a large proportion of the annual precipitation falling during the high sun period (April - September). In the San Juan National Forest, ponderosa pine forests are found at elevations of 2,100 - 2,600 m (6,900 - 8,600 feet), and they occupy about 16% of the Forest. Mean annual precipitation is about 45 - 70 cm (18 - 28 in), and the frost-free period is about 70 - 105 days each year. These forests are associated with the frigid soil temperature regime and the ustic soil moisture regime. The ponderosa pine forest type is widespread in western North America, with generally similar climatic conditions throughout the region (Moir et al. 1997; but see Pearson 1951). Within this broad category of ponderosa pine forest climates (including areas outside the South Central Highlands Section), mean annual air temperatures range from 4 - 7 $^{\circ}$ C (39 – 45 $^{\circ}$ F), and mean annual precipitation ranges from 52 - 66 mm (20 – 26 in) (Moir et al. 1997).

Ponderosa pine forests in the South Central Highlands Section are generally dominated almost exclusively by ponderosa pine trees (*Pinus ponderosa* var. *scopulorum*). Rocky Mountain juniper (*Juniperus scopulorum*) is often present, usually in low densities. Gambel oak (*Quercus gambelii*) is an understory dominant in many of these forests. Other common shrub species in ponderosa pine forests of the South Central Highlands Section include mountain mahogany (*Cercocarpus montanus*), serviceberry (*Amelanchier utahensis* and *A. alnifolia*), buckbrush (*Ceanothus fendleri*), bitterbrush (*Purshia tridentata*), and snowberry (*Symphoricarpos rotundifolia*). The ground layer vegetation of ponderosa pine forests is extremely variable, depending on aspect, soil type, overstory canopy cover, grazing and fire history. Common ground layer species include the graminoids *Carex geyeri*, *Danthonia parryi*, *Elymus elymoides, Festuca arizonica, Koelaria macrantha, Muhlenberia montana, Poa fendleriana, P. pratensis*, and the forbs *Achillea lanulosa, Antennaria rosea, Arctostaphylos uva-ursi, Erigeron formosissimus, Fragaria virginiana, Geranium caespitosum, Lathyrus leucanthus, Mahonia repens, Penstemon barbatus, Potentilla hippiana, Pseudocymopterus montanus, Pulsatilla patens,* and *Solidago simplex.*

Elements of the nearby piñon-juniper woodlands may be present in low densities at the lower elevations and in drier or warmer areas within ponderosa pine forests, e.g., Colorado piñon (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*). White fir (*Abies concolor*) and Douglas-fir (*Pseudotsuga menziesii*), elements of the nearby warm, dry mixed conifer forests, may be present in low densities at the higher elevations and in moister or cooler areas within ponderosa pine forests. Quaking aspen (*Populus tremuloides*) also grows with ponderosa pine in some places. The remainder of this chapter focuses on forests where ponderosa pine is the only major tree species present. The warm, dry mixed conifer type, where ponderosa pine may be a substantial component, is discussed in chapter V (mixed conifer forests), and aspen forests are discussed in chapter IV. Ponderosa pine forests of the South Central Highlands Section are associated with the *Pinus ponderosa* series, and the *Pinus ponderosa/Quercus gambelii* and *Pinus ponderosa/Festuca arizonica* plant associations (potential natural vegetation types) of DeVelice et al. (1986).

Although a single species -- ponderosa pine -- dominates the extensive lower-elevation forests of the South Central Highlands Section, there is considerable genetic differentiation among sub-populations of ponderosa pine, and probably within the major shrub and herbaceous species as well. For example, Linhart (1988) and colleagues have demonstrated significant genotypic differences along elevational gradients and even between adjacent slopes having different aspects. In this report we will deal mainly with the importance of structural heterogeneity in ponderosa pine forests, e.g., variation in tree sizes, densities, and spatial aggregation, as well as spatial and temporal variability in disturbance history (see below). However, although we do not treat genetics per se in our analysis, it is important to acknowledge that ponderosa pine and its associated species also display a great deal of genetic heterogeneity and fine-scale adaptation to local environmental conditions and history. One of the goals of ecosystem management is to conserve this kind of within-species genetic diversity (Kaufmann et al. 1994).

B. REFERENCE CONDITIONS

The reference period for this analysis is the two to three centuries prior to about 1870. Localized mining began in the San Juan Mountains in the 1860s (Smith 1996) and Hispanic settlers were present in portions of the Jemez Mountains as early as the mid-1700s (Touchan et al. 1996), but extensive Euro-American settlement began around 1870 in most of the South Central Highlands Section. After 1870, many or most of the ponderosa pine forests in the region were altered

dramatically by livestock grazing, logging, and fire suppression by Euro-American settlers, as discussed in the section on legacies of past human land use (below). The most important natural disturbance process in ponderosa pine forests during the reference period was fire; consequently this section emphasizes variability in fire regimes. Other somewhat less important disturbances included various tree-killing insects (notably bark beetles), pathogens, and parasites. A severe and

prolonged drought in the late 1500s also probably killed great numbers of ponderosa pine and other tree species (Swetnam and Brown 1992, Brown and Wu 2005).

1. Reference Period Fire Regimes in Ponderosa Pine Forests

Historical fire regimes of forests in the South Central Highlands Section were variable through time and space, and can be best understood when viewed in a broad context of latitudinal gradients in ponderosa pine forests throughout the Southwest and Rocky Mountain regions. Forests dominated by ponderosa pine are found from southern Arizona and New Mexico to the Black Hills and British Columbia. Despite the similarity in canopy composition, understory characteristics and local climatic conditions differ substantially along this latitudinal gradient, and historical fire regimes varied along the same gradient.

Ponderosa pine forests in many parts of the Southwest (Arizona and New Mexico) were typically composed of open stands of large trees and grass-dominated understories. The southwestern climate is characterized by a dry period of several weeks in late spring and early summer, followed by a rainy late summer and fall. This fuel structure and climate, combined with frequent ignitions by lightning and indigenous peoples (Allen 2002), resulted in a fire regime dominated by frequent, low-severity fires—referred to below as the "Southwestern model" (e.g., Weaver 1951, Dieterich and Swetnam 1984, Covington and Moore 1994, and Fule et al. 1997, 2002, in northern Arizona … Weaver 1951, Cooper 1960, Baisan and Swetnam 1990 in east-central and southern Arizona …

Savage and Swetnam 1990, Savage 1991 in northwestern New Mexico and northeastern Arizona … Grissino-Mayer 1995 in west-central New Mexico … Swetnam and Baisan (1996), Touchan et al. 1996, Allen 2002 in the Jemez Mountains of northern New Mexico). These fires rarely killed canopy trees, but consumed grass, dead leaves, and dead woody material, and killed small pines and the above-ground portions of shrubs and herbs, thereby periodically thinning the understory and maintaining open stand conditions. Shrubs and herbs responded by rapidly re-sprouting from surviving below-ground structures, and pine seedlings became established in patches of mineral soil exposed by the fire (Cooper 1960, 1961; White 1984; Covington and Moore 1994).

In contrast, ponderosa pine forests in many parts of the central and northern Rocky Mountains have shrub-dominated or generally sparse herbaceous understories. The climate in these regions is characterized by a typically wet spring and early summer followed by a dry late summer and fall. Ignitions by lightning and humans prior to 1870 were somewhat less frequent than in the Southwest (Baker 2002). The result was less frequent fire in the north (Brown and Shepperd 2001), but more variable fire severity—including an important component of high-severity (stand-replacing) fire—referred to below as the "variable-severity model." This complex, variable-severity fire regime has been documented notably in the Colorado Front Range (e.g., Brown et al. 1999, Veblen et al. 2000, Ehle and Baker 2003, Kaufmann et al. 2006, Baker et al. 2006). Low-severity fires that thinned the understory but did not kill the canopy occurred throughout the ponderosa pine zone, and were the

dominant fire type primarily at lower elevations (Sherriff 2004). Less frequent but high-severity fires that killed most of the canopy were especially important on wetter sites and at higher elevations within the ponderosa pine zone (Sherriff 2004). Individual fires often burned at variable severity, i.e., as a stand-replacing crown fire during periods of low humidity and high wind, then as a low-severity surface fire when humidity rose and winds dropped (Kaufmann et al. 2006). Only about 20% of the ponderosa pine zone in Boulder County, Colorado, was characterized by a fire regime of predominantly low-severity fires; the remainder of the area had a mix of low-severity and high-severity fires (Sherriff 2004).

In addition to the geographic gradient just described, fire regimes varied with climatic fluctuations. It is becoming increasingly clear that continental-scale climatic processes and hemispheric teleconnections have important influences on local fire regimes in western North America. In the southwestern U.S. (including the South Central Highlands Section), for example, fires are more numerous and burn greater areas during la Nina episodes than during el Nino episodes (Swetnam and Betancourt 1990, 1998). Recent research in ponderosa pine forests of northern Colorado and Idaho has shown that large erosion events, producing mudflows laden with charcoal and therefore indicating severe fires in the watershed, have occurred during warm periods of the past millennium, whereas cool periods have produced few or no such erosion events, apparently because fires were less severe during those periods (Elliott and Parker 2001, Pierce et al. 2004, Whitlock 2004).

The South Central Highlands Section is located geographically between the Southwest and the Colorado Front Range, and consequently the historical fire regimes in this region likely include elements of both the Southwestern model and the variable-severity model. The Southwestern model of frequent, low-severity fires fits best in the southern extremity of the South Central Highlands Section, notably in the Jemez Mountains where the climate is strongly southwestern and many ponderosa pine understories are grass-dominated. In the Rio Frijoles watershed on the eastern flanks of the Jemez Mountains, for example, composite mean fire intervals at the stand level ranged from 5-18 years during the period 1598-1899, and extensive fires burned a large portion of the watershed at intervals of 5-16 years (Allen 1989, 2002, Baisan and Swetnam 1996, Touchan et al. 1996). However, historical fire regimes probably had more of a variable-severity component in the northern portions of the South Central Highlands Section, notably in the San Juan Mountains and the Uncompahgre Plateau. Climate-wise, the San Juans and Uncompahgre Plateau are quite similar to the Southwest, with a pronounced dry period and frequent ignitions in early summer; but understories are more similar to those farther north. The most important understory characteristic of ponderosa pine forests in the northern portion of the South Central Highlands Section is the abundance of tall shrubs, especially Gambel oak. Oak foliage dries more slowly than grasses, meaning that longer dry periods are needed to promote extensive fire spread, but once ignited the oak foliage releases a great amount of heat and produces high flame lengths

that can reach into the ponderosa pine canopy and produce a crown fire, especially under high-wind conditions. It is sometimes suggested that oak was not formerly abundant in the San Juan country, but has increased unnaturally during the past century because of past fire exclusion. However, available evidence refutes this notion; oak was clearly an important component of these forests prior to 1870 (see the section below entitled "Pre-1870 stand structures in ponderosa pine forests" for details).

We have considerable fire history data for ponderosa pine forests in the San Juan Mountains and Uncompahgre Plateau, based mainly on fire-scar analysis, which primarily reveals the low-severity component of the historical fire regime (Brown and Shepperd 2001, Grissino-Mayer et al. 2004, Brown and Wu 2005). Investigation of the high-severity component of the historical fire regime requires both sampling of fire scars and detailed analysis of tree age structures to detect cohorts of trees that established following a high-severity fire, or stands in which maximum tree ages are much younger than the potential life span of the species but coincide with a documented fire event (e.g., Baker et al. 2006). Brown and Wu (2005) conducted this kind of analysis in a 307-ha unlogged ponderosa pine stand on Archuleta Mesa near Pagosa Springs, and concluded that there was little evidence of extensive high-severity fire; this interpretation was based on the abundance of old trees and fire scars, plus a wide spectrum of canopy tree ages,. They did detect pulses of tree establishment in the early 1600s, early 1700s, and mid 1800s, but interpreted these cohorts as representing improved survival during periods of relatively wet climatic conditions and infrequent fires rather than even-aged recruitment following high-severity fire (cf. Baker et al. 2006).

Additional investigations of this kind are needed in other parts of the South Central Highlands Section before we can confidently determine the importance, or lack of importance, of high-severity fire in ponderosa pine forests during the pre-1870 reference period. Considering the climatic and fuels characteristics of the San Juan region, we hypothesize that the historical fire regime in ponderosa pine forests of this portion of the South Central Highlands Section was a mixed-severity model of some kind, roughly intermediate between the fire regimes found to the north and south. Fire regimes in ponderosa pine of southwestern Colorado could have resembled the fire regimes of the Colorado Front Range, where high-severity fires were especially important, because of the fuel structure created by Gambel oak and other shrubs in the understory. Fire regimes also could have resembled those of Arizona and New Mexico, with an important element of low-severity surface fire, because of the southwestern climate (notably the early summer dry period) and relatively abundant native bunchgrasses along with the shrubs in the understory (this bunchgrass component has been diminished in many stands since EuroAmerican settlement, as described in the section below). We stress, however, that the importance (or lack of importance) of high-severity fires in ponderosa pine forests of the San Juan Mountains and Uncompahgre Plateau has not yet been investigated adequately.

In the sections below, we summarize fire frequency and

seasonality for ponderosa pine forests in the San Juan National Forest during the two to three centuries prior to EuroAmerican settlement in the region, based on the fire scar analyses of Grissino-Mayer et al. (2004). Although we have no information on the overall severity of the fires documented in this study, we suspect that many were of variable severity, especially those that burned in very dry years and were documented in multiple locations.

Fire Intervals:

Intervals between successive fires varied greatly, both through time and across the landscape (Grissino-Mayer et al. 2004). Fire history was reconstructed at nine sites in the San Juan National Forest, representing a wide range of environmental conditions throughout the zone where ponderosa pine is an important species (Figure II-1, Table II-1). The sites were not selected at random, but were chosen to represent a broad elevational gradient across the full extent of the San Juan National Forest. They also were located in places where fire scars could be found—thus they emphasize the surface fire component of the fire regime as noted above. Each site was approximately 50 ha in extent. The sites are arranged in Table II-1 in order from lowest elevation to highest elevation. The first six sites listed are dominated by ponderosa pine, and are discussed here. The last three listed are more appropriately regarded as mixed conifer stands, although they do contain ponderosa pine, and are discussed later in chapter V. Figure II-2 presents the master fire chronology, i.e., a compilation of all fire years detected in any of the nine study sites in the San Juan National Forest. This fire history is based on nearly 200 individual fire scar samples -- one of the larger collections of fire scars anywhere in the southwest.

Fire interval statistics for the period of indigenous settlement are presented in Table II-2. These values are derived from the Weibull distribution, a fire history model that does not require normally distributed data and is being used increasingly for comparisons of fire history among different areas (Johnson and Van Wagner 1985, Johnson and Gutsell 1994, Finney 1995, Grissino-Mayer 1995). The " *> 95% interval* " is defined as the fire interval that is exceeded by 95% of documented intervals between successive fires; smaller intervals can be regarded as significantly short. The " *median interval* " is the midpoint in documented fire intervals: half of the intervals are shorter, half are longer. The " *> 5% interval* " is the fire interval that is exceeded by only 5% of documented fire intervals; larger intervals can be considered significantly long. The table lists two kinds of fire interval statistics: those that describe intervals between successive fires of any size (i.e., that scarred one or more trees anywhere within the study area of ca 1 km^2), and those that describe intervals between relatively extensive fires that scarred at least 50% of the recorder trees within the area (recorder trees are those that were alive at the time of a fire and had already been scarred at least once, hence were capable of recording the fire event). The statistics for the extensive fires (> 50% scarred) probably give the best picture of how frequently fire re-visited a stand as a whole, since the statistics developed from all fires include tiny fires that may have gone out before burning more than a single tree or a few square meters (Baker and Ehle 2001).

Table II-1. Characteristics of sites where fire history was reconstructed in the San Juan National Forest (Grissino-Mayer et al. 2004). See Figure II-2 for locations.

Site	Ranger District	Elevation (feet)	Size of Area	Topography	Aspect	Substrate	Vegetation
Five Pine Canyon (FPC)	Dolores	7400-7800	200 ha	gentle slopes & shallow ravines	SW	Dakota sandstone	Pipo/Quga at lower ecotone with piñon-juniper & mountain shrub
Plateau Creek (PLT)	Dolores	7900-8000	150 ha	gentle slopes, nearly flat	N	Dakota sandstone with some shale or loess	pure Pipo overstory; sparse Quga, high cover of grasses & forbs (inc. Fear)
Turkey Springs (TRS)	Pagosa	8000-8100	200 ha	gentle slopes, nearly flat	N	Dakota sandstone with some shale or loess	pure Pipo overstory; sparse Quga, high cover of grasses & forbs (inc. Fear)
Hermosa Creek (HCK)	Colum-Bine	7900-8400	50 ha	steep slopes & deep ravines	SW	Pennsyl-vanian sandstone, limestone, & shale	Pipo/Quga on southerly aspects, Pipo-Psme / Quga-Syor on northerly aspects
Benson Creek (BCK)	Pagosa	8000-8400	200 ha	moderate slopes & ravines	W	Dakota sandstone (?)	Pipo/Quga
Smoothing Iron (SMI)	Dolores	8100-8400	200 ha	broad ridgetop	SW	Dakota sandstone	Pipo/Quga
Monument (MNT)	Pagosa	8100-8400	200 ha	moderate slopes & ravines	E	Dakota sandstone with some shale (?)	Mixed conifer: Abco, Psme, Pipo, Potr ... diverse shrubs & herbs
Taylor Creek (TCK)	Dolores	8300-8700	100 ha	steep slopes & deep ravines	S	Jurassic siltstone, sandstone, limestone, & shale	Mixed conifer: Psme, Pipo, Potr ... diverse shrubs & herbs
Burnette Canyon (BCN)	Dolores	9000-9300	50 ha	narrow ridge & adjacent slopes	N	Triassic siltstone & sand-stone	Mixed conifer: Psme, Abla, Pipo, Potr ... diverse shrubs & herbs

Table II-2 Fire intervals during the pre-1870 reference period in ponderosa pine forests of the San Juan National Forest. The " > 95% interval " is defined as the fire interval that is exceeded by 95% of documented intervals between successive fires; smaller intervals can be regarded as significantly short. The " median interval " is the midpoint in documented fire intervals: half of the intervals are shorter, half are longer. The " > 5% interval " is the fire interval that is exceeded by only 5% of documented fire intervals; larger intervals are significantly long. The time period selected includes the period when there were 6 or more "recorder" trees (living trees susceptible to fire-scarring) at the site, and therefore varies somewhat among sites.   The last three sites (Monument, Taylor Creek, and Burnette Canyon) are mixed conifer stands in which ponderosa pine is co-dominant with other tree species. See Table II-1 for a description of each site.

| SITE | ---------- ALL FIRES -------- | | | - FIRES THAT SCARRED 50% - OR MORE OF RECORDERS | | | |
	> 95% interval	Median interval	> 5% interval	> 95% interval	Median interval	> 5% interval	PERIOD
Five Pine Canyon	2	6	12	5	13	23	1748-1864
Plateau	2	6	12	5	12	20	1685-1872
Turkey Springs	3	10	21	7	18	33	1748-1880
Hermosa	2	10	25	9	17	25	1715-1890
Benson Creek	2	7	14	6	13	21	1767-1889
Smoothing Iron	2	10	31	14	30	47	1729-1879
Monument	3	18	47	4	21	59	1654-1880*
Taylor Creek	5	18	39	14	22	29	1786-1879
Burnette Canyon	5	28	73	4	27	82	1729-1880*

* During portions of the early part of this period, there were < 6 recorder trees. It was necessary to include this portion of the record to obtain a *statistically* adequate sample for analysis.
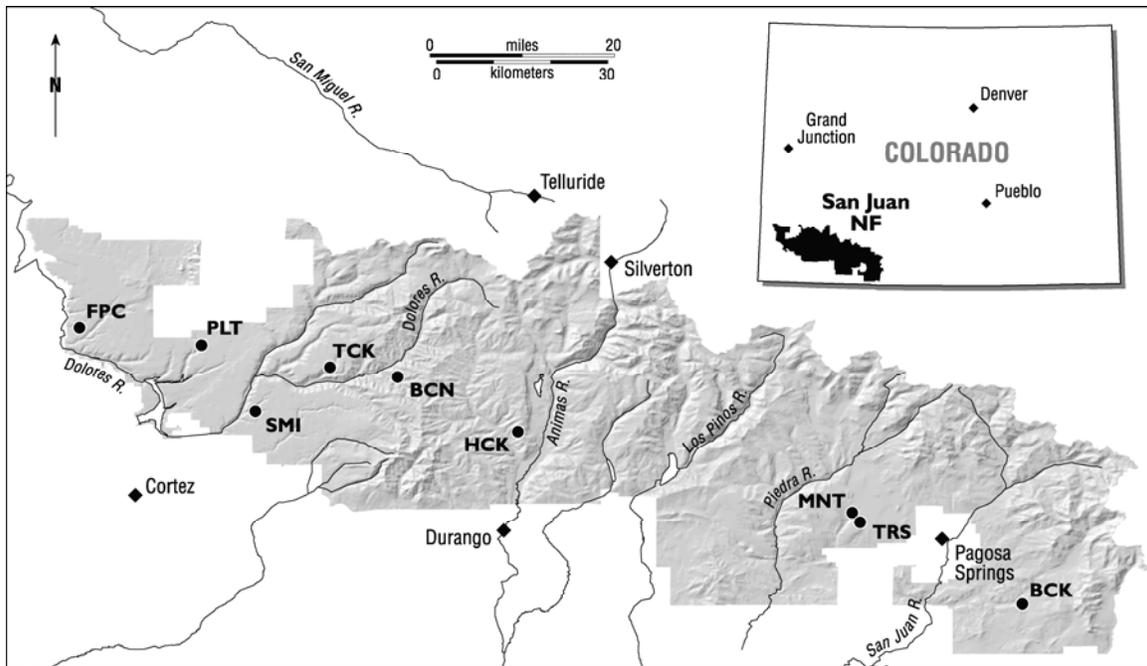
Figure II-1. Locations of fire history study sites in the San Juan National Forest (Grissino-Mayer et al. 2004). See Table II-1 for a description of each site.

Minimum, median, and maximum fire intervals during the reference period generally increased with increasing elevation. Three sites at lower elevations with nearly pure stands of ponderosa pine (Five Pine Canyon, Plateau, and Turkey Springs) had extensive fires every 5 - 33 years ("extensive" fires are those that scarred at least 50% of the recorders, Table II-2). The median interval between extensive fires at these sites ranged from 12 - 18 years. Three sites at somewhat higher elevations with predominantly ponderosa pine but also some Douglas-fir (Hermosa, Benson Creek, and Smoothing Iron) had extensive fires every 6 - 47 years, with median fire intervals of 13 - 30 years.

Fire intervals for fires scarring at least 50% of recorders at the Smoothing Iron site were approximately twice as long as the intervals at the other ponderosa pine sites, even though the elevation of the Smoothing Iron site is about the same as the others. This may be because of the topographic position of the Smoothing Iron site: it lies atop a narrow plateau with steep, poorly vegetated slopes to the west and south. These slopes would retard fires spreading into the area from the west or south, so to burn the Smoothing Iron site fires either would have to be ignited within the site itself or spread from the east or north, against the prevailing winds. In contrast, all of the other five ponderosa pine sites in Table II-2 have no topographic barriers to fire spread, and probably were affected by extensive fires that were ignited some distance away from the site.
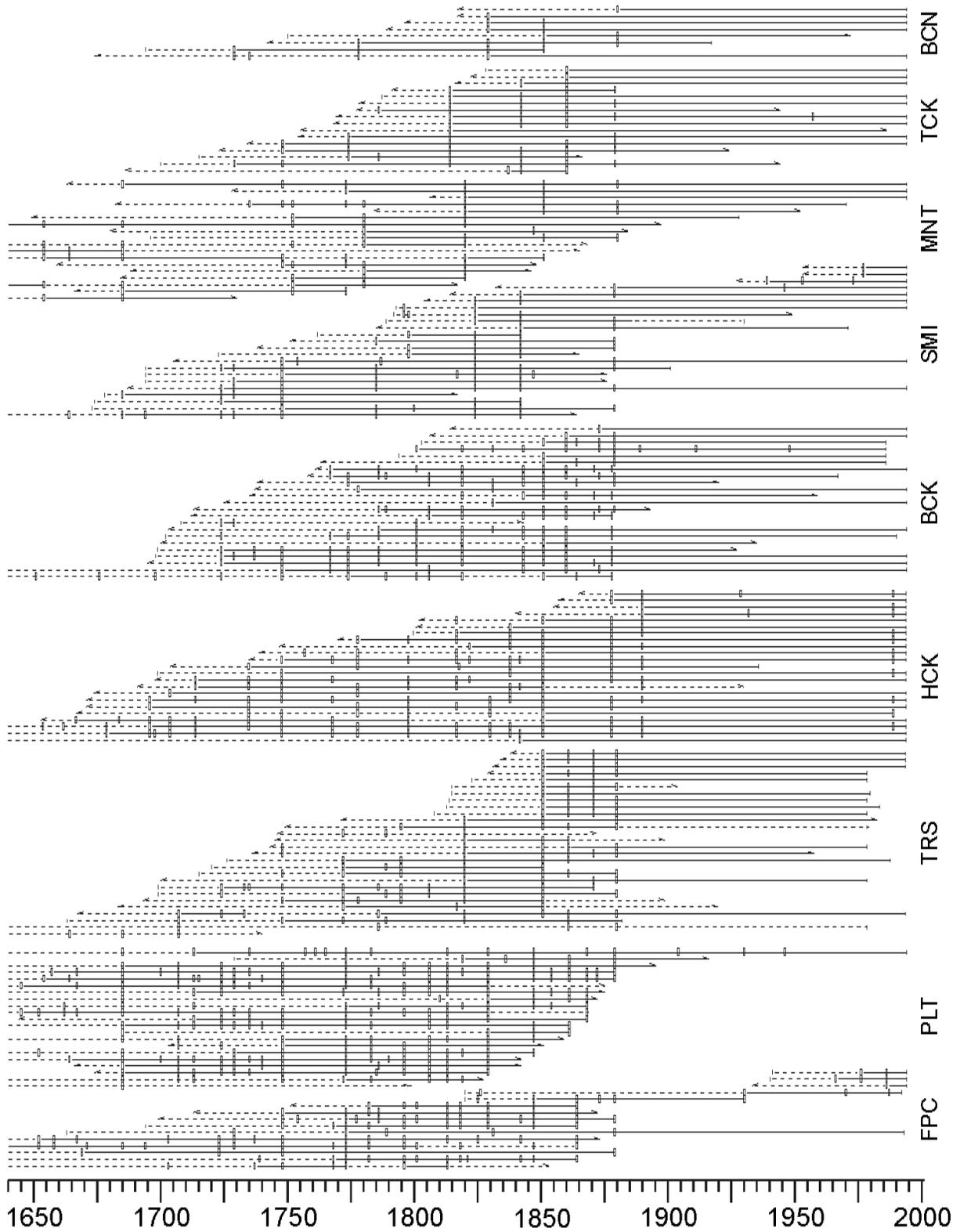
Figure II-2. Master fire chronologies for the nine sites sampled in the San Juan National Forest (Grissino-Mayer et al. 2004). See Figure II-1 and Table II-1 for site locations and descriptions.

Similar fire histories during the period of indigenous settlement have been documented for ponderosa pine forests throughout the mountains of Arizona and New Mexico (Swetnam and Baisan 1996;21-23). The median fire intervals from the San Juan Mountains appear to be a little longer than median intervals from comparable sites in central and southern Arizona; e.g., for all fires the median intervals are 6 - 10 years in the San Juans but only 2 - 5 years in many of the other study areas. However, it is difficult to make direct comparisons because of differences in sample sizes, extent of sampling area, and time periods. The trend of lengthening fire intervals with increasing elevation that we documented in the San Juan National Forest (Table II-2) also can be seen in other parts of the southwest, though the pattern is weaker when all sites in New Mexico and Arizona are examined together (Swetnam and Baisan 1996).

When the fire history data for the Jemez Mountains, located in the southern portion of the South Central Highlands Section (Touchan et al. 1996;39), are compared with those from the San Juan National Forest in the northern part of the Section, the similarities are more striking than the differences. The Jemez sites do appear to have slightly shorter median fire intervals: 4 - 17 years for all fires in Jemez ponderosa pine sites, compared to 6 - 10 years for all fires in the San Juans; and 8 - 22 years for more extensive Jemez fires compared to 12 - 30 years for extensive fires in the San Juans. In general, however, the data suggest that fire frequencies during the period of indigenous settlement were relatively similar in ponderosa pine forests throughout the South Central Highlands Section.

It is important to consider the variation in fire intervals as well as the median fire intervals during the reference period. Figure II-3 shows the statistical frequency of fire intervals detected at each of the nine study sites in the San Juan National Forest. Fires at short intervals may have played an essential role in ponderosa pine seedling recruitment, by creating patches of mineral soil and top-killing competing shrubs such as Gambel oak (Harrington 1985, 1987). However, fires at short intervals also would kill young pine seedlings, so occasional long intervals also would be required to allow pine seedlings to grow large enough to survive fire (Pearson 1950, White 1984). It can be seen in Table II-2 that the minimum, median, and maximum fire intervals in ponderosa pine forests were generally twice as long for extensive fires (>50% scarred) as for fires of any size (all fires). This indicates that many fires were small and localized; thus there would have been patches that escaped fire long enough for new pine saplings to become established, even though the stand as a whole was burning frequently.

Temporal Variability, Synchrony, and Seasonality of Fires:

Fire frequency varied not only from place to place, e.g., along an elevational gradient, but also varied through time. The fire chronology at each of the nine study sites is summarized in Figure II-4, for all fires in the top portion of the figure and for the more extensive fires (those that scarred at least 25% of recorder trees) in the bottom portion. There were two periods
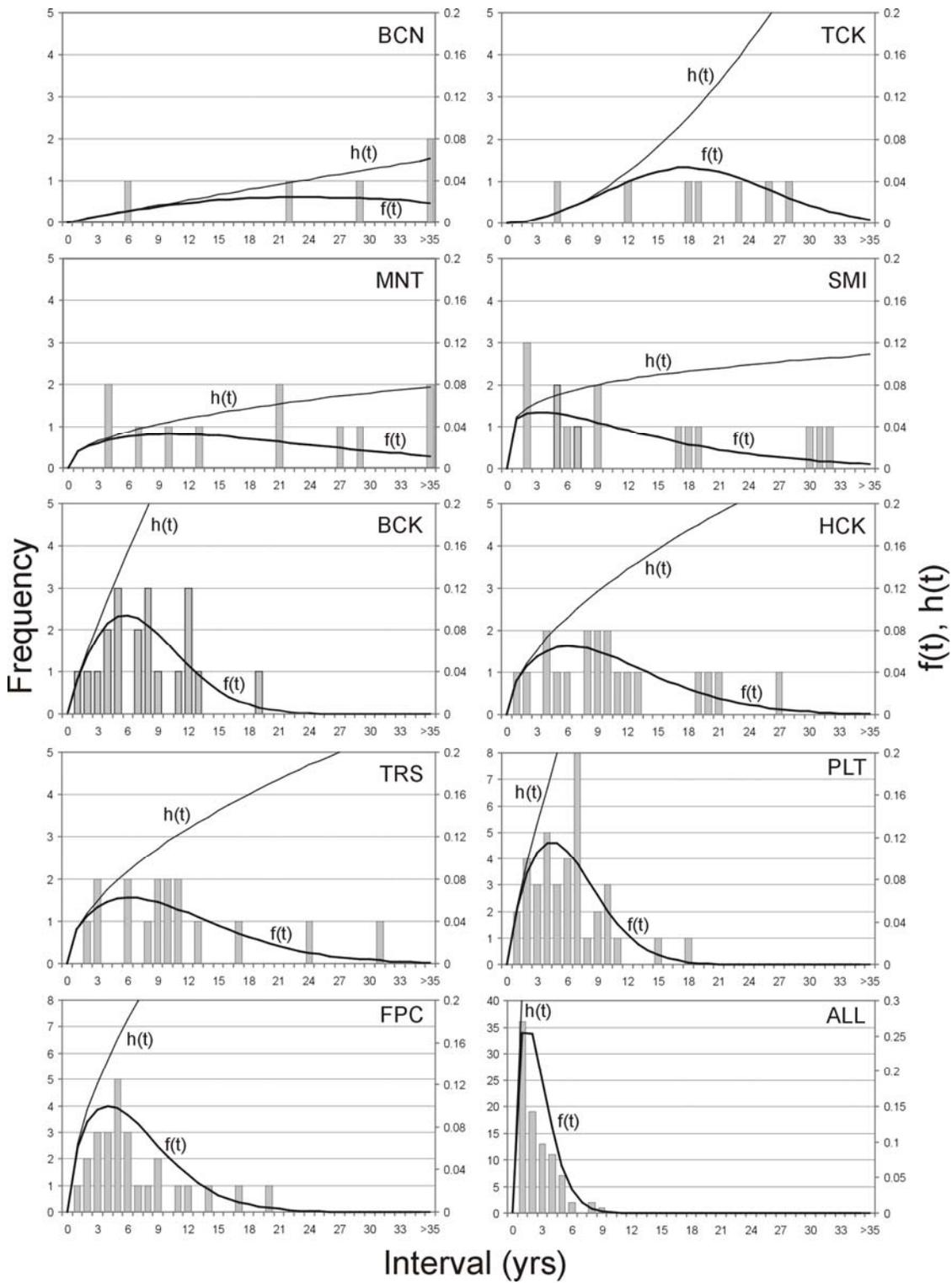
Figure II-3. Statistical frequency of fire intervals at nine fire history sites in the San Juan National Forest (Grissino-Mayer et al 2004). See Figure II-1 and Table II-1 for site locations and descriptions.

Table II-3. Fire occurrence by year and site (see Table II-1 for abbreviations and site descriptions). Numbers indicate the number of fire scars detected in a given year, with the number of recorders (trees that could have been fire-scarred in that year) present in our sample. A dash indicates that no fire scars were detected, even though at least one recorder tree was present. The sites are listed by longitude, from the western-most site on the left to the eastern-most site on the right. General forest type of each site is indicated in parenthesis (pine = ponderosa pine; mc = mixed conifer with ponderosa pine component).   In the column titled "Major fire yr" is the number of sites recording fire in those years for which 4 or more of the 9 sites recorded fire; these are the major years of extensive fire in the San Juan National Forest. The letter in parentheses indicates whether fires were recorded only in the eastern (E) or western (W) portions of the Forest; if no letter is present, then fires were recorded throughout the geographic extent of the Forest. An asterisk in this column designates years that also were among the top 20 most extensive fire years in New Mexico and Arizona, based on a regional network of 63 fire history sites (Swetnam and Baisan 1996).

Year	Major fire yr	FPC (pine)	PLT (pine)	SMI (pine)	TCK (mc)	BCN (mc)	HCK (pine)	MNT (mc)	TRS (pine)	BCK (pine)
1700		--	2/9	--	--	--	--	--	--	--
1703		2/4	--	--	--	--	--	--	--	--
1704		--	--	--	--	--	4/6	--	--	--
1707		--	12/12	--	--	--	--	--	4/4	--
1713		--	9/14	--	--	--	--	--	--	--
1714		--	--	--	--	--	6/8	--	--	--
1715		--	1/12	--	--	--	--	--	--	--
1723		3/3	--	--	--	--	--	--	--	--
1724	4 *	--	12/13	5/5	--	--	--	--	3/3	6/6
1729	6 *	3/4	9/13	5/6	1/1	2/2	--	--	--	2/4
1733		--	--	--	--	--	--	--	2/2	--
1735	5	--	8/13	--	--	1/2	8/10	1/5	1/1	--
1737		2/4	--	--	--	--	--	--	--	2/3
1739		1/5	--	--	--	--	--	--	--	--
1740		--	4/13	--	--	--	--	--	--	--
1748	8 *	7/7	15/16	9/9	3/3	--	10/12	4/6	6/6	5/5
1752		--	--	--	--	--	--	8/9	--	--
1754		1/6	--	1/9	--	--	--	--	--	--
1757		--	1/15	--	--	--	1/12	--	--	--
1761		--	1/15	--	--	--	--	--	--	--
1765		--	1/15	--	--	--	--	--	--	--
1767		--	--	--	--	--	--	--	--	7/7
1768		4/7	--	--	--	--	6/11	--	--	--
1772		--	2/16	--	--	--	--	--	9/9	--
1773		9/9	16/18	--	--	--	--	6/10	--	--
1774		--	--	--	4/5	--	--	--	--	8/9
1777		1/8	--	--	--	--	--	--	--	--
1778	4 (E)	--	--	--	--	3/3	10/14	--	1/8	1/10
1780		--	--	--	--	--	--	10/12	--	--
1782		6/9	--	--	--	--	--	--	--	--
1783		--	10/17	--	--	--	--	--	--	--
1785		--	1/17	7/10	--	--	--	--	--	--
1786	5	2/9	4/17	--	2/6	--	--	--	3/10	8/12
1787		--	--	1/10	--	--	--	--	--	--
1789		1/9	--	--	--	--	--	--	4/10	3/12
1790		--	1/17	--	--	--	--	--	--	--
1795		--	--	--	--	--	--	--	7/10	--

Cont. next page

Year	Major fire yr	FPC (pine)	PLT (pine)	SMI (pine)	TCK (mc)	BCN (mc)	HCK (pine)	MNT (mc)	TRS (pine)	BCK (pine)
1796		7/10	10/17	2/12	--	--	--	--	--	--
1798		--	--	4/14	--	--	10/13	--	--	--
1800		--	--	1/13	--	--	--	--	--	--
1801		4/10	--	--	--	--	--	--	--	10/15
1806		--	15/17	--	--	--	--	--	2/10	6/14
1810		--	1/18	--	--	--	--	--	--	--
1813		7/9	16/17	--	--	--	--	--	--	--
1814		--	--	--	12/12	--	--	--	--	--
1817		--	--	1/12	--	--	10/14	--	1/10	--
1818		6/9	--	--	--	--	1/12	--	--	--
1819		--	5/18	--	--	--	--	--	--	14/16
1820		--	--	--	--	--	--	13/14	15/16	--
1821		1/8	--	--	--	--	--	--	--	--
1822		--	--	--	--	--	3/13	--	--	--
1824		--	--	16/16	--	--	--	--	--	--
1825		2/9	--	--	--	--	--	--	--	--
1826		1/9	--	--	--	--	--	--	--	--
1829		4/8	17/19	--	--	6/6	--	--	--	--
1830		--	--	--	--	--	5/14	--	--	--
1831		1/8	--	--	--	--	--	--	--	5/14
1836		--	1/16	--	--	--	--	--	--	--
1837		--	--	--	1/13	--	--	--	--	--
1838		--	--	--	--	--	12/14	--	--	--
1842	4 (W)*	3/8	--	16/18	10/14	--	4/15	--	--	--
1843		--	--	--	--	--	--	--	--	12/15
1847	4 *	7/10	12/14	1/16	--	--	--	1/11	--	--
1851	5 (E)*	--	--	--	--	5/7	17/19	7/10	24/25	17/20
1854		--	4/12	--	--	--	--	--	--	--
1860		--	--	--	14/16	--	--	--	--	15/21
1861		--	8/11	--	--	--	--	--	17/23	--
1864		8/10	--	--	--	--	--	--	--	4/21
1868		--	8/10	--	--	--	--	--	--	--
1871		--	--	--	--	--	--	--	13/21	4/21
1872		--	2/6	--	--	--	--	--	--	--
1873		1/4	--	--	--	--	--	--	--	6/22
1878		--	--	--	--	--	17/20	--	--	9/22
1879	5 *	4/4	5/5	10/13	7/14	--	--	--	--	8/20
1880		--	--	--	--	3/7	--	4/9	17/21	--
1889		--	--	--	--	--	--	--	--	1/20
1890		--	--	--	--	--	16/23	--	--	--
1904		--	1/2	--	--	--	--	--	--	--
1911		--	--	--	--	--	--	--	--	1/19
1929		--	--	--	--	--	1/22	--	--	--
1930		2/3	1/1	--	--	--	--	--	--	--
1932		--	--	--	--	--	1/22	--	--	--
1939		--	--	1/9	--	--	--	--	--	--
1946		--	1/1	1/9	--	--	--	--	--	--
1948		--	--	--	--	--	--	--	--	1/16
1953		--	--	1/8	--	--	--	--	--	--

Year	Major fire yr	FPC (pine)	PLT (pine)	SMI (pine)	TCK (mc)	BCN (mc)	HCK (pine)	MNT (mc)	TRS (pine)	BCK (pine)
1957		--	--	--	1/10	--	--	--	--	--
1966		1/3	--	--	--	--	--	--	--	--
1970		1/3	--		--	--	--	--	--	--
1973		--	--	1/7	--	--	--	--	--	--
1976		2/4	--	--	--	--	--	--	--	--
1977		--	--	2/9	--	--	--	--	--	--
1986		3/5	--	--	--	--	--	--	--	--
1987		1/5	--	--	--	--	--	--	--	--
1989		--	--	--	--	--	11/21	--	--	--

Figure II-4. Fire chronologies, representing (A) all fires and (B) fires that scarred >25% of the sampled trees (i.e., widespread fires), for each of the nine fire history study sites in the San Juan National Forest (Grissino-Mayer et al. 2004). See Figure II-1 and Table II-1 for site locations and descriptions.

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in which fire frequency was reduced relative to the long-term average. First was from about 1750-1770, which is especially apparent in the lower portion of Figure II-4. This hiatus in fire occurrence may have been related to a shift in broad-scale atmospheric and climatic processes, such as El Niño -

Southern Oscillation (ENSO) or the Pacific Decadal Oscillation (Grissino-Mayer et al. 2004). Alternatively, the very large fires in 1748 may have reduced fuels so extensively that large fires could not occur until fuels had re-developed. Both mechanisms may have come into play to dramatically reduce the frequency of extensive fires during this period of 1750-1770. The second period of reduced fire frequency was from about 1880 to the present, and was mostly a consequence of Euro-American settlement as discussed below.

Each of the nine sites described in Table II-1 has its own more-or-less unique list of fire years during the reference period, i.e., in most fire years there was a fire at one or two sites but not at any of the others. However, there were a few key fire years when many or most of the sites experienced fire (Table II-3). More than half of our nine sites recorded fires in 1729, 1735, 1748, 1786, 1851, and 1879. During these summers the San Juan Mountains must have been shrouded in smoke for several weeks or months as large fires burned across the landscape. The years 1729, 1748, 1851, and 1879 also were major fire years throughout the southwest, apparently because of dry conditions across the entire region (Swetnam and Baisan 1996). The synchronous occurrence of major fire years throughout the Southwest and southwestern Colorado indicates a strong regional climatic control on fire frequency, both during the reference period and continuing today. For example, there tends to be little area burned in the Southwest during the wet El Nino phase of ENSO, but extensive fire activity occurs during the dry La Nina phase (Swetnam and Betancourt 1990). In fact, the major fire years in the Southwest tend to occur when one to three very wet El Niño years -- in which abundant fine fuels are produced -- are followed by a very dry La Niña year -- when the accumulated fine fuels dry out and carry extensive fires (Swetnam and Betancourt 1990, 1998; Grissino-Mayer et al. 2004). Recognition of this climatic pattern in fire occurrence has permitted improved forecasting of upcoming fire seasons in the Southwest.

There also was considerable variation in the seasons of fire occurrence during the reference period among the nine study sites in the San Juan National Forest (Table II-4). Summer fires would generally consume more organic matter and cause greater plant mortality than dormant season fires, because of higher temperatures and large amounts of nutrients and energy reserves in above-ground plant parts during the growing season (Harrington 1985). Thus, the variability in season of burn during the reference period, like the variability in extent of burn and interval between successive burns, was a significant component of the fire regime during the reference period, and may have contributed to pine regeneration and floristic diversity as described above. Over all sites and all fire scars combined, approximately half of the fire scars recorded summer burns and half recorded dormant season burns (spring or fall). However, in the years with big fires (e.g., 1748, 1820, 1851, and 1879), we found fire scars that were formed in all seasons (Figure II-5), suggesting that these very extensive fires burned all summer long.

Table II-4. Number and percentage of fire scars that were formed during different seasons of annual ring growth during the pre-1870 reference period in ponderosa pine forests (Grissino-Mayer et al. 2004). Some fire years are represented by more than one value, since fires in those years scarred more than one tree. Moreover, many prehistoric fires burned through more than one season. Therefore, this table indicates the *relative amount of area burned* during various seasons, not necessarily the seasons when individual fires occurred.

Season of fire scarring is determined by noting where a fire injury is located within an annual ring. This is a well-established dendrochronological method, and the data below on position within annual rings can be regarded as very reliable. However, we do not yet have a complete understanding of the specific months in which annual ring formation occurs in the San Juan Mountains. Moreover, the timing of ring formation probably varies at least somewhat from year to year and among different elevations and topographic positions, in response to variation in temperature and moisture conditions. Based on experience elsewhere in the southwest and on a pilot study conducted in 1995 in the Hermosa Creek area, we estimated the likely month that corresponds to specific positions within an annual ring. These interpretations of month of fire must be regarded as *tentative*, however.

D = dormant season, in the spring before ring growth has begun, or in the fall after ring growth is   completed   (possibly in April & May and in September & October)
E = early early wood   (possibly early to mid June)
M = middle early wood   (possibly late June to early July
L = late early wood   (possible mid to late July)
A = latewood   (possibly August)

SITE	D (spring/fall)	E	M	L	A	EMLA (summer)	PERIOD
Five Pine Canyon	31 (52)	7 (12)	7 (12)	7 (12)	7 (12)	28 (48)	1748-1864
Plateau	142 (82)	5 (3)	0 (0)	9 (5)	18 (10)	32 (18)	1685-1872
Turkey Springs	31 (36)	6 (7)	6 (7)	35 (40)	9 (10)	56 (64)	1748-1880
Hermosa	53 (47)	5 (4)	18 (16)	26 (23)	10 (9)	59 (53)	1715-1890
Benson Creek	61 (62)	4 (4)	15 (15)	15 (15)	3 (3)	37 (38)	1767-1889
Smoothing Iron	26 (42)	4 (6)	1 (2)	10 (16)	21 (34)	36 (58)	1729-1879
Monument	16 (31)	2 (4)	4 (8)	16 (31)	13 (26)	35 (69)	1654-1880
Taylor Creek	29 (72)	3 (8)	0 (0)	6 (15)	2 (5)	11 (28)	1786-1879
Burnette Canyon	7 (50)	0 (0)	0 (0)	6 (43)	1 (7)	7 (50)	1729-1880

---------------------- NUMBER (%) OF FIRE SCARS ------------------------
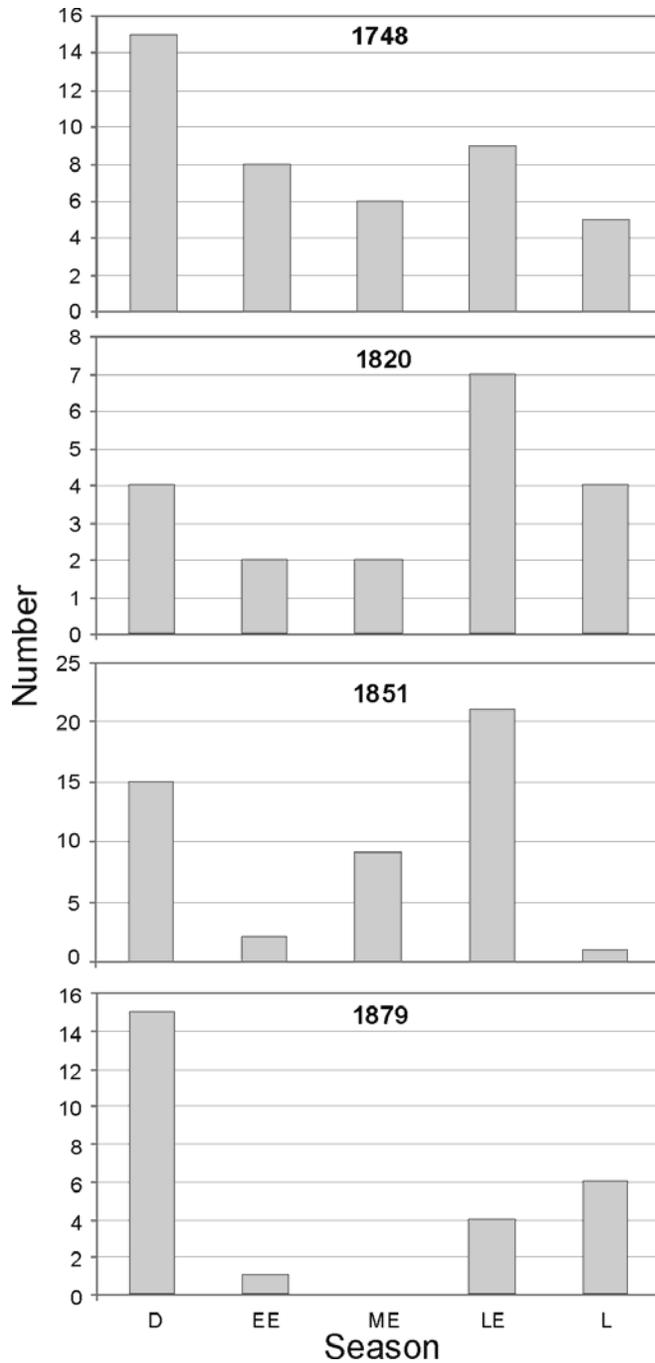
Figure II-5. Seasons when fire scars were formed, for four of the most extensive fire years documented in the San Juan National Forest (Grissino-Mayer et al. 2004). D = dormant season (early spring or late fall); EE = early growing season (mid-June to early July); ME = middle growing season (early July to mid-July); LE = middle to late growing season (mid-July to late July); L = late growing season (late July to August).

There were also striking differences among study sites, e.g., dormant season fires were more prevalent at the Plateau site, whereas summer burns predominated at Turkey Springs (Table II-4). What are the reasons for the differences among sites and over time in the dominant season of fire occurrence during the reference period? At this time we do not know. The Turkey Springs site is adjacent to a popular Ute camping area that was occupied almost continuously during the reference period (Duke 1995), so the abundance of summer burns there could reflect anthropogenic burning. However, the fire interval statistics for the Turkey Springs site (Table II-2) are not much different from the other ponderosa pine sites, suggesting that the Utes camping in this area, in fact, may not have added many ignitions to those caused by lightning. In particular, the Hermosa site, located in a rugged and mostly brushy or densely forested area, probably had less intensive use by Native Americans than the gently rolling forests and grasslands of the Turkey Springs site -- yet fire interval statistics are almost identical at Turkey Springs and Hermosa. These data suggest that the fire history of ponderosa pine forests in this area was controlled more by climate and fuels than by anthropogenic ignitions. Swetnam and Baisan (1996) and Allen (2002) suggest that this same conclusion applies in much of the Southwest. In contrast, anthropogenic ignitions may be more important in ponderosa pine forests of the northern Rockies where lightning is less frequent, but even there lightning may be sufficient to account for fire histories in most areas (Baker 2002). An early written account of the history of the San Juan National Forest (Anonymous 1971;122) states that although Indians were known to set fires to aid in hunting and improve forage in other regions, they rarely did so in the San Juans. Undoubtedly there were localized areas in the West where human activities significantly influenced the fire regime (e.g., Barrett and Arno 1982, Kaye and Swetnam 1999), but our data do not indicate any major human effects on fire frequency, or extent, in ponderosa pine forests during the period of indigenous settlement.

Fire history in ponderosa pine forests of the Uncompahgre Plateau

The most extensive fire history data available for ponderosa pine forests in the South Central Highlands Section are from the Jemez Mountains (Allen 1989, 2002) and from the San Juan National Forest (Grissino-Mayer et al. 2004, and above). We also have information from the Uncompahgre Plateau in an unpublished report by Brown and Shepperd (2003). On the Uncompahgre, mean fire intervals in five stands distributed across the Plateau ranged from 9-17 years (all fires) and from 15-44 years (fires that scarred $\geq$25% of recorder trees). These results are similar to the median fire intervals documented in the San Juan National Forest (Table II-2), but are longer than typical mean fire intervals from Arizona and New Mexico.

2. <u>Other agents of disturbance in ponderosa pine forests</u>

Although fire probably was the major form of disturbance in ponderosa pine forests during the reference period -- at least it is the disturbance about which we know the most -- there were

other important kinds of disturbances as well. Approximately 200 species of insects are known to affect ponderosa pine at some stage from cone to maturity (Schmidt 1988). Perhaps the most important insect in ponderosa pine forests was the mountain pine beetle (*Dendroctonus ponderosae*), discussed below. Dwarf mistletoes and various fungus diseases also are important components of ponderosa pine forests today, and were surely important in the past as well (Hawksworth and Shaw 1988). Unfortunately, we have very little specific information about the intensity and extent of disturbance by these agents during the period of indigenous settlement, and so we have a poor understanding of the overall importance of these disturbances relative to the well-documented effects of fire (Schmid and Mata 1996).

The Mountain Pine Beetle (Dendroctonus ponderosae):

In the 20th century there have been several outbreaks of the mountain pine beetle in ponderosa pine and lodgepole pine forests throughout the Central Rocky Mountain region (McCambridge et al. 1982a, Schmid and Mata 1996). Photos taken by Phillip Coolige in 1908 in what is now the San Juan National Forest (on file at the San Juan National Forest supervisor's office) show ponderosa pine trees described as having been killed by the Black Hills Beetle [mountain pine beetle]. This native bark beetle infests and often kills live trees (Furniss and Carolin 1977). It prefers larger trees (>20 cm (8 in) dbh), though it sometimes kills smaller trees (Sartwell and Stevens 1975). Although the intensity and extent of these recent outbreaks may have been influenced by

20th century conditions such as fire exclusion and unusually high tree densities (see below), it is likely that more-or-less comparable beetle outbreaks also occurred in ponderosa pine forests during the reference period (also see Baker and Veblen 1990). Schmid and Amman (1992) estimated that the intervals between successive mountain pine beetle outbreaks may range from 50 - 100 years, depending on the intensity of the previous outbreak. Outbreaks may last from 2 - 14 years (Schmid and Mata 1996). Between outbreaks (which is most of the time), the beetles persist at low, endemic populations, killing an occasional tree but having little impact on the forest as a whole.

From a timber production standpoint, mountain pine beetle outbreaks are clearly detrimental, because they selectively kill the larger, more valuable trees in a stand, and may in fact destroy a timber stand (Schmid and Amman 1992). From a broader ecosystem perspective, however, the effects are more complex. In lodgepole pine forests, and probably also in dense ponderosa pine, primary productivity shifts from an overwhelming concentration in the canopy to a more equitable distribution among canopy, understory, and ground layer strata (Romme et al. 1986). Herbaceous growth may increase because of increased light and water availability beneath the dead canopy trees (McCambridge et al. 1982b), and surviving canopy and understory trees also show increased growth rates immediately after the outbreak (Romme et al. 1986). Elk, deer, and other grazers and browsers may benefit from the increased forage production, and the large dead trees may provide feeding,

nesting, and perching sites for a variety of bird species -- whereas other animal species, such as the red squirrel that eats pine seeds, and bark and foliage gleaning birds like chickadees, nuthatches, and brown creepers, may decrease in abundance after the outbreak (Schmid and Amman 1992).

In stands of mixed species composition, selective mortality of the dominant canopy pine can hasten the process of succession, shifting canopy dominance to the more shade-tolerant Douglas-fir, white fir, or other species not attacked by the beetles. If the beetle outbreak is not followed by fire, then ponderosa pine may not become reestablished on the site because its seedlings generally do not tolerate deep organic litter layers or shade (discussed below). Thus, it is possible (though never documented) that severe mountain pine beetle outbreaks in the past may have converted stands having low initial density of ponderosa pine to forests composed entirely of other tree species -- or even conceivably to shrublands of Gambel oak, which are commonly found within extensive tracts of ponderosa pine forest throughout the South Central Highlands Section, and whose origin is difficult to explain.

The reduction in canopy transpiration following a severe beetle outbreak potentially may lead to increased groundwater recharge and stream flow for several years if heavy mortality occurs over a large portion of a watershed (documented for outbreaks in lodgepole pine and spruce-fir forests, but not for ponderosa pine per se; Schmid and Amman 1992). Risk of crown fire increases during and for about 2 years after the outbreak, while beetle-killed trees are covered with dead needles (Schmid and Amman 1992, Bentz et al.

2009), but then may actually decrease because the standing defoliated trees reduce the continuity of canopy fuels. As the large beetle-killed trees gradually fall over a period of several decades, they increase the heavy fuels on the ground and the chances of very intense surface fires (Brown 1975).

Other Disturbances:

There were many other agents of disturbance in ponderosa pine forests of the reference period. Dwarf mistletoe (*Arceuthobium vaginatum*) is a parasitic vascular plant that grows on branches and stems of ponderosa pine, causing reduced tree growth, distortion of stems and branches, and eventual death of the tree in some situations (Hawksworth and Wiens 1995). Although mistletoe is considered a pest from the viewpoint of timber production, it is a component of the native flora, and its abundance is positively correlated with bird density and diversity (Bennetts et al. 1996). *Armillaria* spp. and other pathogenic fungi cause root diseases and eventually may kill ponderosa pine trees (Hawksworth and Shaw 1988). Pandora moth (*Coloradia pandora*) defoliates ponderosa and lodgepole pines, causing reduced tree growth and sometimes mortality (Schmid and Mata 1996). Severe, localized windstorms can blow down all or most of the trees within small areas of a few hectares or less. We know almost nothing about the intensity and extent of these disturbances in ponderosa pine forests of the reference period (Schmid and Mata 1996), but they probably had much less impact on forest structure and function at the landscape scale than did periodic fire and mountain pine beetle outbreaks. Perhaps the most important effect of

mistletoe and root diseases at the landscape scale was their interaction with fire and beetles (Knight 1987). Trees weakened by mistletoe, insects, or disease may be more susceptible to beetle infestation, and the witches brooms produced by mistletoe-infested trees may ignite more readily than normal branches.

3. Pre-1870 stand structures in ponderosa pine forests

In ponderosa pine forests where fires were frequent and of low severity during the reference period (i.e., where the fire regime was the Southwestern model), these fires thinned out the smaller trees and brush, and maintained open forests of large, widely spaced trees with abundant herbaceous growth in the open areas between trees. This interpretation is supported in part by numerous photographs and accounts of

19th-century travelers in the Southwest, who described "park-like" forests in which one could see for great distances and which provided ample forage for pack animals. Table II-5 contains a sampling of these early accounts. A similar open structure of ponderosa pine forests also is suggested by early written accounts and photographs from elsewhere in the Mountain West, e.g., the Colorado Front Range (Veblen and Lorentz 1991), the Black Hills of South Dakota (Progulske 1974, Grafe and Horsted 2002; but see Shinneman and Baker 1997), western Montana (Gruell et al. 1982), and the interior Columbia River Basin (Everett et al. 1994, Skovlin and Thomas 1995). However, early photos and written descriptions of dense ponderosa pine stands also exist, e.g., from the Black Hills (Shinneman and Baker 1997) and several other locations throughout the Rocky Mountains (Baker et al. 2006).

Table II-5. Some descriptions of ponderosa pine forests written by early Euro-American explorers in the South Central Highlands Section.

"In the beautiful valley of the Conejos River, after striking the timbered region, we found luxuriant bunch-grass covering the ground as thickly as it could stand. In November it was still green about the roots, and was eagerly eaten by our starved mules. Pinus ponderosa formed open clumps, and under protection of these trees it attained what seemed to be its maximum growth."
-- J. T. Rothrock, 1878 (cited in Cooper 1960;131)

"Throughout the bull [ponderosa pine] type there is good cattle range, consisting of blue-stem grass beneath the trees and bunch grass in the parks. The underbrush is very heavy, chiefly oak brush, choke-cherry, scarlet thorn, and wild rose. Reproduction of bull pine is poor."
-- C. DuBois, 1903 (DuBois 1903;7)

"Pagosa lies in the heart of that splendid pine forest, which coveres a tract one hundred and thirty miles east and west by from twenty to forty miles north and south. Here the trees grow tall and straight, and of enormous size. No underbrush hides their bright, clean shafts, and curiously enough, it is only in special locations that any low ones are to be found."
-- Ernest Ingersoll, 1885 (cited in Paulson and Baker 2006; 177)

Some caution must be used in interpreting these early written records and photographs (Forman and Russell 1983, Whitney 1994). Early travelers and explorers obviously were not unbiased observers on a mission to collect objective data for the benefit of future ecologists. Many may have placed undue emphasis on things like grass cover because forage was so important for their pack animals. Their observations also were strongly influenced by their previous experiences and expectations. Nevertheless, taken together, the contemporary written and photographic evidence suggests that an open forest structure characterized many—though not all--of the ponderosa pine forests throughout western North America when numbers of literate Europeans first encountered them in the mid to late 1800s.

Ponderosa pine forests of the volcanic Jemez Mountains, in the southernmost portion of the South Central Highlands Section, were similar in structure to other ponderosa pine forests of the Southwest during the reference period, i.e., a predominance of open stands of large trees (Allen 1989, 2002). The picture is less clear for the northern portions of the South Central Highlands Section. The only 19th-century written record we know of is an account of forests in the Conejos River Valley in 1878 (Table II-5). We also have DuBois' 1903 description of the proposed San Juan Forest Reserve, which may be colored somewhat by the effects of early grazing and timber cutting (see the section on legacies of previous land use, below). Zier and Baker (2006) present seven early photos (1874 - 1948) of the San Juan Mountain country, including two in which ponderosa pine is prominent.

Unfortunately, most of the historic photographs of this region date from the early to mid 20th century, when major changes in forest structure already may have begun to develop.

Despite the paucity of direct evidence, we can infer from contemporary writings and from observations today, that there were some important differences between ponderosa pine forests of southwestern Colorado and those in other portions of the Southwest during the period of indigenous settlement. For example, most forests in southwestern Colorado probably did not have quite the abundance of grass described for ponderosa pine forests in Arizona. The early explorers noted that grass was more abundant on soils derived from lava flows than on soils derived from sedimentary rocks (Cooper 1960;131), and most ponderosa pine forests of the San Juan Mountains and Uncompahgre Plateau are found on sedimentary substrates. Nevertheless, a number of native bunchgrasses are found in ponderosa pine forests of southwestern Colorado, especially in places that have received only light grazing (Paulson and Baker 2006), suggesting that grass cover was somewhat abundant in these forests during the reference period—though it probably is impossible to know just how abundant the grass cover was in any particular place.

The abundance of shrubs (notably Gambel oak) in ponderosa pine forests of the South Central Highlands Section, especially in the northern portions of the Section, is one of the features that makes this region especially distinctive. Although the cover and density of shrubs may have changed in some areas during the last century (see the section on legacies of past land use,

below), oak and other shrubs certainly were present, and probably abundant, in many ponderosa pine forests of the South Central Highlands Section during the period of indigenous settlement, especially in portions of southwestern Colorado. For example, Dominguez and Escalante, in writing of their trip through southwestern Colorado in 1776 (near present-day Pagosa Springs, Durango, and Mancos), made frequent mention of small or dwarf oak and other shrubs growing in association with pines (Bolton 1950;140151). DuBois (1903; p. 7), in an early survey of the proposed San Juan Forest Reserve, wrote of the bull pine [ponderosa pine] type that "The underbrush is very heavy, chiefly oak brush, choke-cherry, scarlete thorn, and wild rose." Similarly, a local Forest Ranger with the initials R.S.W. wrote in 1915 that "Oakbrush, in the vicinity of the Durango National Forest, occupies a wide belt ranging in altitude from approximately 1,800 – 2,700 m (6,000 - 9,000 ft), which is identical with that of western yellow pine [ponderosa pine] with which oakbrush is always associated to a certain extent, depending on the density of the pine stand." ("Cumulative silvical report – San Juan National Forest;" on file at San Juan National Forest supervisor's office, includes entries from 1909-1958). The few photos of ponderosa pine forests from ca. 1900 clearly show oak present in the understory of ponderosa pine stands (W. Baker, personal communication).

In addition to the written and photographic evidences described above, we have a few direct measurements of likely ponderosa pine forest structure during the period of indigenous settlement. Based on detailed sampling of extant trees, dead trees, and stumps,

Fule et al. (1997) determined that a stand in northern Arizona had an average of 65 ponderosa pine per hectare in 1883 (plus 83 tree-sized stems/ha of Gambel oak and other species). Basal area of ponderosa pine trees was approximately 12 m^2/ha. In another northern Arizona stand, using similar methods, Covington and Moore (1994), estimated that the density and basal area of ponderosa pine, prior to European settlement in the late 1800s, were 43 trees/ha and 14 m^2/ha, respectively. Madany and West (1983) found 56 ponderosa pine > 100 years old per hectare on two isolated mesas in southern Utah that had never been logged or grazed by livestock (though the fire history on these mesas was unique in some respects; see Madany and West 1983).

In the San Juan National Forest, we estimated the density and spatial pattern of ponderosa pine trees in pre-1900 forests by measuring the old stumps and occasional old trees left by early 20th-century century loggers at the Plateau, Smoothing Iron, and Five Pine Canyon sites (Table II-1). Sampling quadrats ranging in size from 25 - 10,000 square meters (8 replicates of each size) were randomly located, and the number of old stumps and trees was tallied in each quadrat. Care was taken to avoid counting stumps of younger trees produced by recent logging operations; the old stumps were recognizable from their weathered condition. Living trees old enough to have been present in the stand before 1900 were distinguished from younger trees (post-1900) by the deeply furrowed, yellow bark and stout branches of old trees. A few of the trees may have been misclassified, but these morphological characteristics are reasonably reliable for distinguishing

trees older or younger than about 100 years (Pearson 1950, White 1984). Within the same quadrats we also tallied the number of trees that had become established after 1900 (discussed below).

The density of pre-1900 and post-1900 trees was computed using the data from the largest quadrats (5000, 7500, and 10000 square meters), and the data from all quadrats was used to compute Morisita's index (Morisita 1959, Brower and Zar 1977). Morisita's Index is related to the variance/mean ratio, and is widely used to detect the spatial scale at which organisms are aggregated. In general, the quadrat size for which Morisita's Index is highest is approximately the size of the clumps themselves. There can be more than one scale of clumping in a stand.

The results of this analysis indicated that these ponderosa pine forests in the San Juan Mountains contained an average of 25 - 65 trees/ha during the late 1800s (Table II-6a). Average stump diameter ranged from 51 – 69 cm (Table II-6b). The reasons for the differences among sites are unknown; there are no obvious patterns related to elevation, topography, or geographic location. Our sampling procedure probably under-estimates the density of pre-1900 trees, and over-estimates their average size, since some of the smaller stumps may have completely decomposed. Nevertheless, these results indicate that there were comparable densities of large trees in at least some parts of northern Arizona and southwestern Colorado during the period of indigenous settlement. The variability in tree density is as important as mean density. By computing the mean, plus or minus one standard error of the mean, we can estimate the range of values expected to encompass 67% of all measured samples. This range, which represents our best estimate of the historical range of variability in ponderosa pine forest density, was from 11 - 98 trees/ha in the five sites in the San Juan National Forest (Table II-6a).

A similar reconstruction of historical forest structure was made for ponderosa pine forests on the Uncompahgre Plateau (Binkley et al. 2008). In eleven plots, 0.5-1.0 acre in size, average tree density in 1875 was estimated to be 55 trees/acre (range 30 - 90 trees/acre) and average basal area was estimated at 55 ft^2/acre (range 20 - 90 ft^2/acre). The stands in 1875 contained a few large trees (some with diameters >0.9 m (3 feet)), relatively numerous trees of medium size (0.3 – 0.6 m (1 - 2 ft) diameter), and a few smaller trees.

These results from the San Juan Mountains and the Uncompahgre Plateau demonstrate that ponderosa pine forests of the South Central Highlands Section were different in some important ways from ponderosa pine forests of northern Arizona. Both density and basal area tended to be higher in the South Central Highlands Section than in forests near Flagstaff, where basal area averaged 35 ft^2/acre and ranged from 13 - 62 ft^2/acre in 1910 (Moore et al. 2004).

Note that the sampling described above was conducted in places where stumps of trees that were alive prior to 1900 were visually conspicuous. This may have biased our sampling towards open stands of large trees. We do not know how common dense stands of small trees might have been in the pre-1900 landscape; additional research would be required to answer this question (e.g., Baker et al. 2006).

Table II-6. Pre-1900 stand structure in five sites dominated by ponderosa pine in the San Juan National Forest, Colorado, based on sampling old trees and stumps in quadrats of 4,000 - 10,000 m^2. See Table II-1 for site descriptions.

(a) Tree densities. Range of variability refers to the mean plus or minus one standard error. RD refers to ranger district.

(b) Stump diameters.

(a) Site	Mean density (trees/ha)	Range of variability (trees/ha)	Number of samples	Comments
Benson Creek	24	11 - 36	32	Pagosa RD ... thick brush ... trees & stumps
Smoothing Iron	37	13 - 60	32	Dolores RD ... thick brush ... trees & stumps
Plateau Creek	42	22 - 63	36	Dolores RD ... easy visibility ... mostly stumps
Five Pine Canyon	50	30 - 70	32	Dolores RD ... thick brush ...mostly stumps
Turkey Springs	65	33 - 98	24	Pagosa RD ... easy visibility ... trees & stumps

(b) Site	Mean diameter (cm)	Standard deviation (cm)	Smallest - largest sample (cm)	Number of samples
Benson Creek	51	14	20 – 90	216
Smoothing Iron	66	27	20 - 150	193
Plateau Creek	69	24	20 - 130	226

Although the relative extent of open vs. closed ponderosa pine forests in the low elevation landscapes of southwestern Colorado during the period of indigenous settlement is currently unknown, we do have some partial information. An 1888 photograph of Durango, taken from the slopes of Smelter Mountain, provides a clear view of Animas City Mountain on the north side of the town. In this photo, approximately one fourth to one third of the ponderosa pine forest area appears dense; the other two-thirds to three-fourths appears open and park-like. We believe that little logging had yet occurred on Animas City Mountain in 1888, so this photo may depict the patterns of forest structure that developed on the landscape through the action of fire, insects, and forest growth. However, it is a single snapshot that provides little detail on stand-level forest structure, and more information obviously is needed.

Another piece of evidence for a heterogeneous ponderosa pine forest landscape comes from a recently discovered map depicting a timber inventory of the westside pine zone in the San Juan National Forest made in the early 1920s, just before industrial-scale logging began in the area (southwestern archive collections, Fort Lewis College library). Despite some obvious biases and probable inaccuracies of this early map of timber volumes, it is apparent from the information on this map that the pre-logging pine forest was not uniform even throughout this area of relatively similar topography and climate.

The park-like scenario for pre-1800 ponderosa pine forests has been so well developed, both here and elsewhere in the West, that it may be tempting to assume that these forests were relatively homogeneous throughout the South Central Highlands Section. However, we stress that in fact there was considerable variability in density and structure. Because one important goal of ecosystem management is to conserve the natural heterogeneity and variability of the systems being managed, we need to be cognizant of the variation in ponderosa pine forests as well as the central tendencies (Kaufmann et al. 1994).

In addition to the landscape-level heterogeneity in ponderosa pine forest structure described above, there was important spatial variability at the stand level. The pre-1900 trees documented both in the San Juan Mountains and on the Uncompahgre Plateau were arranged in distinctly clumped patterns, which were conspicuous both visually and statistically. Early commentators also emphasized the clumping of ponderosa pine trees; e.g., the Cumulative Silvical

Report in 1913, page 1 (on file at the San Juan National Forest supervisor's office) describes a "broad strip of semi-humid country, bounded approximately by the altitudinal lines of 6500 and 8500 feet. On this strip occurs an open stand of western yellow pine [ponderosa pine] timber, interspersed with pinon [sic] points, oak brush hillsides, open parks, spruce and balsam canyons, cottonwood bottoms and aspen slopes. The pine never occurs in broad dense stands evenly distributed, but always in clumps. These clumps vary in density and in their proximity to one another." Among the stands in the San Juan Mountains where we reconstructed pre-1900 stand structure, there was no single clump size; rather the size of the clumps differed from place to place: major peaks in Morisita's Index (i.e., sizes of clumps) for pre-1900 trees were detected at 100, 300, 800, and 2500 m^2 at Smoothing Iron; at 700 and 1600 m^2 at Five Pine Canyon; and at 900 and 2500 m^2 at Plateau (Figure II-6). The trees on the Uncompahgre Plateau also exhibited a clumped spatial pattern, aggregated at distances <23 m (75 ft) with either a random or relatively uniform arrangement. Ponderosa pine forests in northern Arizona showed similar strong aggregation of pre-1900 trees (Pearson 1950;121). Using nearest-neighbor analysis, White (1984) determined that clumps of pre-1900 ponderosa pine near Flagstaff, Arizona, generally contained 3 - 44 trees and covered areas of 200 - 2900 m^2.

A major gap in our understanding of ponderosa pine forest structure during the reference period has to do with snags and downed wood. Popp et al. (1992) and Mehl (1992) have developed criteria for old-growth ponderosa pine forests in the Southwest

and Rocky Mountains (these criteria are summarized in Table 5.1 of Paulson and Baker 2006). For example, the recommended minimum density of standing dead trees ranges from one 14 in (35 cm) dbh snag to two 10 in (25 cm) dbh snags, and downed logs should be a minimum of two pieces having 12 in (30 cm) diameter. However, these guidelines are based on sampling the structure of old forests as they exist today; it is uncertain to what degree these characteristics applied to ponderosa pine forests during the reference period. Relatively frequent fire would be expected to reduce the amount of dead wood on the forest floor via consumption, but might increase the density of snags via mortality. But this is speculation: in truth we simply do not know densities or quantities of dead wood in the pre-1870 ponderosa pine forests of the South Central Highlands Section.
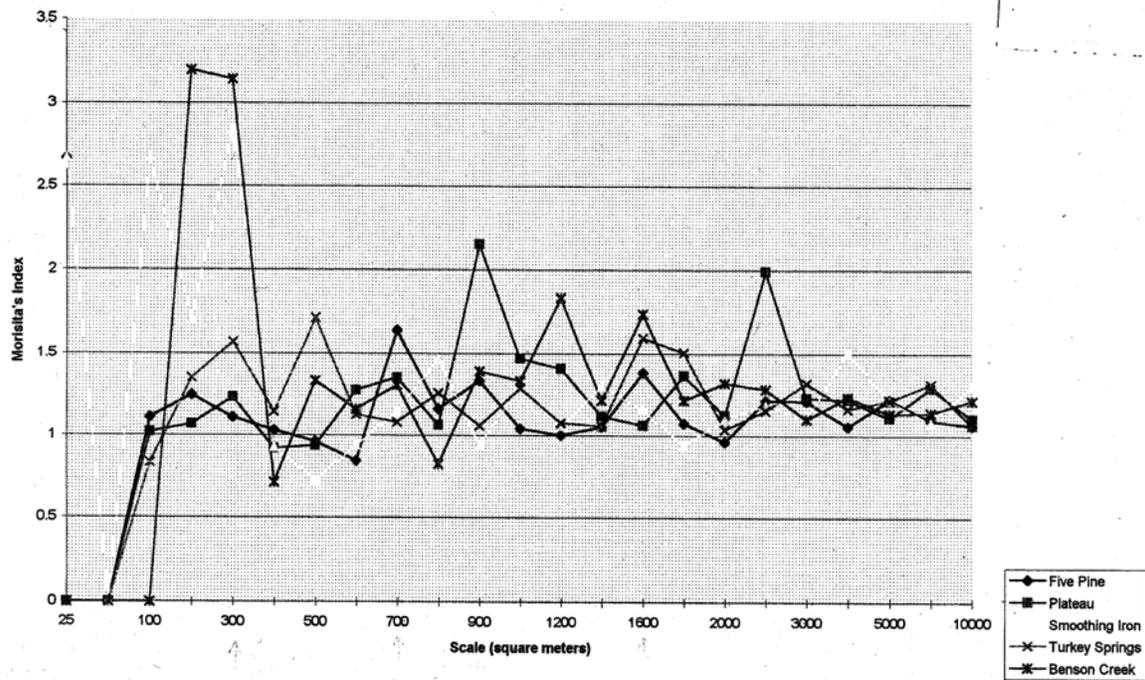
Figure II-6. Aggregation patterns of trees alive before 1900 in ponderosa pine forests of the San Juan Mountains. Spatial scales (horizontal axis) where values of Morisita's Index are highest (vertical axis) represent scales at which trees were aggregated, i.e., approximate sizes of clumps of trees (Morisita 1959, Brower and Zar 1977).

Functional Implications of Forest Structure during the Reference Period:

The overall low density and clumped dispersion of ponderosa pine trees in many pre-1900 forests probably was an important component of the habitat that supported a diverse assemblage of wildlife. For example, Abert's squirrel is almost entirely dependent on ponderosa pine for all stages of its life history (Patton 1977). It nests in clumps of large trees with interlocking branches, and feeds on ponderosa pine seeds, buds, and twigs. Other species that prefer dense stands of ponderosa pine include sharp-shinned hawk, Cooper's hawk, and goshawk (Reynolds 1983), pygmy nuthatch, hermit thrush, western flycatcher, and black-headed grosbeak (Finch et al 1997). Another suite of species prefers more open habitats, and probably were concentrated in the little glades that existed between clumps of trees, e.g., deer mouse, brush mouse, Mexican woodrat, and cottontail (Patton 1988), rock wren, and spotted towhee (Finch et al. 1997).

Still other species, including deer and turkeys, probably foraged in the open spaces between clumps of trees, where herbaceous and shrub production was greater, then moved into the clumps of trees for roosting, thermal, and hiding cover (Patton 1988). Open glades probably also contained a distinctive set of plant species that required more light

and could tolerate lower nitrogen availability (Moir 1966), while another set of plant species may have been concentrated in the special micro-habitat that existed under the clumps of trees. This description of pre-1900 patterns of herbaceous plant distributions is necessarily speculative, however, because these patterns were profoundly altered by heavy livestock grazing in the late 1800s (see below). Clumping of ponderosa pine trees may have retarded the spread of mistletoe, since the seed dispersal distance for this parasitic plant is limited.

In addition to providing diverse wildlife habitat, the clumping of mature ponderosa pine trees probably had an important influence on basic ecological processes of energy flow and material cycling. Parsons et al. (1994), measured soil nitrogen concentrations and rates of nitrogen mineralization and nitrification in closed-canopy stands of lodgepole pine and in gaps where trees were experimentally removed. Soil nitrogen levels were extremely low beneath the canopy, but greater in openings created by removal of 30 + trees. Openings within ponderosa pine forest canopies tend to accumulate more snow and to yield more water to below-ground water tables than do areas of closed pine canopy. Beneath a tree canopy, increased interception and transpiration act to reduce the amount of water percolating below the tree rooting zone (Brown et al. 1974, Baker 1988).

C. LEGACIES OF EURO-AMERICAN SETTLEMENT AND CURRENT CONDITIONS

Many of the ponderosa pine forests of the South Central Highlands Section have been dramatically altered during the last century. The major

agents of change have been logging, grazing, and elimination of fire, although climatic variability also has been involved. Native Americans cut some

trees and ignited some fires during the period of indigenous settlement, and some tribes (e.g., the Navajo) grazed substantial numbers of livestock in some areas. However, the logging, grazing, and fire exclusion introduced by Euro-American settlers in the late 1800s, were unprecedented in their intensity and extent. It was these recent human activities, interacting with the background biota, soils, and climatic variability, that created the ponderosa pine forests that we see in the area today. Thus, the ponderosa pine forest landscapes of the South Central Highlands Section represent a cultural landscape, i.e., a landscape that has been shaped as much by human activities as by natural processes (Foster 1993, Bunting and Romme 2001). We now examine each of these cultural legacies in turn.

1. Legacies of Logging ("High-Grading") in the Late 1800s and Early 1900s

There was extensive but scattered logging activity throughout the ponderosa pine forests of the South Central Highlands Section beginning in the late 1800s (Paulson and Baker 2006). However, most of the early logging is not well documented in historical records. Much of the logging was conducted by small, local outfits, who left no trace of their operations except a few stumps in the forest (Duane Smith, personal communication). Logging in the San Juan Mountains began as early as 1875, when small sawmills were constructed near Silverton and Parrott City to provide lumber for mining activities. The first sawmills in the Pagosa Springs area were built in 1901 - 1902, and sawmills operated in the

Spring Creek and Canyon Creek areas beginning in 1907 (Anonymous, Forest History, Volume 1). Several small-scale logging operations are documented in the Dolores River Valley before 1905 (Dishman 1982). In addition to these commercial sawmills, ranchers and settlers obtained the timber, fence posts, and fuel wood that they needed from the private and national forest lands surrounding their homes. Nearly all of the timber produced in these early days was used locally for buildings, mine supports, and railroad ties, and ponderosa pine was the predominant timber species because of its easy accessibility (Anonymous, Forest History, Volume 1).

The scale and impact of logging increased dramatically in the 1890s with the advent of railroad logging. Ormes (1975) describes several small logging companies and narrow-gauge railroad companies that operated in the vicinity of Tierra Amarilla, Chama, and Pagosa Springs from the mid 1890s through the early 1920s. To be commercially feasible, a railroad logging operation generally had to remove nearly all merchantable trees within reach of the tracks (Pearson 1950, Whitney 1994). According to the San Juan National Forest (1962, p. 38), "Most of the accessible ponderosa pine timber in the vicinity of Pagosa Springs and south into New Mexico was railroad-logged prior to 1918." DuBois (1903;9) wrote at the turn of the 20th century that "The supply of bull pine [ponderosa pine] on the lower slopes is nearly exhausted, and as soon as it is, the sawmill men will move on to the spruce."

Large-scale, commercial timber operation in the western San Juan Mountains began in 1924 when the New Mexico Lumber Company purchased 4

million board feet in what was then called the Montezuma National Forest north of Dolores (Figure II-1), and began construction of a large mill in McPhee as well as an extensive network of roads and railroads for hauling timber to the mill (Mausolf 1982, Smith 1982). Timber was hauled to the mill by railroad and also by trucks beginning in the 1930s. By 1950, essentially all of the old growth ponderosa pine forests of this region had been liquidated. Heavy logging of ponderosa pine stands in the Piedra River and Pine River Districts of the San Juan National Forest occurred between 1935 and 1955 (San Juan National Forest 1962, p. 38).

In sum, it appears that nearly all of the accessible ponderosa pine forests were affected by the logging operations of the late 1800s and the first half of the 20th century. This early logging usually involved "high-grading", i.e., selectively removing the highest quality trees (generally the large, old-growth trees) and leaving the smaller individuals or species of lesser monetary value. Consequently, unlogged, old-growth ponderosa pine stands today are very few in number, and nearly all are in locations where it was physically impossible or uneconomic to log.

Direct Effects of Early High-Grade Logging:

The most obvious effect of the early logging of ponderosa pine forests is the general lack of large, old trees and snags in these forests today. Tables II-7 and II-8 summarize the current size and age class distributions of ponderosa pine in the westside pine zone, based on RIS data from the San Juan National Forest. This 180,000-acre area north of Dolores, Colorado, is where the Montezuma

Lumber Company operated from the mid-1920s to late 1940s (Smith 1982).

The forest today is overwhelmingly dominated by relatively small, young trees. Eighty-four percent of stems greater than breast height are 30 cm (12 in) or smaller in diameter; 94% are 40 cm (16 in) or smaller (Table II-7). Very large trees (> 60 cm (24 in) dbh) comprise less than 1% of the population. Average tree size in the entire study area today is only about 20 cm (8 in), whereas the average diameter of stumps in the Plateau Creek study area (located within the westside pine zone) is 68 cm (27 in) (Table II-6).

Almost 90% of sampled trees were 90 years old or less; 94 % were 110 years old or less (Table II-8). Trees that germinated prior to 1870 (>120 years old today) comprise less than 4% of the population. (These are ages at breast height that were collected in the late 1970s and 1980s, so the trees actually are 10-20 years older than the numbers indicate.) For comparison, Table II-9 contains the age class distribution of ponderosa pine in the lower Hermosa drainage, near Durango, Colorado, which has never been logged (but which has been subjected to fire suppression). The Hermosa site contains a wide spread of tree ages, including many trees >300 years old. Presumably the westside pine zone had a similar age-class distribution prior to the heavy logging that began in the 1920s.

Not only are the very large trees scarce in the westside pine zone today, they are gradually disappearing through natural mortality. The diameters of trees that had died within the past five years at the time when stand exams were made were compared to the diameters of living trees in the study area. The ratio of percent recently died to percent live trees

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Table II-7. Size class distribution of ponderosa pine trees in the 180,000 acre westside pine zone of the San Juan National Forest as of 1994. Data from RIS files, San Juan National Forest.

Diameter Class (inches at breast height)	Trees / acre	Cumulative % of Trees of This Size or Smaller
< breast height	27	21
1 - 4	23	40
4 - 8	32	65
8 - 12	24	84
12- 16	13	94
16 - 20	5	98
20 - 24	2	99
24 +	< 1	100

Table II-8. Age class distribution of ponderosa pine trees in the 180,000 acre westside pine zone of the San Juan National Forest as of 1994. Data from RIS files, San Juan National Forest.

Age Class (years at breast height)	Frequency (% of all trees sampled)	Cumulative % of Trees of This Age or Younger
1 - 10	1	1
10 - 20	1	2
20 - 30	2	4
30 - 40	8	12
40 - 50	14	26
50 - 60	21	47
60 - 70	24	71
70 - 80	13	84
80 - 90	5	89
90 - 100	3	92
100 - 110	2	94
110 - 120	2	96
120 +	4	100

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reveals that there is a disproportionate mortality occurring in the very small and very large size classes; i.e., they are dying in greater numbers than would be predicted solely on the basis of their abundance (Table II-10). Trees in the middle range of size classes (30 – 60 cm (12 - 24 in)) are dying at lower rates than would be predicted by their abundance alone. This pattern is to be expected, since the middle-sized trees are around 100 years old, at their prime; the smaller

trees are subject to suppression by larger trees; and many of the very large trees are approaching their maximum life spans.

The scarcity of large, old trees in the westside pine zone today is due primarily to the intensive logging of the early 20th century. However, the abundance of 50-80 year old trees is the result of a complex of interactions among early high-grade logging, grazing, fire exclusion, some unique climatic events in the early 20th century, and the forest management that has been practiced in these forests since about 1950. These interactions, and the current conditions in ponderosa pine forests of the South Central Highlands Section, are discussed below.

Table II-9. Age class distribution of canopy trees in the Hermosa area north of Durango, CO, in 1993. This area has never been logged, though fires have been suppressed.

Age Class (years at breast height)	Trees / ha	Cumulative % of Trees of This Age or Younger
1 - 20	0	0
20 - 40	0.5	1
40 - 60	1.0	3
60 - 80	6.0	17
80 - 100	7.5	33
100 - 120	1.0	36
120 - 140	2.0	40
140 - 160	6.0	53
160 - 180	8.5	72
180 - 200	4.5	82
200 - 220	1.5	86
220 - 240	2.0	90
240 - 260	1.0	92
260 - 280	0.5	93
280 - 300	1.0	96
300 +	2.0	100

Indirect Effects of Early Logging:

Many or even most of the ponderosa pine forests throughout the South Central Highlands Section today have a dense canopy of similar-sized and similar-aged pine. Only in a few places does one find pine stands with trees of many sizes (including very large) and multiple canopy layers. This homogeneous forest structure provides good habitat for some wildlife species, e.g., turkey, elk, and deer. However, many of the species that could potentially inhabit the area probably are absent, or at unusually low densities because of unsuitable habitat. For

example, many species of cavity-nesting birds do best in old stands containing large ponderosa pine or Douglas-fir trees. There are only a few threatened or endangered plant species in the ponderosa pine forests of the South Central Highlands Section, but many herbaceous plant species probably are greatly reduced in numbers or even locally extirpated because they cannot tolerate the shade and deep organic litter on the forest floor of the extensive, homogeneous ponderosa pine forests that now dominate these areas.

Table II-10. Ratio of the density of recently dead trees to the density of living trees of different size classes in the 180,000 acre westside pine zone of the San Juan National Forest as of 1994. Data from RIS files, San Juan National Forest

Diameter Class (inches at breast height)	Ratio: Dead Trees / Acre Live Trees / Acre
1 - 4	1.7
4 - 8	1.5
8 - 12	0.4
12- 16	0.2
16 - 20	0.2
20 - 24	0.2
24 - 28	0.8
28 - 32	1.1
32 - 36	0.9

An important component of forest wildlife habitat that can be quantified with existing data is the availability of nesting sites for cavity-nesting birds. Secondary cavity nesters (birds that cannot excavate their own holes, so must nest in natural cavities or holes created by woodpeckers and other primary cavity nesters) make up approximately one-third of breeding bird species in ponderosa pine forests, and may comprise 40 - 55 % of all breeding pairs. Examples of birds that depend heavily on snags include the violet-green swallow, pygmy nuthatch, western bluebird, and mountain chickadee (Balda 1975). Researchers have recommended minimum suitable tree diameters for cavity nesters, ranging from 10 in (25 cm) (Reynolds et al. 1985) to 20 in (50 cm) (Horton and Mannan 1988; also see Scott 1978, Cunningham et al. 1980). Minimum recommended densities of snags range from 1.73 - 2.68 snags/acre (Balda 1975) to 5.2 snags/acre (Cunningham et al. 1980; also see Scott 1978, 1979). It has been suggested recently that these recommended densities are higher than actually existed prior to the era of fire exclusion in ponderosa pine forests, and higher than is really necessary to maintain functional communities; research is underway to better characterize snag requirements and snag dynamics. Nevertheless, we

based this assessment on the current recommendations for snag densities.

We determined the availability of snags within the westside pine zone, north of Dolores, Colorado, where industrial-scale logging occurred in the 1920s - 1940s (described above), using RIS data from the San Juan National Forest. Three classes of trees that could provide cavity-nesting habitat were: large living trees with broken tops, large hard snags (i.e., undecomposed standing dead stems), and large soft snags (partially decomposed standing dead trees). Using even the lowest recommendations for suitable tree diameter and snag density, and including live trees with broken tops (which usually are not included in snag densities), we found that the availability of cavity-nesting trees in the westside pine zone is well below any recommended level (Table II-11). Live trees >25 cm (10 in) dbh with broken tops were found to be only 0.1 tree/acre; soft snags >25 cm (10 in) dbh were 0.3 stems/acre; and hard snags >25 cm (10 in) dbh were 0.04 stems/acre. These results indicate that all classes of cavity-nesting trees combined, amount to only 0.44 stems/ha in the westside pine zone - - compared to recommended densities of at least 1.73 snags/acre.

Table II-11. Availability of snags in the 180,000 acre westside pine zone of the San Juan National Forest as of 1994 (data from RIS files, San Juan National Forest). Recommended snag densities are from the literature (see text).

Snag Category	Snags / Acre
Snags > 10 inches dbh with broken tops	0.1
Soft (decayed) snags > 10 inches dbh	0.3
Hard snags (not decayed) > 10 inches dbh	0.04
Total snags (all categories)	*0.44*
Minimum recommended snag density (all categories)	*1.73*

The scarcity of cavity nesting sites in these forests suggests that bird populations are probably well below the numbers that may have been sustained during the period of indigenous settlement. This is significant not only in relation to the aesthetic pleasures of seeing and hearing wild birds, but also in relation to overall forest health. Woodpeckers, for example, are primary cavity nesters, and one of their preferred foods is bark beetles (a current threat to ponderosa pine forests in the region; see below). There is some evidence that woodpecker predation can retard or even prevent mountain pine beetle outbreaks. Studies also have shown that predation by small insect-eating birds, such as warblers, can substantially reduce populations of plant-injuring insects. In the ponderosa pine forests studied by Balda (1975) in northern Arizona, 81% of the secondary cavity nesting species were insectivorous, and many fed on

insects not only in summer but also throughout the winter when some insect populations may be especially vulnerable to predation. Thus, the lack of snags in ponderosa pine forests of the South Central Highlands Section may interact with other current stand conditions (tree density and fire exclusion; see below) to impair the natural regulation of insect populations and increase the risk of devastating insect outbreaks.

Another indirect effect of early logging in ponderosa pine forests is related to present-day timber supplies. Large diameter, high-quality saw logs are becoming increasingly scarce in southwestern Colorado, and throughout the west. Given current ecological and socio-political conditions, it seems unlikely that such material will become much more available for a long time. At the same time, we have excessive quantities of small diameter material (anonymous, undated report entitled "Changing conditions in southwestern forests and implications on land stewardship," USDA Forest Service, Southwestern Region). There is a great need and opportunity now to create new technologies and markets that can make use of these smaller diameter logs, as described below.

The legacy of early high-grade logging will be with us for a long time. One cannot quickly re-grow large, old trees and snags. The best means now available for increasing the availability of suitable nesting cavities for birds and bats probably is to (i) protect existing snags and large trees from damage by fire, logging, and firewood gathering, and (ii) increase the numbers of very large trees. As these large trees gradually age and die, they will create good nesting conditions. These ideas are developed more fully in Chapter VII.

2. <u>Legacies of Excessive Livestock Grazing in the Late 1800s and Early 1900s</u>

Fleischner (1994;630) writes that "Livestock grazing is the most widespread influence on native ecosystems of western North America." Since almost no areas escaped heavy livestock grazing in the late 1800s and early 1900s, we have no good reference areas against which to measure specific effects and legacies of early, uncontrolled grazing. The few ungrazed sites that have been found in the southwest usually are rugged, inaccessible, and small in extent -- and therefore not representative of the pre-grazing landscape as a whole. It is difficult, therefore, to reconstruct the structure, composition, and dynamics of herbaceous plant communities in the South Central Highlands Section prior to the arrival of great numbers of sheep and cattle in the late 1800s, or to quantify the changes brought about by early livestock grazing. It is also difficult to disentangle the specific effects of grazing from effects of fire exclusion and climatic variability, since, as Leopold (1924) observed early on, all of these influencing agents have been operating together and have interacted with one another to produce the conditions that we see today.

Nevertheless, we do know that the impact of early grazing on herbaceous plant communities throughout the west was substantial (Fleischner 1994, Belsky and Blumenthal 1997). Since those early days, livestock have been regulated on the public lands, and techniques of

sustainable grazing have been developed for both public and private lands; indeed, grazing can be a valuable tool of land stewardship when conducted wisely (Knight 2002, White 2008). However, many of the legacies of the uncontrolled grazing that occurred in the late 1800s and early 1900s are with us still.

Timing and Intensity of Early Livestock Grazing:

Tewa Puebloan people may have had livestock in portions of the Jemez and Nacimiento Mountains (Figure I-1) as early as the late 1600s, though this is not yet well documented (Touchan et al. 1996). However, it is fairly certain that Hispanic settlers were grazing animals by the mid-1700s or even earlier in other portions of the Jemez and Nacimiento Ranges (Cerro Pedernal site, Touchan et al. 1996). The Navajo people began grazing sheep and goats as early as the mid-1700s in the Chuska Mountains of northwestern New Mexico and northeastern Arizona (Savage 1991) and in portions of the Jemez and Nacimiento Ranges (Continental Divide site, Touchan et al. 1996). Prior to the Civil War, there were more livestock, mainly sheep, in the upper Rio Grande region of New Mexico than anywhere else in the southwest (Denevan 1967). Exact numbers of grazing animals in any particular area prior to the late 1800s are unknown, but they apparently were high at some times and in some places. According to one estimate, there were half a million sheep on the Navajo Reservation in 1846 (Savage 1991;281). Ponderosa pine forests may have been favored locations for grazing sheep and other livestock, because they offered excellent pasturage, water, and isolation from raiders (Savage 1991).

The ecological effects of Native Americans' and early Hispanic settlers' livestock in ponderosa pine forests of the South Central Highlands Section may have been locally intense, but they apparently did not affect all of the forests in the region. Ubiquitous impacts of heavy livestock grazing were not felt until the 1870s and 1880s, when Anglo settlers began to arrive in increasing numbers (Savage 1991). Development of railroads also facilitated rapid increases in numbers of livestock. Denevan (1967;696) reports that the number of sheep in New Mexico rose from several hundred thousand to 4 - 5 million in the 1880s. In the northern portions of the South Central Highlands Section (e.g., the San Juan National Forest and Uncompahgre Plateau), major effects of livestock grazing began in the 1870s and 1880s.

Cattlemen arrived in the Dolores River valley in the mid-1870s and were the first homesteaders in that area (Dishman 1982). Charles Goodnight brought some 1500 head of cattle to the Pagosa Junction area in 1875; the Pine - Vallecito River country began to be settled by ranchers in the late 1870s; ranches were established in Montezuma County beginning in 1878; and cattle were brought into the Florida - Pine River areas in 1879 (Anonymous, Forest History, Volume 1). The first ranches in the Uncompahgre country were established in the late 1870s but livestock grazing began to increase rapidly after expulsion of the Ute people in 1881; by 1890 there were reported to be 30,000 cattle in the Norwood area (Rockwell 1999;74). Ralph Shaw, a Forest Ranger from 1907 - 1920, reported that there were several large herds of 10,000 or more head, as well as numerous smaller herds up to 5,000

animals in the San Juan Mountain country during the 1880s and 1890s (Anonymous, Forest History, Volume 1). Evidently the major livestock in most of the San Juan Mountains and Uncompahgre Plateau in the late 1800s were cattle. Sheep were not abundant in the Dolores area until after 1900 (Dishman 1982) or on the Uncompahgre Plateau until 1915 (Rockwell 1999;55). DuBois (1903) indicates that cattle were the major grazers in ponderosa pine forests, at least in the area around Pagosa Springs, and that sheep were grazed primarily at the higher elevations in summer and below the forest zone in winter.

Heavy livestock grazing continued into the 20th century. As homesteads occupied more and more of the productive river bottom lands, stockmen made more extensive use of upland forests and meadows in the national forests. DuBois (1903;11) estimated that 268,000 sheep were being grazed in the San Juan Mountains, ranging from the foothills to the crest of the range over the course of a summer season. He also estimated 19,160 cattle in the country around Pagosa Springs and upper Rio Grande valley (DuBois 1903;17). In 1931, there were reported to be 19,000 cattle and 149,000 sheep grazing on the Montezuma and San Juan National Forests (Anonymous, Forest History, Volume 1). However, all of the commentators of the time emphasized the difficulty of obtaining accurate estimates of the total number of livestock in the area, due to inadequate record-keeping by officials and the number and diversity of livestock operators. Also, because of an obvious bias against Hispanic sheepherders on the part of the predominantly Anglo writers of the time, the number of animals being herded by Hispanic people may have been either downplayed or exaggerated significantly, depending on the particular conclusions being advanced by the writer.

The early grazing was largely unregulated, and soon became excessive in many areas (Dishman 1982). Range conditions reportedly began deteriorating in the 1890s (Rockwell 1999;53), and this degradation accelerated over extensive areas between 1904 and 1920 (Anonymous, Forest History, Volume 1). At the beginning, in the 1880s, a quarter-section of grassland was thought sufficient to support five or six head of cattle; as individual sites became overgrazed, cattlemen simply found new unexploited grassland areas for their animals. However, a shortage of grazing lands was reported in the Dolores Valley area as early as the late 1880s and 1890s, due to homesteading and overgrazing. Sheep were introduced in the Dolores country around 1910, in part because "sheep could graze on winter ranges and depleted grasslands that could no longer support cattle (Dishman 1982;32). In 1903, DuBois (1903;8) wrote: "Although at present sheep raising brings more money into the country than any of the others [industries], unless regulated it will soon destroy the summer range and have to be given up...." DuBois (1903;17) also described 11 out of 31 valleys as either "plainly overstocked" or "showing signs of overstocking" with cattle.

Ecological Impacts of Early Heavy Livestock Grazing:

What were the specific ecological effects of early livestock

grazing in ponderosa pine forests of the South Central Highlands Section? We will never know fully. However, we can gain some insights from the commentaries of people who were present at the time. DuBois (1903;11) wrote of early cattle grazing: "In low gulches and flats where large herds of cattle have been ranged year after year, the bunch grass is pulled up by the roots and gradually dies out. On overstocked cow ranges the white roots of bunchgrass can be seen all over the ground." He goes on to say that cattle grazing has had little impact on the vegetation of high-elevation forests, but "In the pine type, where the range is blue-stem grass, cattle do good [enhance tree reproduction] by keeping down the grass and preparing a seedbed. They never touch pine seedlings." (DuBois 1903;18).

Even though comparisons of grazed areas with ungrazed relict areas are not ideal, since the ungrazed areas usually are at least somewhat different from the grazed areas in terms of environment and history, these kinds of comparisons provide some our only current opportunities to elucidate grazing-induced changes (Belsky and Blumenthal 1997, Paulson and Baker 2006). Madany and West (1983) studied two small, inaccessible mesas in southern Utah that had never been grazed, and compared them with an extensive plateau area nearby that had a long history of livestock grazing. All three study areas were at comparable elevations and supported forests of ponderosa pine with Gambel oak understory. Cover of graminoids and forbs combined was 49% on the ungrazed mesas but only 5% in the grazed area. Herbaceous plants were co-dominant with the woody plants (shrubs

and tree saplings) on the ungrazed mesas, but only sparsely represented in the woody-dominated understory of the grazed area. Grazing did not appear to have completely extirpated any herbaceous species, although at least one species (*Muhlenbergia montana*, a warm-season bunchgrass) was present in the grazed area only in a spot where grazing pressure was lighter than in the area as a whole.

In an earlier study, Rummell (1951) compared the herbaceous plant communities in open ponderosa pine forests growing on two small basalt mesas in central Washington. Both were dominated by unlogged ponderosa pine forest, but one had been heavily grazed for 40 years while the other was ungrazed. Total percent ground cover in the grazed forest was less than half the cover in the ungrazed forest. Species richness was 1/3 greater in the ungrazed forest, and a dense growth of graminoids (especially *Calamagrostis rubescens* and *Carex geyeri*) created a distinctive hummock type of vegetation in portions of the ungrazed forest that was not present in the grazed forest. Total graminoid cover and biomass in the grazed forest were less than one-third the cover and biomass in the ungrazed forest. Arnold (1950) observed similar compositional differences in grazed and ungrazed (exclosures) ponderosa pine forests near Flagstaff, Arizona. The plants most sensitive to grazing were highly palatable bunchgrasses (*Muhlenbergia montana, Festuca arizonica, Poa fendleriana*, and *Blepharoneuron tricholepis*); these decreased rapidly under heavy grazing. Grasses that were more resistant to grazing (e.g. *Boutelous gracilis, Sitanion hystrix Sporobolus interruptus, Aristida fendleri*, and *A. arizonica*), as well as

several unpalatable perennials and weeds, increased in importance as the bunchgrasses declined under continual grazing pressure.

In sum, heavy livestock grazing, especially that which occurred in the late 1800s and early 1900s, has left a long-lasting legacy of altered plant community composition in ponderosa pine forests throughout the South Central Highlands Section. Some highly palatable species, such as the bunchgrasses *Muhlenbergia montana* and *Festuca arizona,* may have been locally extirpated from many stands, and the relative abundance of surviving species has been drastically altered. Grazing, along with fire exclusion and some climatic events (see below), led to a shift from dominance or co-dominance of herbs in the ground layer of ponderosa pine forests to an overwhelming dominance of woody vegetation (shrubs and tree saplings) in many areas. We may never know the pre-grazing composition of many ponderosa pine forests, though we can make some reasonable guesses (see below). One more key legacy of early, heavy livestock grazing is the virtual elimination of low-severity surface fire as a dominant ecological process in ponderosa pine forests of the southwest. This effect is described in the next section.

3. Legacies of Fire Exclusion

The formerly frequent occurrence of widespread, low-intensity fire ended abruptly around 1880 in most ponderosa pine forests throughout the Southwest (Swetnam and Baisan 1996). A dramatic reduction in fire frequency occurred earlier in some areas, e.g. in the mid-1700s in portions of Jemez Mountains (Touchan et al. 1996), and in the 1830s in the Chuska Mountains of northwestern New Mexico and northeastern Arizona (Savage and Swetnam 1990, Savage 1991). Also, reduced fire frequency was not apparent until much later -- the middle of the 20th century -- in isolated portions of El Malpais in west-central New Mexico (Grissino-Mayer 1995). Despite these exceptions, however, most ponderosa pine forests of the southwest -- including the South Central Highlands Section -- experienced a sudden and remarkably synchronous alteration of fire regimes in the latter part of the 19th century (Covington and Moore 1994, Swetnam and Baisan 1996).

Small fires continued well into the 20th century in many areas, indicating that lightning and humans were still igniting fires. The major change, however, was that these fires were nearly all very small. The extensive fires of the previous centuries were no longer a part of the landscape. For example, we documented a few wildfires in the 20th century in several of our study sites in the San Juan National Forest, as well as a prescribed fire within the last decade at the Hermosa site (Figures II-1 and II-2). However, none of these fires was extensive, and the intervals between even these small fires have been significantly longer than the intervals between extensive fires during the reference period. Over most of the ponderosa pine zone in the South Central Highlands Section, there were no extensive fires in over 100 years -- prior to the Cerro Grande fire in 2000, and the Missionary Ridge, Million, and Burned Canyon fires of 2002.

What caused this abrupt cessation of formerly frequent and

extensive fires in ponderosa pine forests of the southwest? It was not Smoky Bear, because organized and effective fire control programs were not instituted over large areas until well into the 20th century (Pyne 1982). The current consensus among researchers is that the single most important cause of the initial alteration of fire regimes in the southwest was the introduction of large herds of livestock by Euro-American settlers. The animals grazed off the grasses and herbs that formerly carried light fires through the forest. Lightning and humans still started fires, but they could not spread across bare ground. Throughout most of the southwest region, the cessation of frequent, extensive fires coincided with the arrival of large herds of cattle and sheep in the late 1800s (Swetnam and Baisan 1996). In those areas where livestock grazing began earlier than the late 1800s, e.g., in portions of the Jemez and Chuska Mountains, fire frequencies in ponderosa pine forests also decreased at about the same time (Touchan et al. 1996, Savage and Swetnam 1990, Savage 1991). Well into the 20th century, heavy grazing was seen as a tool for reducing fire danger in southwestern forests (e.g., Leopold 1924).

Grazing began to be regulated more effectively on federal lands beginning in the 1930s (e.g., with the Taylor grazing act of 1934), but by this time the Forest Service was developing an effective fire suppression capability. The "10 am policy", which sought to control any wildfire by 10 am the next morning after it was discovered, was introduced in the late 1930s (Pyne 1982). In retrospect, it now appears that it was a mistake to put out all of the fires in ponderosa pine forests, but at that time it was commonly believed that all

fires were a threat to the forests. We now know that by suppressing all fires we simply created the potential for even more severe fires in the future (Covington and Moore 1994).

When fires occur today in ponderosa pine forests, they sometimes are much hotter and destructive than they were before 1880, because of the great quantities of dead pine needles, branches, and other heavy fuels that have accumulated during 100+ years without fire. Cooper (1960) determined from early written accounts and interviews with "old-timers" that crown fires were rare in Arizona ponderosa pine forests before the turn of the 20th century. However, several recent fires in ponderosa pine forest have been destructive crown fires. Fires of the intensity and extent of some recent burns probably are unprecedented when compared to fire behaviors during the reference period. For example, portions of the 2000 Cerro Grande and 2002 Missionary Ridge fires may have been unprecedented with respect to the size of patches of high-severity burn -- though other portions of the Cerro Grande and Missionary Ridge fires burned in a manner similar to pre-1870 fires. Savage and Mast (2005) sampled ponderosa pine forests in Arizona and New Mexico that had burned as crown fires in the late 1940s through mid 1970s, and found that most of these areas were either regenerating as dense ponderosa pine stands (vulnerable to another stand-replacing fire) or had been converted at least temporarily to non-forested grass or shrub communities.

Fires also are becoming increasingly difficult to control. Indeed, wildfire statistics show that over the last decade the amount of acreage burned each year on national forests in New

Mexico and Arizona has been steadily rising -- despite continued, aggressive fire suppression (Moir et al. 1997). This is due in part to the dry summers we have experienced recently, but another important reason is the fuel conditions that now exist in many ponderosa pine forests (Covington and Moore 1994). A more recent analysis by Westerling et al. (2006) reveals a dramatic increase in the frequency of large fires since the mid-1980s throughout the western U.S. This increase is associated with earlier springs, earlier snow-melt, and longer fire seasons, and may be a harbinger of what the future holds with global climate change. Thus, the issue of fuel conditions in ponderosa pine forests of the South Central Highlands Section (and elsewhere) is even more pressing as we anticipate ever more fires in the future.

Thus, an important legacy of the last century of human activity is the absence of what was once a key ecological process -- fire. Ironically, the ponderosa pine forests of the South Central Highlands Section, which were profoundly shaped by an environment containing frequent fire, are now seriously at risk from uncontrollable and damaging wildfires.

4. Interactions: Logging, Grazing, Fire, Climate, and Today's Forests

A unique combination of historical events -- the elimination of extensive fires, removal of grass competition by livestock, and periods of unusually wet weather coinciding with good cone crops -- all came together in the early decades of the 20th century, and allowed an abundant cohort of ponderosa pine seedlings to become established throughout the Southwest (Covington and Moore 1994). This early 20th century age class dominates many ponderosa pine forests today (Table II-8), and is conspicuous even in the rare stands that have never been logged (e.g., Table II-9). At about the same time that this "irruption" of young trees (Covington and Moore 1994) was occurring, the older trees were being removed by high-grade logging. Thus, many or even most of the ponderosa pine forest stands present today in the South Central highlands Section have a relatively uniform stand structure: they tend to be relatively dense, most of the trees are small to medium sized, and most are 70 - 100 years old (e.g., Table II-12). In two of the stands where we sampled density and dispersion of stumps to reconstruct the canopy structure of pre-1900 forests -- Five Pine Canyon and Plateau Creek (Tables II-1 and II-6) -- the current canopy tree density is about seven times greater than before 1900. The strong clumping pattern of the forests during the reference period also has been largely lost in today's dense stands. Trees that established after 1900 have filled in many of the former open spaces between clumps, and the trees are thicker in the areas of the former clumps as well.

In addition to the changes just described for ponderosa pine forests subjected to grazing, logging, and fire exclusion in the late 19th and early 20th centuries, ponderosa pine has expanded into grasslands and shrublands in many areas during the past hundred years. In an analysis of 146 pairs of historic and modern photographs, Zier and Baker (2006) found that in about of half the photos showing an ecotone between ponderosa pine forest and either grassland or shrubland, the forest edge had moved outward during the period

between photos. Some of this forest expansion was probably due to effects of grazing and fire exclusion, but it was seen even in locations where grazing and fire exclusion were likely minor factors - suggesting that climate changes during the 20th century were a major mechanism driving the expansion of ponderosa pine forests (Zier and Baker 2006).

Table II-12. Age class distribution of canopy ponderosa pine trees in the Plateau Creek study site (Table II-1) in 1994. This 0.25-ha stand was heavily logged between 1924 and 1948.

Decade of Germination	Trees / Hectare	% of Total Trees
1830	10	
1840	10	
1850	20	
1860	20	14
1870	10	
1880	10	
1890	10	
1900	270	
1910	80	
1920	90	83
1930	80	
1940	20	3
Total	*630*	*100*

Forest management in many areas since about 1950 has emphasized selective logging of these ponderosa pine forests that are developing after the unique historical events of the early 1900s. By continually removing the larger trees as they enter the saw-timber size classes, current management in many areas tends to perpetuate a stand structure of relatively homogeneous, small, dense trees, a sparse herb stratum, and a thick organic litter layer on the forest floor. This strategy may maximize wood fiber production, and the resultant forests are aesthetically appealing to many people, but there is a growing sense that the ponderosa pine forests covering much of the South Central Highlands Section today are at risk.

One consequence of the high density and small tree size of many ponderosa pine stands is that the forests are vulnerable to outbreaks of insects and disease. The most serious threat is the mountain pine beetle, which has experienced dramatic outbreaks elsewhere in the west in recent years (Schmid and Mata 1996, Bentz et al. 2009, Raffa et al. 2008). In the 180,000-acre westside pine zone of the San Juan National Forest, nearly 40,000 acres (41% of the lands that were inventoried) recently were rated at moderate or high

risk for a major mountain pine beetle outbreak (Angwin and Raimo 1994:5). Some 58,000 acres (59 % of inventoried lands) were rated at low risk (Table II-13). However, Angwin and Raimo (1994) emphasized that their method, developed originally for the Black Hills region in South Dakota, probably underestimates actual beetle risk. A second, less intensive, inventory conducted in 1994 suggested that most of the study area actually may be in the moderate to high risk category (Angwin and Raimo 1994:6). The risk rating is based primarily on stand density and tree diameter; dense stands of small trees usually receive a relatively high rating because the trees are under stress and therefore vulnerable to attack by the beetles. The lack of snags and suitable habitat for cavity-nesting birds and bats may also be holding populations of insectivores at abnormally low levels, thereby increasing the likelihood of outbreaks.

Table II-13. Mountain pine beetle risk in the 180,000 acre westside pine zone of the San Juan National Forest, based on stand structure, as of 1994 (data from Angwin and Raimo 1994).

Risk Rating	Number of Acres
Low Risk	57,000
Moderate Risk	31,000
High Risk	8,000
Not Inventoried	19,000
Total	*115,000*

The forests also are at risk of uncontrollable, destructive wildfires. As described above, fuel loads appear to be substantially higher in many stands than they would have been before 1870, because of the lack of recent, extensive fires. The fuels are comprised of un-decomposed litter and duff on the forest floor, as well as the closely-spaced crowns of trees and shrubs. If fires are ignited today under dry weather conditions, there is a greater probability of the fire burning the canopy and inflicting significant tree damage and mortality than was usual in pre-1870 fires. The risk of personal property loss is also rising as expensive homes are built in ponderosa pine forests on private lands adjacent to National Forest lands (Theobald and Romme 2007). The heightened risk of destructive forest fire in ponderosa pine forests of the South Central highlands was brought home to local residents during the severe fire seasons of 2000 (e.g., Cerro Grande fire) and 2002 (e.g., Missionary Ridge and Million fires).

A final concern about the current age and size class structure of ponderosa pine forests in the South Central Highlands Section has to do with pine regeneration -- or lack thereof. In the westside pine zone of the San Juan National Forest, less than about 20% of

the pine stands for which data are available have adequate regeneration (defined as 50+ trees/acre with dbh less than or equal to 2 inches (5 cm)). Regeneration of pine in pine-oak stands has been a problem throughout southern Colorado, since the early years of the 20th century (e.g., DuBois 1903). The reasons why pine regenerates poorly are not well understood, but they probably involve seedling suppression by the thick organic layer on the forest floor, competition from oak and other perennial plants (Harrington 1985, 1987), and possibly poor seed crops or drought in the years after good seed crops.

D. SUMMARY

Ponderosa pine forests are found at lower to middle elevations throughout the South Central Highlands Section. Ponderosa pine (*Pinus ponderosa* var. *scopulorum*) is the major canopy species, often with Gambel oak (*Quercus gambelii*) dominating the understory. Many ponderosa pine forests were composed historically of open stands of large trees, but denser stands also were present. Pre-1900 canopy densities and basal areas in open stands in the San Juan Mountains and Uncompahre Plateau were generally higher than in ponderosa pine forests of northern Arizona. Trees typically grew in clumps interspersed with open areas, and there was wide variation in the size of clumps and open areas.

The most important natural disturbance process in ponderosa pine forests prior to Euro-American settlement was fire; other natural disturbances included various tree-killing insects (notably bark beetles), pathogens, and parasites. Located geographically between the Southwest and the Colorado Front Range, historical fire regimes of ponderosa pine forests in the South Central Highlands Section were intermediate between the Southwestern model of frequent, low-severity fires and the variable-severity model of the Front Range which included a component of high-severity fire. Major fire years in the South Central Highlands Section, as in the Southwest, usually occurred when one to three very wet El Niño years -- in which abundant fine fuels were produced -- were followed by a very dry La Niña year -- when the accumulated fine fuels dried out and carried extensive fires. Approximately half of documented fires before 1900 in the San Juan Mountains burned in the summer season, and half burned in dormant seasons (spring or fall). However, in the years with very extensive fires (e.g., 1748, 1820, 1851, and 1879), fires apparently burned all summer long.

After about 1870 (earlier in some places), most of the ponderosa pine forests in the region were altered dramatically by livestock grazing, logging, and fire suppression by Euro-American settlers. By the mid-20th century, nearly all of the old growth ponderosa pine forests of this region had been liquidated. Consequently, unlogged, old-growth ponderosa pine stands today are very few in number. The most obvious legacy of the early logging of ponderosa pine forests is the general lack of large, old trees and snags in these forests today; most stands are dominated by relatively small, young trees. Early livestock grazing was

largely unregulated, and range conditions began deteriorating as early as the 1890s. As a result, some highly palatable species, such as the bunchgrasses *Muhlenbergia montana* and *Festuca arizona,* may have been locally extirpated from many stands, and the relative abundance of surviving species has been drastically altered. Heavy livestock grazing also disrupted the historical fire regime as the animals removed the grasses and herbs that formerly carried light fires through the forest. Lightning and humans still started fires, but they could not spread across bare ground. Grazing began to be regulated more effectively on federal lands beginning in the 1930s, but by this time the Forest Service was developing an effective fire suppression capability. Over most of the ponderosa pine zone in the South Central Highlands Section, there were no extensive fires throughout the 20th century. When fires occur today in ponderosa pine forests, they sometimes are more severe than before 1900, in part because of the greater fuels that have accumulated during 100+ years without fire. Earlier springs, earlier snow-melt, and longer fire seasons since the mid-1980s also have contributed to larger and more severe fires in ponderosa pine and other vegetation types throughout the West.

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CHAPTER III: SPRUCE-FIR FORESTS

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A. VEGETATION STRUCTURE AND COMPOSITION

Forests at the highest elevations in the South Central Highlands Section are dominated by Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*). The taxonomic status of the fir in this region is somewhat problematic: one can find individuals that look like the corkbark fir (*Abies lasiocarpa* var. *arizonica*) of New Mexico and Arizona, others that resemble *Abies lasiocarpa* var. *lasiocarpa* of northern Colorado and the northern Rocky Mountains, and still others that appear intermediate between these two forms. Because the San Juan Mountains are at the boundary between the two varieties of subalpine fir, apparently both grow here and probably interbreed as well. In this report, we refer simply to *A. lasiocarpa* (or *A. bifolia* Murray, as recommended in the new Flora of North America).

Spruce-fir forests are the coldest and wettest forest type in the South Central Highlands Section, occurring at elevations of 2,720 – 3,580 m. They are associated with the subalpine climatic zone, the cryic soil temperature regime, and the udic soil moisture regime.

Spruce-fir forests cover a large portion of the high slopes, ridges, and valleys of the South Central Highlands Section, and because of underlying environmental variation throughout this extensive habitat, the forests are quite variable in composition and structure (DeVelice et al. 1986, Spencer and Romme 1996). On the eastern side of the continental divide, where winter snowfall is generally reduced by a rain shadow effect, some stands lack the subalpine fir component (e.g., in the vicinity of Creede, CO). These stands may be pure Engelmann spruce, a mix of Engelmann spruce and bristlecone pine (*Pinus aristata*), or even pure bristlecone pine on very dry sites. On the west side of the continental divide, nearly all stands contain both spruce and fir, and bristlecone pine is rare or absent. Throughout the South Central Highlands Section, however, fir gradually becomes less abundant as one approaches upper timberline, such that stands just below timberline tend to be predominantly spruce (Alexander 1974). On warmer, drier sites at lower elevations, a warm-dry phase of spruce-fir forest can be recognized where species from the mixed conifer zone, especially Douglas-fir (*Pseudotsuga menziesii*) and white fir (*Abies concolor*), grow alongside Engelmann spruce and subalpine fir. Quaking aspen (*Populus tremuloides*) also occurs in many places throughout the spruce-fir forest type.

Near alpine timberline, spruce and fir typically grow sparsely, and often in a stunted, wind-shaped growth form called krummholz. A recent study using repeat photography documented some 20th-century establishment of spruce and fir in areas above the previous timberline, though overall the magnitude of change was small (Zier and Baker 2006).

The ground layer vegetation also is highly variable throughout the spruce-

fir forest type. A rich mixture of mesophytic herbs (e.g., *Anticlea elegans, Aquilegia elegantula, Arnica cordifolia, Artemisia franserioides, Bromopsis canadensis, Carex geyeri*, *Erigeron eximius*, *Fragaria vesca, Geranium richardsonii*, *Goodyera oblongifolia, Lathyrus leucanthus, Ligusticum porteri*, *Luzula parviflora, Maianthemum stellatum, Mertensia ciliata*, *Oreochrysum parryi, Orthilia secunda, Osmorhiza depauperata, Pedicularis racemosa, Pyrola minor,* and *Viola canadensis*) and shrubs (e.g., *Lonicera involucrata, Ribes montigenum, Rubacer parviflorum, Sambucus microbotrys,* and Vaccinium myrtillus*) may be present, but composition varies among, and within, sites. Table III-1 depicts the composition of the ground layer in an old-growth spruce-fir forest, growing on a relatively moist site at 3,300 m, in the central San Juan Mountains (Romme et al. 1996). The canopy and understory of this stand are composed almost exclusively of Engelmann spruce and subalpine fir, with spruce exceeding fir in terms of both stem density and basal area (Table III-2).

Table III-1. Ground layer species having at least 1 % cover, and percentage of sample plots in which each species was recorded at >1%, >5%, and >25% (n = 49), in an old-growth spruce-fir forest growing at 3,300 m near the head of Martinez Creek in the San Juan National Forest.

Species	Percent of Plots Where Cover >1 %	Percent of Plots Where Cover >5 %	Percent of Plots Where Cover >25%
Aconitum columbianum	2	0	0
Actaea rubra ssp. arguta	53	0	0
Androsace septentrionalis	2	0	0
Alnus tenuifolia	2	0	0
Aquilegia elegantula	20	0	0
Arnica cordifolia	65	2	0
Bromopsis ciliata	69	0	0
Caltha (Psychrophila)leptosepala	4	0	0
Cardamine cordifolia	16	4	0
Carex geyeri	43	0	0
Carex spp (other than C. geyeri)	33	0	0
Corydalis caseana	6	0	0
Corallorhiza maculata	2	0	0
Corallorhiza trifida	2	0	0
Delphinium barbeyi	41	10	0
Epilobium (Chamerion) angustifolium	90	0	0
Epilobium hornemannii	14	0	0
Erigeron eximius	65	20	0
Fragaria spp (americana and ovalis)	96	0	0
Galium boreale	8	0	0
Geranium richardsonii	100	12	
Goodyera oblongifolia	50	0	0
Haplopappus (Oreochrysum) parryi	31	2	0

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Heracleum (lanatum) sphondylium ssp. montanum	10	0	0
Lathyrus leucanthus	90	4	0
Ligusticum porteri	82	8	0
Listera cordata	10	0	0
Lonicera (Distegia) involucrata	98	0	0
Luzula parviflora	57	0	0
Mertensia ciliata	100	6	0
Mitella spp (pentandra and stauropetala)	29	0	0
Moneses uniflora	27	0	0
Osmorhiza obtusa (depauperata)	100	0	0
Oxypolis fendleri	29	2	0
Pedicularis racemosa	86	0	0
Polemonium occidentale	2	0	0
Potentilla pulcherrima	4	0	0
Pseudocymopterus montanus	65	0	0
Pyrola minor	76	0	0
Ranunculus inamoenus	2	0	0
Ribes montigenum	59	0	0
Ribes spp (wolfii and coloradense)	98	2	0
Rosa woodsii	2	0	0
Rubus parviflora	63	2	0
Sambucus microbotrys	14	0	0
Saxifraga odontoloma	6	0	0
Senecio triangularis	20	0	0
Smilacina racemosa (Maianthemum amplexicaule)	2	0	0
Smilacina (Maianthemum) stellata	73	0	0
Sorbus scopulina	2	0	0
Streptopus amplexifolius	86	0	0
Symphoricarpos oreophilus	20	0	0
Taraxacum officinale	33	0	0
Thalictrum spp (fendleri and sparsiflorum)	88	0	0
Vaccinium myrtillus	86	57	10
Valeriana capitata	2	0	0
Veratrum californicum (tenuipetalum)	2	0	0
Vicia americana	18	0	0
Viola spp (canadensis and adunca)	94	0	0
Zygadenus (Anticlea) elegans	57		

Table III-2. Density, basal area, and diameter at breast height of living, standing dead, and fallen dead trees, and ages at breast height of canopy and understory trees, in an ancient forest growing at 3,300 m near the head of Martinez Creek, San Juan National Forest, Colorado. Numbers under each tree species are means, with standard deviations in parentheses and sample size indicated by n. The column titled P indicates the significance of the difference between spruce and fir, based on the Wilcoxon signed rank test for density and basal area, and on a pooled variance t-test for diameters and ages. Species were combined for density and basal area of fallen dead trees because of difficulty in identifying boles in advanced stages of decomposition.

	Picea engelmannii	*Abies lasiocarpa*	P
Density of living canopy trees (stems/ha)	201 (84) n = 49	121 (90) n = 49	0.000
Density of living understory trees (stems/ha)	478 (360) n = 49	328 (265) n = 49	0.024
Density of standing dead canopy trees (stems/ha)	15 (15) n = 49	20 (24) n = 49	0.138
Density of fallen dead trees, all species combined (stems/ha)	204 (67) n = 49		
Basal area of living canopy trees (m^2/ha)	28.63 (13.88) n = 49	11.67 (8.33) n = 49	0.000
Basal area of living understory trees (m^2/ha)	4.51 (2.78) n = 49	3.22 (2.58) n = 49	0.019
Basal area of standing dead canopy trees (m^2/ha)	3.26 (4.03) n = 49	2.61 (2.73) n = 49	0.361
Basal area of fallen dead trees, all species combined (m^2/ha)	22.43 (11.91) n = 49		
Diameter of living canopy trees (cm)	40 (13) n = 240	36 (11) n = 208	0.000
Diameter of living understory trees (cm)	9 (6) n = 643	9 (6) n = 1023	0.112
Diameter of standing dead canopy trees (cm)	50 (19) n = 58	39 (11) n = 75	0.000
Diameter of fallen dead trees that could be identified by species (sound or in early stages of decomposition)	49 (19) n = 141	40 (12) n = 141	0.000
Age at breast height of living canopy trees in 1994 (years)	157 (54) n = 67	122 (40) n = 48	0.000
Age at breast height of living understory trees in 1994 (yr)	84 (37) n = 78	89 (35) n = 43	0.425

B. REFERENCE CONDITIONS

High-elevation landscapes of the South Central Highlands Section during the period of indigenous settlement were composed of intermingling spruce-fir, aspen, cold-wet and cool-moist mixed conifer forests, bristlecone pine woodlands, meadows, grasslands, willow carrs, and sparsely vegetated rock fields. This assemblage of distinctive vegetation types formed a coarse-grained mosaic (scale of 10 – 10,000s of ha) produced in part by abiotic gradients and discontinuities in elevation, topography, and substrate (Peet 1981, 1988). Within the forest patches was a somewhat finer-grained but still relatively coarse mosaic (scale of 1 - 1000s of ha) of successional stages following stand-replacing fires that had occurred at various times in the past (Patten and Stromberg 1995, Wadleigh and Jenkins 1996). Finally, within the patches of late successional conifer forest, was a fine-grained mosaic (scale of 0.1 - 10s of ha) of living and dead trees produced by periodic insect outbreaks.

The two most important and ubiquitous kinds of disturbance in high-elevation landscapes during the period of indigenous settlement were stand-destroying fire and bark beetle outbreaks (Baker and Veblen 1990, Veblen et al. 1994, Veblen 2000). There also were numerous other kinds of disturbances, including: snow avalanches, windthrow, spruce budworm outbreaks, and a variety of other tree-killing insects and fungi. Stand-destroying fires generally initiated stand development and maintained a relatively coarse-grained mosaic of successional stages across the subalpine landscape. During the long intervals between fires, however, and especially

as stands reached the later stages of development, the disturbance regime of individual stands was dominated by chronic, fine-scale processes involving insects, fungi, and wind, that killed individual trees or small groups of trees (Veblen et al. 1989, 1991a; Lertzman and Krebs 1991, Roovers and Rebertus 1993, and see below).

1. High-Elevation Fire Regimes: General

Fire Frequency and Fire Size:

Fires in high-elevation forests of the Rocky Mountains were infrequent and usually very small, because late-lying snowpacks and frequent summer rain showers kept fuels too wet to burn throughout most of the growing season. However, in the rare dry years when weather and fuel conditions were suitable for extensive burning, large portions of the subalpine landscape were burned in severe, stand-destroying fires. These infrequent but extensive lethal fires left a lasting imprint on the landscape, in the form of patches of forest undergoing succession more-or-less synchronously (Romme 1982, Johnson 1992, Veblen et al. 1994, Veblen 2000).

We determined that the fire turnover time (the average time required to burn an area equal in size to the entire study area, or the average interval between successive fires at a single site) in a spruce-fir dominated landscape of the western San Juan Mountains (Figure I-1) was ca. 300 years during the reference period from the mid - 1700s through the mid - 1900s (see below). Many individual stands escaped fire for many hundreds of years. Despite our

inability to pinpoint the exact fire turnover time (because fire intervals are commonly longer than the life spans of individual trees), it is clear that during the period of indigenous settlement, extensive fires were far less frequent in the spruce-fir zone than in the forest zones at lower elevations. Fire history studies in high-elevation forests elsewhere in the Rocky Mountains have found similar results. Return intervals of major, stand-destroying fires varied with elevation, topography, and geographic area -- but usually were in the hundreds of years (Romme and Knight 1981, Romme 1982, Baker and Veblen 1990, Veblen et al. 1994, Wadleigh and Jenkins 1996, Romme et al. 1996, Baker and Kipfmueller 2001, Buechling and Baker 2004). In general, fire intervals were longest at the highest elevations and in moist depressions and valley bottoms, and somewhat shorter near the lower elevational border of the spruce-fir zone and on dry slopes and ridges.

Post-fire Succession in Spruce-Fir Forests:

Fire was followed by either rapid or gradual re-establishment of the dominant tree species via seedling establishment of the conifers, or root-sprouting of aspen (Stahelin 1943, DeByle and Winokur 1985, Johnson and Fryer 1989, Veblen et al. 1991a, Little et al. 1994, Patten and Stromberg 1995, Turner et al. 1997). Especially in the lower-elevation portion of the spruce-fir zone, fires often were followed by rapid development of aspen forests with spruce and fir in the understory (see Chapter IV). After several hundred years without further disturbance, the spruce and fir largely displaced the aspen and created a nearly pure spruce-

fir forest once again. At the highest elevations, and in other places where aspen was not present, spruce and fir directly re-colonized burned sites; this was a very slow process and hundreds of years were required for a forest to again develop under these circumstances. In limited areas, especially in the northeastern portion of the South Central Highlands Section, burned forests were initially colonized by lodgepole pine (*Pinus contorta* var. *latifolia*). This is a rare forest type in the South Central Highlands Section (see chapter VI), but it is very common in northern Colorado and the northern Rocky Mountains. The overall result of variability in the physical environment, disturbance history, and trajectories of post-disturbance succession, was a complex landscape mosaic of stands that each had a more-or-less distinctive structure and composition.

Aplet et al. (1988) described the general successional sequence following fire in subalpine forests of the central Rocky Mountains where neither lodgepole pine nor aspen is an important pioneer species. Both Engelmann spruce and subalpine fir seedlings become established at roughly the same time after the fire (colonization phase). Once a closed canopy is formed, spruce recruitment drops to a very low level, but the more shade-tolerant fir continues to establish (spruce exclusion phase). After about 300 years, the aging individuals of the pioneer tree cohorts begin to die and canopy gaps are created. This results in a second episode of spruce recruitment, and produces a bimodal size and age distribution of spruce (spruce re-initiation phase). If a stand escapes fire for many centuries, all of the pioneer fir and spruce will die and the forest will

be comprised entirely of individuals that became established beneath the canopy (second generation spruce-fir forest phase). The bimodal structure of spruce may be lost, and the forest structure may reach a more-or-less "steady-state" condition (Peet 1981).

This final "steady-state" stage of stand development has not been described in detail for Rocky Mountain spruce-fir forests, because most stands burn again before they reach this late stage of succession. In fact, nearly every stand of aspen, lodgepole pine, or spruce-fir forest in the central Rocky Mountain region, including even old-growth stands, still contains living trees that became established in the aftermath of the last stand-destroying fire (Aplet et al. 1988, Roovers and Rebertus 1993, Brown et al. 1995; but see the section on ancient forests below). However, several studies have characterized the structure and dynamics of stands in the spruce re-initiation phase, which is often regarded as "old growth." Engelmann spruce generally dominates or co-dominates the canopy but subalpine fir dominates the understory (Peet 1981, Alexander 1987, Veblen 1986a,b, Roovers and Rebertus 1993, Knight 1994). The abundance of fir is due to the greater ability of fir seedlings to establish on deep organic litter and to tolerate the shady conditions of the understory (Knapp and Smith 1982, Peet 1981, Veblen et al. 1989). Spruce maintains canopy dominance in the canopy because of its greater longevity and lower mortality rates (Oosting and Reed 1952, Veblen 1986a,b, Veblen et al. 1991a).

2. Fire History in Spruce-Fir Forests of the Western San Juan Mountains:

Although several studies have documented fire history in southwestern ponderosa pine forests, there have been few studies of fire history in southwestern spruce-fir forests (Grissino-Mayer et al. 1995, Veblen et al. 1994, Patten and Stromberg 1995, Wadleigh and Jenkins 1996). One reason for the paucity of spruce-fir studies is that the characteristic fire regime of infrequent but lethal fires leaves few fire-scarred trees from which prehistoric fire years can be determined. Therefore, it is necessary to use indirect methods, based on dating and measuring the spatial extent of stands that have developed in the wake of lethal fires, to estimate quantitative parameters of the fire regime (Johnson and Gutsell 1994). Because no previous research of this kind had been conducted within the South Central Highlands Section or nearby regions, we conducted a fire history study in a representative portion of the San Juan National Forest to provide basic information on the disturbance history of high-elevation landscapes in this region. Because fire scars are rare and postfire succession is slow and variable in spruce-fir forests, the results of this study were not nearly as precise as those from other forest types. Nevertheless, our estimates of historic fire years and turnover time for the spruce-fir forest type as a whole were sufficient for a broad initial characterization of the historical fire regime in spruce-forests of the South Central Highlands Section.

Fire History Methods:

We selected two Land Type Associations in the central and western portions of the San Juan Mountains that are dominated by high-elevation forest

types, comprising an area of several thousand hectares. The study area extended roughly from Molas Pass westward to the Dolores River. Using a GIS, we placed 100 random points within forested areas throughout these two land type associations, at a density of approximately one point per square kilometer. We then located as many of the points as possible on the ground and estimated the time that had elapsed since the last lethal fire (below). We were able to sample a total of 65 of these stands in 1996; the remaining points on the map were either physically inaccessible or proved not to be forested (due to edaphic factors or past clear-cutting).

The sampling procedure in each stand was as follows. We first visually assessed the stand for evidence of a fire within the last ca. 250 years. This evidence included charred wood on the forest floor and/or what appeared to be a roughly even-aged cohort of spruce trees with abundant lower branches (indicating that self-pruning due to light competition had not occurred or had begun to occur only recently). As discussed below, we had previously characterized stand structure in five stands having known fire dates within the past 150 years, and that information was the source of key characteristics to look for to estimate whether a stand of unknown fire history had likely burned within the past 250 years. If these evidences were not found, i.e., if there was little or no charred wood, and the trees appeared all-aged, including some obviously old trees that had shed most of their lower branches, then the stand was classified as > 250 years old and no additional sampling was done other than recording site information (elevation, aspect, slope, geologic substrate, and dominant plant species). However, in stands that had evidence of fire within the last ca. 250 years, we collected 15 - 20 increment cores from the spruce that appeared to form an even-aged cohort, in addition to the site information. The cores were returned to the lab, sanded, and their ages estimated by ring counting (cores were not cross-dated).

Once we had increment cores from stands that appeared to have developed after a lethal fire within the last ca. 250 years, it was necessary to determine how soon after the last fire the extant trees had become established, because there may be a considerable time lag in high-elevation forests (Stahelin 1943, Bollinger 1973, Little et al. 1994). We located five spruce-fir stands in which the year of the last fire could be determined precisely either from written records or from fire scars adjacent to the stand. These stands included the Lime Creek burn of 1879 (historical documentation) plus four stands near the headwaters of Rio de los Piños (fire scars dating from 1851 and 1864). In each of these stands we took an increment core from every tree within a circular plot of sufficient size to provide at least 20 samples from each major tree species in the stand. The cores were sanded and aged, and the lag time between the fire and tree establishment was ascertained as described below.

Estimating Stand Ages and Prehistoric Fire Dates:

Age structures for the five sampled stands having known fire history revealed that abundant establishment of spruce usually did not begin until several years after the fires, and that the delay was greater in stands

at higher elevations or where no adult trees survived the fire. The intervals between the year of the fire and the years of spruce establishment are summarized in Table III-3. A few spruce trees became established within the first 10 years in three of the four stands below 3,300 m (11,000 ft), but abundant spruce establishment did not occur in these stands until 10 - 30 years after the fire. In the Granite Lake stand, at 3,150 m (10,400 ft), initial establishment began somewhat later (10-20 years after the fire), and abundant establishment did not occur until 40 - 50 years after the fire. The longer delay at Granite Lake was probably due to inadequate local seed source resulting from complete mortality of trees in the pre-fire stand; in the other two stands at about this same elevation at least some of the adult trees had survived the fire and provided a seed source. We sampled only one stand above 3,300 m (11,000 ft), and in it there was a 30-year lag before the first spruce establishment occurred, and an 80-year delay before abundant spruce began to grow.

Table III-3. Intervals between year of fire (based on fire scars) and years of spruce establishment in spruce-fir stands that developed after documented fires in the San Juan National Forest, Colorado. Fire years were determined from written records (Molas Pass stand) or from fire scars collected in close proximity to the stand (all others).

Study Area	Fire Year	Presence of relict spruce that survived the fire	Elevation (feet)	Years between fire and initial spruce establishment	Years between fire and abundant spruce establishment
Divide Lake South	1851 or 1864	absent or rare	10,100	1 - 10	10 - 20
Canon Paso	1851 or 1864	abundant	10,300	10	30
Molas Pass	1879	some	10,400	1 - 10	30
Granite Lake	1851 or 1864	absent or rare	10,400	10 - 20	40 - 50
Elk Lake	1851	few	11,350	30	80

We used these patterns in timing of post-fire spruce establishment to help interpret the tree age data from the stands that we sampled in our extensive survey of spruce-fir fire history. For those stands having an apparent even-aged spruce cohort with persistent low-hanging branches, and charred wood on the forest floor, we examined the ages of dominant trees to identify the oldest individuals and the most abundant age classes. For stands below 3,000 m (10,000 ft), we estimated that the last fire had occurred within the decade prior

to the date of establishment of the oldest tree in the stand (unless it was so much older than all the others that it appeared to be a relict that survived the fire) or within two decades prior to the establishment of the most abundant age class. For stands between 3,000 – 3,300 m (10,000 - 11,000 ft), we estimated that the fire had occurred within one or two decades prior to the establishment of the oldest trees and within two to four decades prior to the dates of the most abundant age class. For stands above 3,300 m (11,000 ft), estimation of fire date is problematic, given the extremely long time lags that may occur between fire occurrence and tree establishment (Table III-3). However, none of the stands above 3,300 m (11,000 ft) in our random sample showed evidence of recent fire (i.e., none had an apparent even-aged cohort, persistent low branches, or charred wood on the forest floor), so all were classified as "too old to date the last fire," i.e., greater than 250 years old.

Once we had established an approximate decade within which the last fire had occurred in our extensive survey stands, we estimated the actual year of the fire as follows. We assumed that large fires in the subalpine zone occurred in the past -- as they do today -- only in unusually dry summers. In most years, the weather and fuels remain too wet throughout the fire season to allow for extensive fire spread in this high-elevation environment. In summers dry enough to allow fires in spruce-fir forests, we would expect an unusual number of fires at lower elevations, also. Past fires years in ponderosa pine forests of the San Juan National Forest are well documented by our collection of fire-scars from nine study areas distributed across the region (see chapter II of this

report). Therefore, we summarized all of the years in which fires occurred at three or more of our nine ponderosa pine study sites (Table III-4), and assumed that these represented most of the years of severe burning conditions. Based on this information, the years in which extensive fires most likely would have occurred in the spruce-fir zone were determined. In summary, for each stand in our extensive spruce-fir survey we determined the likely decade of the last fire (as described above), and then selected the fire year from Table III-3 that most closely matched the decade of fire occurrence.

Fire History and High-Elevation Fire Turnover Times:

By this method, we estimated that eight major fire years occurred within our spruce-fir survey area during the last 250 years (Table III-5). The most extensive fires, as indicated by the relative number of sample stands that originated at these times, occurred in 1879, 1842, 1829, and 1806. We also detected apparently less extensive fires in 1851, 1817, 1786, and 1748. Some of these fires, especially the ones in the 1700s, probably were more extensive than our data indicate, because some of their area was burned again in more recent fires. Of the 65 stands that we examined, 27 had evidence of fire within the last 250 years, and were assigned one of the fire dates above. The remaining 38 stands (58% of the sample) had not burned within the last 250 years, and were so old that it was not possible to determine the actual time since the last fire without very intensive sampling which was not feasible in this study. For purposes of calculating fire history statistics (see below), we aggregated our

time-since-fire estimates into 50-year classes (Table III-5). This was because of the inherent uncertainty in the actual year of the last lethal fire in many of our sampled stands, because we had only tree establishment dates on which to base our estimates. However, even if some of our estimates are off by as much as a decade, we believe that they are all placed within the correct half-century, and so we assumed this resolution in our statistical analyses.

Johnson and Gutsell (1994) recommend applying time-since-fire data of the kind presented in Table III-5 to a statistical model such as the Weibull distribution, from which useful descriptive statistics can be obtained. However, our data did not fit the Weibull model, or any other standard distribution contained within the program "Best-Fit." The main reason for poor fit was the large number of old stands (> 250 years since fire) in our data set (Figure III-1). Therefore, rather than attempting to interpret fire history from a poorly fit statistical model, we derived some very simple fire history statistics from inspection of the distribution of stand ages.

It can be seen in Table III-5 that the median stand age in our sample was > 250 years. This means that half of the stands sampled were > 250 years old in 1996, and half were 250 years old or younger. The quartile of the distribution falls in the 150 - 200 year old class; meaning that a quarter of the stands are 150 - 200 years old or younger, and three-quarters are 150 - 200 years old or older. We cannot compute an actual fire interval or turnover time from these data, because the majority of sampled stands were too old to estimate the actual date of the last fire. However, it is apparent that the average interval between successive fires at a point in the landscape, or the time required for an area equal to an entire land type association to burn, would be measured in centuries.

What kind of a landscape mosaic existed in the spruce-fir zone during the period of indigenous settlement? We subtracted 100 years from the ages of the stands listed in Table III-5 to estimate the distribution of stand ages in 1896, near the end of the reference period. It can be seen in Table III-6 that about one-tenth of the landscape may have been covered by young stands (< 50 years old), and a little over a third by stands < 100 years old. Most (58 %) of the forest stands in the subalpine landscape were > 150 years old, i.e., mature or old-growth. Given that ca. half the stands were <150 years old, we assumed that approximately twice this amount of time would be required for the cumulative burned area to equal the size of the entire study area, resulting in an estimated turnover time of ca. 300 years.

Table III-4. Years of extensive prehistoric fires in the San Juan National Forest, determined from analysis of 180 fire scars collected in nine ponderosa pine forest stands distributed across the Forest. The table lists years, with time elapsed since the fire (as of 1996) in parentheses.

Years in which four or more of the nine ponderosa pine study areas recorded a fire	**Years in which three of the nine ponderosa pine study areas recorded a fire**
	1880 (116 yr)
1879 (117 yr)	
1851 (145 yr)	
1847 (149 yr)	
1842 (154 yr)	
	1829 (167 yr)
	1817 (179 yr)
	1806 (190 yr)
	1796 (200 yr)
	1789 (207 yr)
1786 (210 yr)	
1778 (218 yr)	
	1773 (223 yr)
1748 (248 yr)	
1735 (261 yr)	
1729 (267 yr)	
	1724 (272 yr)
1685 (311 yr)	
	1667 (329)

Table III-5. Estimated dates of the last lethal fire in 65 spruce-fir forest stands in the San Juan National Forest.

Time Period	Total number of stands dating from fire during each 50-yr period	Individual Fire Years (number of stands dating from each)
1947 - 1996	0	--
1897 - 1946	0	--
1847 - 1896	7 stands	1879 (5 stands)
		1851 (2 stands)
1797 - 1846 (quartile)	17 stands	1842 (4 stands)
		1829 (7 stands)
		1817 (1 stand)
		1806 (5 stands)
1747 - 1796	3 stands	1786 (1 stand)
		1748 (2 stands)
pre – 1747 (median)	38 stands	Unknown

Table III-6. Distribution of sampled stand ages in spruce-fir forests of the San Juan National Forest at the end of the reference period (ca 1897), estimated by subtracting 100 years from 1996 stand ages.

Stand Age (years)	Number (%) of Stands	Cumulative % of Stands Younger than a Specified Age	
0 - 50	7 (11 %)	11 %	< 50 yr old
50 - 100	17 (26 %)	37 %	< 100 yr old
100 - 150	3 (5 %)	42 %	< 150 yr old
> 150	38 (58 %)		
Total	65 (100 %)		
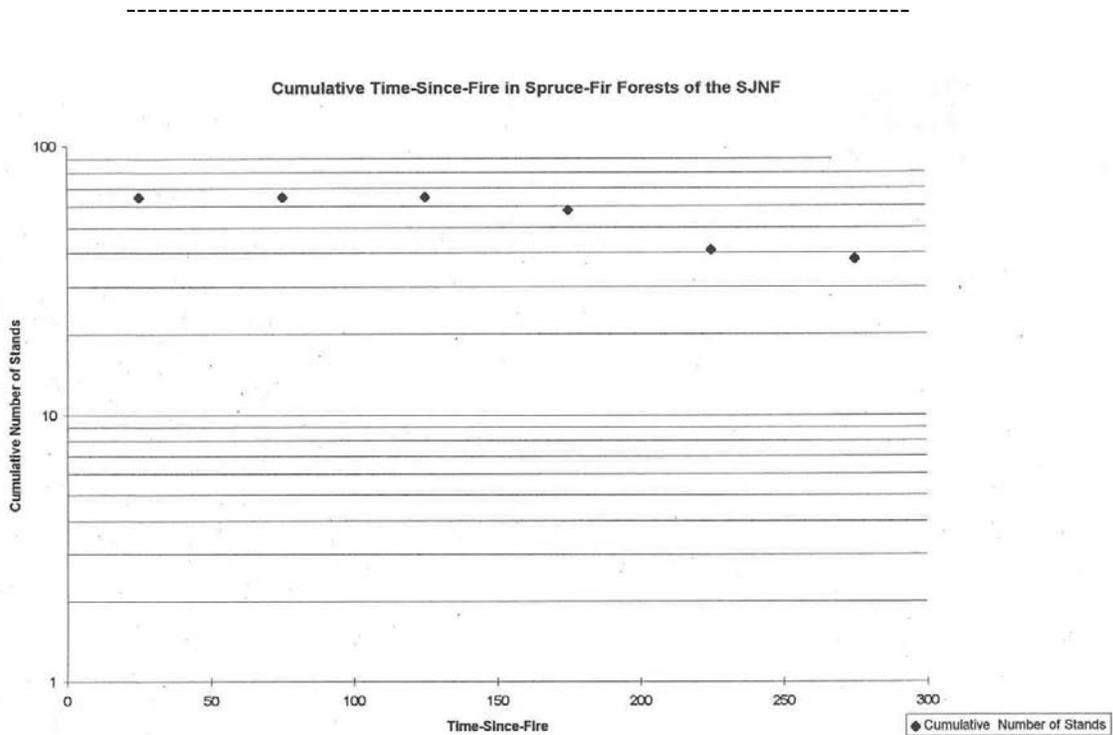
Figure III-1. Cumulative time-since-fire distribution for 65 spruce-fir forest stands sampled in the San Juan National Forest.

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3. Ancient spruce-Fir Forests:

An area of special ecological significance in the South Central Highlands Section is an ancient spruce-fir forest comprising 180 ha at the headwaters of Martinez Creek in the San Juan National Forest (Tables III-1, 2; and Romme et al. 1996). This stand apparently has not burned for 600 years or longer, nor has it been affected by major windthrow events or severe insect outbreaks. In fact, this may be the oldest spruce-fir forest described to date in the

Rocky Mountain region, in terms of the time that has elapsed since the last lethal fire or other major disturbance. It contains no evidence of a post-fire tree cohort or of directional successional changes (Figure III-2). Instead, it appears to have more-or-less stable structure and composition, maintained by a chronic, fine-scale disturbance regime of individual tree deaths and small gap formation by fungi, insects, and wind (Peet 1981, Veblen 1986a,b, Aplet et al. 1988, Lertzman and Krebs 1991, Roovers and Rebertus 1993, Veblen 2000).

We do not know if this particular stand is unique, or if there are other comparably ancient stands on mesic sites in the South Central Highlands Section, because spruce-fir forests in this region have been little studied (Romme et al. 1992). Most research on Rocky Mountain subalpine forests has been conducted in areas farther north. Even though floristic composition is similar in spruce-fir forests throughout the region (Peet 1988), there are important ecological differences between the northern and southern forests. For example, northern Colorado and Wyoming are characterized by a wet spring and early summer, followed by a dry late summer and fall, whereas the southern Rockies have a dry early summer but a wet late summer and fall (Mitchell 1976). The southern Rocky Mountains also may receive a greater proportion of precipitation in the form of rain rather than snow (Alexander 1974).

The same spruce species, *Picea engelmannii*, is a dominant canopy tree in both northern and southern portions of the central Rockies, but the subalpine fir in the south (*Abies lasiocarpa* var *arizonica*, corkbark fir) is morphologically distinct -- and

presumably also physiologically and ecologically distinct -- from the northern fir (*Abies lasiocarpa* var *lasiocarpa*). Engelmann spruce also appears to be physiologically and ecologically different in northern vs. southern populations. The most striking difference is in life span; spruce in Wyoming and northern Colorado may live from 500 - 850 years, but the oldest spruce yet documented in Arizona or southwestern Colorado (including the Martinez Creek study area) was less than 400 years old (Swetnam and Brown 1992, Brown et al. 1995). The reason(s) for shorter life spans in the southwest are unknown, but may be related to longer growing seasons with more reliable late summer moisture, resulting in more rapid tree growth and earlier onset of senescence (Kaufmann 1992), and/or enhanced activity of fungal and insect pathogens.

The relative youthfulness of the individual trees that dominate the ancient forest near the head of Martinez Creek was surprising. Despite the apparent absence of any major disturbance for 600+ years, the average age at breast height of the canopy trees was only 157 years for spruce and 122 years for fir. The oldest tree found in the area was a 400-year old spruce. Many of the trees, including some of the oldest, had narrow rings near the center, indicating that they had become established beneath a mature canopy and grew slowly for many decades before a local canopy gap developed and they were able to grow into the canopy. This finding of relatively young trees in a very old stand has important implications for efforts to evaluate old-growth status of spruce-fir forests in the southern Rockies. Some of the most ancient *stands* (in terms of time since the

last major disturbance) may be dominated by comparatively young trees. The striking differences in tree ages within old-growth spruce-fir forests also illustrates the potential hazards of extrapolating from relatively well-studied regions (e.g., northern Colorado and southeastern Wyoming) to less well-studied regions (e.g., the South Central Highlands Section).
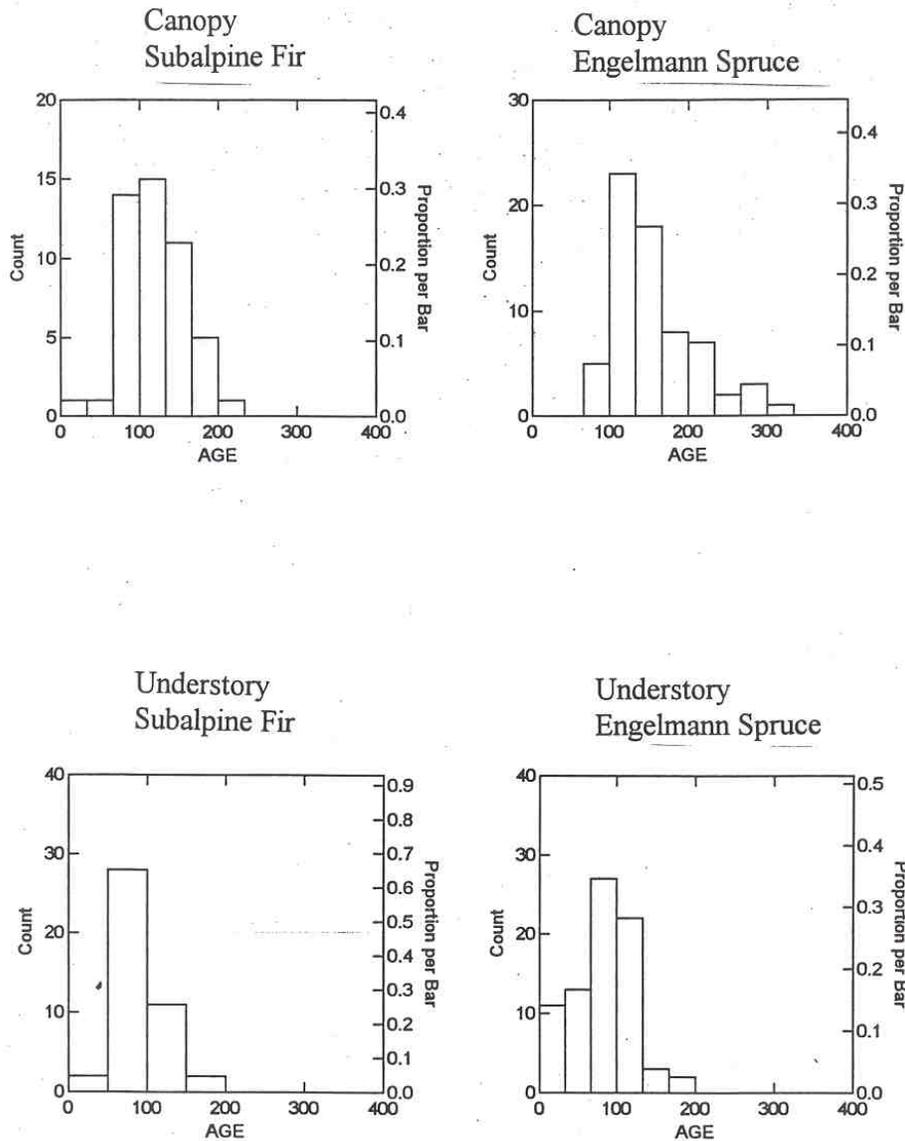
Figure III-2. Age structure of canopy trees in an ancient spruce-fir forest near the head of Martinez Creek, San Juan National Forest.

4. Insects and Diseases in High-Elevation Forests:

Disturbance by bark beetles:

The spruce beetle (*Dendroctonus rufipennis*) is a native insect whose larvae feed on the phloem of large, living or dead, Engelmann spruce trees (Furniss and Carolin 1977, Schmid and Frye 1977, Schmid and Mata 1996). Most of the time, the beetles persist in low-density, endemic populations that have little impact on forest structure. Periodically, however, populations explode into an outbreak, and the beetles may kill millions of mature pine or spruce trees over areas of thousands of hectares. The mechanisms that cause a shift from endemic to outbreak conditions are poorly understood; however, it is known that spruce beetle outbreaks may begin when windthrow creates large quantities of dead, large-diameter spruce, in which the beetles can breed rapidly (Schmid and Mata 1996). Severe outbreaks may result in the death of nearly all large-diameter spruce trees in a stand, but small-diameter host trees and other tree species usually are not attacked by the beetles (Miller 1970, Veblen et al. 1991b, Veblen 2000).

Beetle outbreaks are followed not by establishment of new tree seedlings, but by accelerated growth of previously suppressed sub-canopy and understory individuals (Veblen et al. 1991b, 1994; Eisenhart and Veblen 2000). Return intervals of extensive mountain pine beetle and spruce beetle outbreaks are shorter than the intervals between successive stand-destroying fires, but are still in the range of decades to centuries (Veblen et al. 1994, Schmid and Mata 1996). A widespread spruce beetle outbreak during the late 1800s has been documented in the mountains of Colorado and northern New Mexico (Baker and Veblen 1990). In northwestern Colorado, widespread canopy disturbances (probably caused by spruce beetles) occurred in spruce-fir forests in the periods 1716-1750, 1827-1845, 1860-1870, and 1940-1960 (Eisenhart and Veblen 2000). The most recent major spruce beetle outbreak in Colorado began in 1998 in the mountains near Steamboat Springs, following a large blowdown in 1997, and is on-going (Baker et al. 2002, Kulakowski and Veblen 2002).

Other Agents of Disturbance:

Snow avalanches were localized but significant disturbances that removed forest cover from certain topographic locations (Johnson 1987, Veblen et al. 1994). Severe, localized windstorms also may blow down all or nearly all of the trees within an area ranging from tens to thousands of hectares (Baker et al. 2002, Veblen et al. 1989, 1991a, 2001). Windthrown trees often were predisposed to falling because of root rot (caused by fungal infection, e.g., *Armillaria* spp.), but with the extreme winds that occasionally occur in the high country even healthy trees can be felled. For instance, a severe storm in October, 1997, blew down a large percentage of trees in an area of more than 10,000 ha of subalpine forest on the Routt National Forest in northern Colorado. The greatest mortality of canopy trees generally was seen in older stands, in spruce-fir stands, in dense stands, and in stands located closer to ridgetops (Baker et al. 2002, Kulakowski and Veblen 2002). Stand-level characteristics such as density and species composition had the greatest influence on mortality

patterns in areas of moderate-severity wind-throw; stand-level characteristics had relatively little influence where the winds were most severe (Veblen et al. 2001). Although a recently blown down forest looks devastated, many of the smaller trees survive, and the response involves accelerated growth of the survivors as well as new seedling establishment (Veblen et al. 1989, Kulakowski and Veblen 2003). The frequency of major windthrow events appears to vary greatly throughout the Rocky Mountain region. Major blowdowns appear to be most frequent in the Front Range of Colorado (Veblen et al. 1989, 1991a), but less frequent in the South Central Highlands Section.

The western spruce budworm (*Choristoneura occidentalis*) is a native insect that feeds on new foliage of subalpine and other firs, and can kill or reduce growth rates in host trees. Despite its name, the spruce budworm usually does not feed on spruce trees, but prefers instead white fir, subalpine fir, and Douglas-fir (Furniss and Carolin 1977, Hadley and Veblen 1993, Swetnam and Lynch 1994). Low-level, endemic populations of spruce budworm are nearly always present in spruce-fir and mixed conifer stands, where they have little impact on stand structure or dynamics. However, periodic outbreaks of either local or regional extent may suppress or kill great numbers of trees over large areas (Swetnam and Lynch 1994, Schmid and Mata 1996). Spruce budworm outbreaks occurred at intervals of 20 - 33 years in southern Colorado and northern New Mexico during the 18th and 19th centuries (Swetnam and Lynch 1994). The spruce budworm kills predominantly smaller fir trees and saplings, whereas the spruce beetle tends to kill primarily the larger spruce trees in a stand.

Both spruce and fir may succumb to a variety of fungal pathogens, including *Armillaria* spp. which sometimes causes small, localized patches of heavy mortality (Holah et al. 1993). A variety of other tree-killing insects and fungi also influenced stand structure and development, but their effects generally were overshadowed by the disturbance agents described above (Schmid and Mata 1996).

5. Interactions Among Disturbances:

The ecological effects of fires, insect outbreaks, and other kinds of disturbance were spatially heterogeneous, and there were important interactions among these disturbances ((Baker and Veblen 1990, Veblen 2000). Wind-throw created conditions in which an extensive spruce beetle outbreak could develop, but young forests (< ca 100 yr old), developing after stand-destroying fire or avalanche, were rarely attacked by the beetles (Veblen et al. 1994, Bebi et al. 2003, Kulakowski et al. 2003). Young stands also appear to be generally less susceptible to wind-throw than older stands (Baker et al. 2002, Kulakowski and Veblen 2002).

It is widely believed or suggested that blow-downs and bark beetle outbreaks set the stage for catastrophic wildfires because of increased fuel loads (e.g., Geizler et al. 1980, Schmid and Amman 1992, McCullough et al. 1998). Although this interpretation sounds intuitive, there is very little scientific evidence to support it, and about an equal amount of scientific evidence to contradict it. For example, following the 1940s spruce beetle outbreak in the White River National Forest of northern

Colorado, which resulted in dead trees over an expanse of hundreds of square kilometers, there was no increase in the number of fires compared to unaffected subalpine forests (Bebi et al. 2003). Large severe fires did occur in this region in 2002 (a severe drought year), but the forests that were affected by the 1940s outbreak and by the on-going post-1998 outbreak did not burn more extensively or more severely than forests that had not been affected by outbreaks (Bigler et al. 2005). The major reason for the lack of strong effect of insect outbreaks on subsequent fire behavior in spruce-fir forests is that fires in these ecosystems are controlled primarily by weather conditions (Schoennagel et al. 2004), not by fuels. Under drought conditions, especially when accompanied by high winds, fires burn severely whether or not there has been previous insect-caused mortality.

C. LEGACIES OF EURO-AMERICAN SETTLEMENT AND CURRENT CONDITIONS

In general, the effects of Euro-American activities have been less ubiquitous, or less intense, in high-elevation landscapes than in low-elevation landscapes of the American West -- although in localized areas the impacts certainly have been just as great. Furthermore, some elements of the current disturbance regime at high elevations appear to be little changed from the period of indigenous settlement. For example, insect outbreaks and major wind-throw events have continued to occur throughout the 20th century (Schmid and Mata 1996). Extensive outbreaks of mountain pine beetles occurred in the Front Range of Colorado during the 1970s, and a major spruce beetle outbreak affected the White River Plateau in northwestern Colorado in the 1940s (Miller 1970). Spruce budworm outbreaks affected many areas in northern New Mexico and southern Colorado in the 1970s (Swetnam and Lynch 1994). The most recent large blowdown occurred in October, 1997, when more than 1000 ha of subalpine forest were knocked over in a severe storm event on the Routt National Forest in northern Colorado (Baker et al. 2002, Kulakowski and Veblen 2002).

1. Effects of Suppression:

The effects of Euro-American activities on the fire regime are less straightforward in high-elevation landscapes than in low-elevation landscapes. Table III-5 shows that no fires occurred after 1879 among the 65 spruce-fir stands that we sampled, although some moderately large fires were reported in other parts of the spruce-fir zone during the early years of the 20th century by contemporary observers (Anonymous, Forest History, San Juan National Forest). Few large fires have occurred in spruce-fir forests of the South Central Highlands Section during the 20th century, but given the long fire-return intervals that naturally characterize these systems, the current fire-free intervals may not be far outside the range of variability in fire intervals that characterized the period of indigenous settlement. Certainly the time that has elapsed since the last fire in any individual stand is not exceptionally long, but larger landscape units (e.g.,

watersheds) may have gone for longer than usual without a large fire. There was an increase in fire frequency in high-elevation forests during the late 19th century period of European settlement in many parts of the Rocky Mountains, including much of Colorado and at least parts of Utah (Veblen and Lorenz 1991, Wadleigh and Jenkins 1996). The reasons for increased burning in the late 19th century included human ignitions, but climatic conditions during this period were equally or more important than increased ignitions (Veblen et al. 2001). Sherriff et al. (2001) concluded that fire suppression since the early 1900s had *not* reduced fire frequency in high-elevation forests of national parks and national forests in the Colorado Front Range.

Even though active fire suppression has been a widespread policy throughout most of the 20th century, our experience with the Yellowstone fires in 1988, and with recent fires in the Canadian Rockies (Romme and Despain 1989, Bessie and Johnson 1995, Weir et al. 1995) indicates that large fires in subalpine forests are controlled primarily by regional weather conditions, i.e., prolonged drought associated with persistent blocking high-pressure ridges, often accompanied by strong winds. Subtle differences in fuel loads, stand structure, and topography generally have only a small influence on fire frequency and severity in subalpine forests (Schoennagel et al. 2004), and fire suppression efforts are largely ineffective under the conditions of extreme fire weather under which the most extensive burning occurs. These observations suggest that fire control efforts in the high-elevation forests of the South Central Highlands Section

would not have been very effective in controlling fires had severe fire weather conditions occurred more often than they have in this century. Thus, the principal reasons for the paucity of high-elevation fires during this century may be related more to lack of ignitions or wet weather conditions than to direct fire suppression (but see Baker and Kipfmueller 2001, who did find a measurable effect of fire suppression in a subalpine forest in southeastern Wyoming).

Perhaps the greatest effect of 20th century fire suppression on high-elevation forests actually has been indirect, i.e., by excluding the formerly frequent low-elevation fires that sometimes burned into higher elevations as well. Some studies suggest that low-elevation fires were a major source of fires in the higher elevation forests (Barrett 1994, Baisan and Swetnam 1997), but more research is needed to determine the relative importance of local lightning ignitions in subalpine forests vs. fires sweeping in from lower elevations. The behavior and ecological effects of the few high-elevation fires that have occurred during the 20th century, do not appear to be different from those of fires during the period of indigenous settlement.

Our recent recognition of the overarching influence of weather and climate on the fire regimes of spruce-fir forests (Schoennagel et al. 2004) leads to one additional important consideration for high-elevation landscapes of this region. The last extensive fires in high-elevation forests of the South Central Highlands Section occurred during the Little Ice Age. A warming trend has been documented during the 20th century in many parts of the world, and projections call for continued warming and potential shifts in precipitation

patterns during the next century (Overpeck et al. 1990, Flannigan and Van Wagner 1991, Dale et al. 2002, Whitlock et al. 2003, Westerling et al. 2006). Therefore, fire frequency and extent in high-elevation forests during the next century may be quite different from the reference period used in this study (the several centuries just prior to the late 1800s, i.e., the Little Ice Age). If the warming trend is associated with increased drought stress, then we may see increased fire frequencies and a shift in the landscape mosaic to younger age classes -- even in Wilderness Areas where logging does not occur (Romme and Turner 1991). However, if precipitation also increases during the next century, then the low fire frequencies of the 20th century may continue.

2. Legacies of Logging and Road-Building

Although the impacts of 20th century fire suppression are equivocal, other human activities since the late 1800s have clearly altered disturbance regimes and vegetation structure of high-elevation landscapes in the South Central Highlands Section (Table III-7). These activities are related mainly to timber harvest, and to the extensive network of roads that have been constructed to support timber harvest, fire control, and recreation. In some ways, these are novel kinds of disturbance -- unprecedented in the history of the landscape -- and as such, they may potentially cause more serious changes in these ecological systems than the fires, insect outbreaks, and other disturbances that the biota have experienced throughout their evolutionary history. In much of the South Central Highlands Section, these activities did not occur extensively until after World War II. For example, there still was almost no logging and little road building in spruce-fir forests of the central San Juan Mountains as late as 1950 (McGarigal et al. 2001).

Effects of Logging:

The most conspicuous anthropogenic disturbance in high-elevation forests of the late 20th century is clear-cut logging. This method was used extensively during the 1950s - 1970s, but generally has been discontinued in spruce-fir because of problems with stand regeneration (Noble and Ronco 1978). Although a relatively small proportion of the total land area was affected by clear-cutting without adequate reforestation, the legacy of this activity will last for many decades or centuries. Direct effects of clear-cutting, without adequate tree regeneration, include a reduction in total forest area and the fragmentation of mature forest. Indirect effects include a reduction in the future timber base, as well as reduced habitat quality for some old-growth forest species that are sensitive to habitat alteration, e.g., brown creeper, Swainson's thrush, red-backed vole, dusky shrew, masked shrew, and pine marten (Raphael 1988, Hansen and Urban 1992, Hutto et al. 1993, Ruggiero et al. 1994). The legacy of clear-cuts without reforestation is not altogether "bad": most of these areas appear to have adequate plant cover to retard erosion (predominantly herbs and shrubs rather than trees), and the removal of the forest probably increased stream flow for agriculture and municipalities (Troendle 1987), and improved habitat for non-forest species (e.g., elk, deer).

Table III-7. Major legacies of past forest use and management in high-elevation forested landscapes of the South Central Highlands Section.

Human Activity	Direct Effects	Indirect Effects
Clear-cut logging, without adequate tree regeneration	* Reduction in total forest area * Fragmentation of mature forest	-- Reduction in future timber base -- Reduced habitat quality for old-growth plant & animal species -- Improved habitat for non-forest species
Clear-cut logging, with adequate tree regeneration	*Fragmentation of mature forest * Development of successional stands that lack snags & large, coarse woody debris * Homogeneous disturbance intensity throughout logging units * Shift to younger stand age classes on productive portions of landscape	-- Reduced habitat quality for old-growth plant & animal species -- Inadequate habitat for some early successional species (cavity nesters & perching birds) -- Long-term reduction of soil organic matter, soil microbial activity -- Reduced landscape heterogeneity, sharper stand boundaries, compared to former fire-created patterns -- Restriction of old-growth forests to less productive sites
Partial timber cuts	* Maintenance of some mature forest cover * Reduced structural contrast between logged & unlogged areas	-- Eventual loss of large trees, snags, and large coarse woody debris, with associated effects on soil structure & biota -- Eventual reduction in habitat quality for old-growth plant & animal species -- Extensive road system & frequent re-entry required
Extensive road building for logging, grazing management, and fire control	* Fragmentation of the forest landscape * Easy human access in most areas	-- Spread of non-native weeds -- Human disturbance of wildlife -- Easy accessibility & control for prescribed fire
Fire suppression	* Some reduction in total area burned, compared to previous century (maybe less than we think)	-- Loss of habitat for early post-fire successional species -- False sense of security with regards to future uncontrollable fires

Clear-cutting, even when followed by regeneration, results in fragmentation of mature forests and a shift in the forest patch mosaic to younger age classes. Moreover, the intensity of organic matter removal in clear-cutting is unprecedented in the evolutionary history of these forests, because fires, windstorms, and insect outbreaks all leave most of the large

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woody material in place (Spies et al. 1988, Wei et al. 1997). Indeed, young stands developing after clear-cutting are structurally very different from young stands developing after stand-destroying fire (Hutto 1995). In particular, regenerating clear-cuts lack the large dead trees, both standing and fallen, that characterize recently burned forests. Clear-cuts also lack the blackened soil, the temporary reduction in herbaceous plant cover, and brief nutrient pulse associated with fire. The intensity of disturbance tends to be more spatially homogeneous in logged areas than in burned areas, and edges between disturbed and undisturbed patches are sharper.

Since the 1970s, silvicultural methods in spruce-fir forests have emphasized partial cutting techniques rather than clear-cutting. Although partial cutting has less immediate impact than clear-cutting, its long-term legacies may be just as significant, particularly if the partial cutting is simply the first entry in a long-term even-aged management strategy (Table III-7). Since most partial cutting involves selective removal of the larger trees, there is a gradual shift to smaller size classes. Large snags and fallen logs -- key structural elements for many wildlife species and soil processes -- gradually disappear. Also, if a stand is entered repeatedly for partial cutting over a long time period, then a road system must be maintained.

Effects of Roads:

Of all the novel kinds of disturbances that humans have introduced in high-elevation forest of the South Central Highlands Section during the last century, roads may be the most ubiquitous and significant long-term legacy of our activities. Please see Chapter VII (Opportunities and Challenges) for an assessment of the impact of roads on the landscapes of this region.

D. SUMMARY

Spruce-fir forests, dominated by Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*), are the coldest and wettest forest type in the South Central Highlands Section, occurring at elevations of 2,720 – 3,580 m. The ground layer vegetation in these forests is highly variable but typically includes a rich mixture of mesophytic herbs.

The two most important and ubiquitous kinds of disturbance in high-elevation landscapes during the period of indigenous settlement were stand-destroying fire and bark beetle outbreaks. Stand-destroying fires generally initiated stand development and maintained a relatively coarse-grained mosaic of successional stages across the subalpine landscape. During the long intervals between fires, however, and especially as stands reached the later stages of development, the disturbance regime of individual stands was dominated by chronic, fine-scale processes involving insects, fungi, and wind, that killed individual trees or small groups of trees

Fires in high-elevation forests of the Rocky Mountains were infrequent and usually very small, because late-lying snowpacks and frequent summer rain showers kept fuels too wet to burn throughout most of the growing season.

However, in the rare dry years when weather and fuel conditions were suitable for extensive burning, large portions of the subalpine landscape were burned in severe, stand-destroying fires. We determined that the fire turnover time (the average time required to burn an area equal in size to the entire study area, or the average interval between successive fires at a single site) in a spruce-fir dominated landscape of the western San Juan Mountains was ca. 300 years. Many individual stands escaped fire for many hundreds of years. For example, an ancient spruce-fir forest comprising 180 ha at the headwaters of Martinez Creek in the San Juan National Forest apparently has not burned for 600 years or longer, nor has it been affected by major windthrow events or severe insect outbreaks. Fires often were followed by rapid development of aspen forests with spruce and fir in the understory, especially in the lower-elevation portion of the spruce-fir zone. At the highest elevations, and in other places where aspen was not present, spruce and fir directly re-colonized burned sites; this was a very slow process and hundreds of years were required for a forest to again develop under these circumstances.

The spruce beetle (*Dendroctonus rufipennis*) is a native insect whose larvae feed on the phloem of large, living or dead, Engelmann spruce trees. Most of the time, the beetles persist in low-density, endemic populations that have little impact on forest structure. Periodically, however, populations explode into an outbreak, and the beetles may kill many or most of the large-diameter spruce trees over areas of thousands of hectares. However, small-diameter host trees and other tree species usually are not attacked by the beetles.

Beetle outbreaks are followed not by establishment of new tree seedlings, but by accelerated growth of previously suppressed sub-canopy and understory individuals. Return intervals of extensive mountain pine beetle and spruce beetle outbreaks are shorter than the intervals between successive stand-destroying fires, but are still in the range of decades to centuries

In general, the effects of Euro-American activities have been less ubiquitous, or less intense, in spruce-fir forests than in lower-elevation forest types of the South Central highlands Section--although in localized areas the impacts have been just as great. Despite policies of fire exclusion throughout much of the 20th century, the fire regime in spruce-fir forests apparently has changed relatively little during the past century, primarily because large fires in subalpine forests are controlled primarily by regional weather conditions. Few large fires have occurred in spruce-fir forests of the South Central Highlands Section during the 20th century, but given the long fire-return intervals that naturally characterize these systems, the current fire-free intervals may not be far outside the range of variability in fire intervals that characterized the period of indigenous settlement. Certainly the time that has elapsed since the last fire in any individual stand is not exceptionally long, but larger landscape units (e.g., watersheds) may have gone for longer than usual without a large fire.

The most conspicuous anthropogenic disturbances in spruce-fir forests are clear-cut logging and road-building. These activities generally came later to the spruce-fir forest zone than to forests at lower elevations. For example, there was almost no logging and little road building in spruce-fir

forests in a large portion of the central San Juan Mountains as late as 1950 Clear-cutting was used extensively during the 1950s - 1970s, but generally has been discontinued in spruce-fir forests because of problems with stand regeneration. Since the 1970s, silvicultural methods have emphasized partial cutting techniques rather than clear-cutting. Since most partial cutting involves selective removal of the larger trees, there is a gradual shift to smaller size classes. Large snags and fallen logs--key structural elements for many wildlife species and soil processes-- gradually may disappear. Of all the novel kinds of disturbances that humans have introduced in high-elevation forest of the South Central Highlands Section during the last century, roads may be the most ubiquitous and significant long-term legacy of our activities. The impacts of roads and logging in high-elevation forests of this region are assessed in Chapter VII (Opportunities and Challenges).

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CHAPTER IV: ASPEN FORESTS

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A. VEGETATION STRUCTURE AND COMPOSITION

Aspen forests form an important vegetation type between about 2,000 and 3,300 m throughout the mountainous portions of Colorado, Utah, and northern New Mexico (Robbins 1910, Cottam 1929, Langenheim 1962, Mueggler 1976, Mitton and Grant 1996, Knight and Reiners, 2000). They are common between elevations of 2,730 – 3,330 m in the San Juan Mountains of southwestern Colorado (Romme et al. 1992). Rocky Mountain aspen forests are found in areas having cool, relatively dry summers, and snowy winters, with annual precipitation of 40 to >100 cm (16 to >40 in) (Mueggler 1976). The lower elevational limit of aspen roughly corresponds with an average annual temperature of 7 ^{0} C (45 ^{0} F) or with an annual water isopleth of at least 3 cm (7.5 in); the upper elevational limit probably is determined mainly by a short growing season (Jones and DeByle 1985a). Aspen grows on a wide variety of geologic substrates and soil types throughout the west, though best growth usually is seen on deep, loamy soils having high nutrient availability ((Mueggler 1976, Jones and DeByle 1985e). The San Juan Mountains in Colorado support some of the tallest aspen trees in the southwest, with individuals exceeding 30 m (100 ft) (Jones and Schier 1985). Aspen is dioecious; female trees tend to predominate at lower elevations whereas male trees are more common at higher elevations (Grant and Mitton 1979).

Aspen forests range from pure aspen stands to stands that are co-dominated by aspen and conifer trees. At the lowest elevations, aspen stands may interdigitate with ponderosa pine forests or grasslands, while at higher elevations the aspen is found in association with mixed conifer forests, spruce-fir forests, or meadows. Aspen forests vary greatly in species composition and stand structure. However, they are generally characterized by a luxuriant understory of deciduous shrubs and herbs with little bare ground exposed (Reynolds 1969). Forbs dominate the herbaceous stratum in most stands, but grasses and forbs are equally abundant in some stands (Mueggler 1985). Some common understory herbaceous species include *Bromus ciliatus, Carex geyeri, Delphinum barbeyi, Fragaria ovalis, Geranium richardsonii, Lathyrus leucanthus, Ligusticum porteri, Osmorhiza obtusa, Senecio serra, Thalictrum fendleri, Vicia americana,* and *Viola nuttallii* (Langenheim 1962, Morgan 1969). *Pteridium aquilinum* (brackenfern) is also very common (Jeffery Redders, *personal communication*) Snowberry (*Symphoricarpos oreophilus*) is a dominant shrub in many aspen forests of the San Juan Mountains and elsewhere in the South Central Highlands Section.

Aspen forests often are associated with relatively rich, fertile soils having a deep A1 horizon and nearly neutral soil pH, high organic

matter content and base saturation, and a relatively high availability of nitrogen, calcium, and other plant nutrients (Langenheim 1962, Morgan 1969, Bartos and DeByle 1981, Jones and DeByle 1985e). Pure or nearly pure stands of aspen often are found on mollisols, Argic Pachic Cryoborolls, and Pachic Cryoborolls, all of which are soils with a very dark, organically enriched mineral soil layer (Cryer and Murray 1992). Soils under aspen also usually support a great diversity and abundance of macro-invertebrates, such as earthworms, snails, spiders, and ants (Hoff 1957). It has been suggested that aspen and its associated species actually create these rich soil conditions (Jones and DeByle 1985e) by bringing up nutrients from deep soil layers (as much as 2 m deep) and returning them to the soil in the form of nutrient-rich, readily decomposable litter (Bartos and DeByle 1981, Parker and Parker 1983, Jones and DeByle 1985e). It has been further suggested that when conifers replace aspen through natural succession, the mollic horizon may be replaced by an albic horizon, soil pH may drop to 6.0 or lower, and there may be loss of base saturation, cation exchange capacity, nutrient availability, and soil organic matter (Cryer and Murray 1992).

However, these generalizations about aspen soils have not been adequately tested, and may be partially or wholly incorrect (Dan Binkley, personal communication). For example, Giardina et al. (2001) documented greater rates of litter decomposition and of C and N release under lodgepole pine stands than under nearby aspen stands. Moreover, Jeff Redders, ecologist on the San Juan National Forest, has conducted extensive vegetation and soils surveys, and has found that the commonly assumed differences in soils beneath aspen vs. conifer stands are not nearly as consistent as the literature cited above would suggest (Jeff Redders, *personal communication*). Binkley (personal communication) also reports that both aspen and conifer stands can be found on all of the soil types listed above. It is important to be aware of this fundamental uncertainty about aspen soil relationships, because it is sometimes assumed (i) that soils differences and species-driven changes over time are at least partially responsible for successional changes from aspen to conifer dominance, and (ii) that these kinds of soils characteristics constrain the capacity for aspen and conifers to become established on a given site following disturbance. Neither of these assumptions is well supported by research.

1. Aspen Morphology, Population Structure, & Genetic Variability

Aspen is a clonal species in which a single genetic individual may cover a large area (up to several ha) and be represented by tens, hundreds, or even thousands of individual stems, or ramets (Jones and DeByle 1985c, Mitton and Grant 1996). All or many of the ramets share a single extensive root system, although root connections tend to break as stands grow older, and some stems develop independent root systems even though they are genetically identical with the other ramets in the clone (Day 1944, Barnes 1966, Peterson and Peterson 1992, Shepperd 1993, Shepperd and Smith 1993). The root system consists of both shallow, laterally spreading roots, and deep sinker roots (Barnes 1966, Peterson and Peterson 1992) The stems of a single aspen clone

usually all have the same sex, stem form, branching habit, bark color, phenology, leaf morphology, spring and fall foliage color, and predisposition to disease and insect attack (Barnes 1966). An aspen clone could live almost indefinitely, as long as its root system remains intact and it continues to produce new stems via root sprouting or suckering. We have no methods for determining the absolute ages of aspen clones in the Rocky Mountains, but some may be thousands of years old (Baker 1925, Cottam 1954, Barnes 1966). Aspen regularly flower and produce viable seed that germinates readily in a greenhouse, but seedlings are rarely found in the wild because they cannot survive even temporary drying or competition from other established plants (Barnes 1966, Grant and Mitton 1979, Peterson and Peterson 1992, Romme et al. 1997). The great majority of aspen trees that one encounters in the South Central Highlands section probably originated as root sprouts from pre-existing clones. However, genetic studies in Yellowstone and Rocky Mountain National Parks reveal unexpectedly high levels of genetic diversity, which indicate that seedling recruitment must have occurred episodically in the studied populations (Tuskan et al. 1996 and unpublished data). It follows that at least some seedling establishment has occurred in the South Central Highlands Section, probably during cool, moist years of the 20th century, and perhaps especially near upper timberline (Elliott and Baker 2004).

Although individual trees within an aspen clone are genetically identical, the genetic variation among clones is tremendous. Substantial differences among clones, even among those growing adjacent to one another in the same habitat, have been documented with respect to timing of bud break and leaf expansion in the spring, leaf drop in the fall, susceptibility to fungus-caused diseases, leaf morphology, root and shoot growth patterns, carbohydrate reserves in the roots, root depths and diameters, and ability of new sprouts to form independent root systems (Cottam 1954, Egeberg 1963, Wall 1971, Barnes 1975, Schier and Campbell 1978, Hinds 1985, Peterson and Peterson 1992). Aspen stands that we recognize in the field may be composed of one or many genetically distinct clones.

2. Stable vs. Seral Aspen Communities

Aspen may be found in pure stands, without any other tree species, or as a seral species in stands going through succession towards eventual domination by conifers. The conifers that gradually replace aspen in seral stands of the South Central Highlands Section usually are Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) at higher elevations, and white fir (*Abies concolor*) and Douglas-fir (*Pseudotsuga menziesii*) at middle elevations. Ponderosa pine (*Pinus ponderosa*) is a co-dominant with aspen in some stands at the lowest elevations, but it is unclear whether either of these species is replacing the other or whether the mixed composition is stable.

The successional status of pure aspen stands is uncertain. Some people view pure aspen stands as a stable vegetation type that resists establishment of conifers, perhaps because of soil conditions unfavorable for conifers. This interpretation is untested, and probably is doubtful, considering the uncertainties outlined above about differences between aspen and conifer

soils.   Other people suggest that that such stands were severely disturbed in the past and that successional conifer replacement will eventually occur over very long time periods (centuries or millennia).   This interpretation also is untested.   We report offer an interpretation of stable vs. successional aspen below, but stress that more research is needed before we can confidently interpret the long-term dynamics of aspen stands in this region.

Regardless of the long-term successional status of pure aspen stands, it probably makes sense to regard them as a stable vegetation type within the intermediate time scales that managers have to deal with (Fetherolf 1917, Baker 1925, Langenheim 1962, Morgan 1969, Severson and Thilenius 1976, Mueggler 1985, Crawford et al. 1998, Kurzel 2004, Kulakowski et al. 2004, Zier and Baker 2006).   Stable aspen stands have an uneven age structure and lack conifers (Mueggler 1976, 1985, Betters and Woods 1981, Smith and Smith 2005). They tend to be associated with certain combinations of elevation, topography, and substrate, but the patterns of association are weak.   Baker (1925) reported that "heavy-soiled flats" are not favorable for conifers and support primarily aspen, whereas rocky soils favor conifers.   Pfister (unpublished dissertation, cited in Mueggler 1976) reported that stable aspen stands were common at lower elevations, but that higher-elevation aspen stands tended to be seral to conifers.   Similarly, K. T. Harper (personal communication, cited in Mueggler 1985) has observed that stable aspen communities are commonly found at mid-elevations and on southerly exposures, but that seral communities predominate at higher elevations and on northerly exposures.   Soils under pure aspen stands often are richer than soils under a mix of aspen and conifers (Parker and Parker 1983, Cryer and Murray 1992), but the generality and significance of this difference are uncertain, as described above.

Stable vs. Seral Aspen in the Western San Juan National Forest:

To better understand the patterns and causes of stable vs. seral aspen forests in the South Central Highlands Section, we surveyed 100 aspen stands in the western portion of the San Juan National Forest, representing the full range of elevation, substrate, and topographic conditions in this portion of the San Juan Mountains (Romme et al. 2001).   We found that stable aspen stands were strongly associated with lower elevations, and weakly associated with shale substrates rather than sandstones or igneous rocks, whereas seral stands were found primarily at the higher elevations.   Most of the stable stands were located adjacent to an extensive zone of ponderosa pine, where pre-1870 median fire intervals were around 10 years.

We suggest the following interpretation of this pattern. Fires were ignited frequently in the ponderosa pine and warm-dry mixed conifer zones, and probably often spread into the aspen forests located upwind and uphill.   The resulting fire intervals in the low-elevation aspen forests were longer than in the ponderosa pine forests, but still shorter than the time required for conifer seedlings to reach reproductive age; hence conifer seed sources were eventually eliminated in this area. However, the aspen responded to frequent fire by re-sprouting from the roots, thus maintaining its local

dominance. Median fire intervals in aspen forests at the higher elevations, remote from the ponderosa pine and warm-dry mixed conifer zones, were substantially longer than in aspen at lower elevations. High-elevation aspen stands are in proximity to spruce-fir forests, where median fire intervals were > 100 years (chapter III). Hence, conifer seed sources persisted at the higher elevations, and most aspen stands remained seral. Baker (1925) similarly suggested that recurrent fires (ca 50-year rotation) may help maintain pure aspen forests by eliminating conifer seedlings and saplings. Based on this study and a review of the literature, we hypothesize that many stable aspen stands in the South Central Highlands Section, especially those at lower elevations, may have developed in response to very short fire intervals in the past, and now persist without conifer invasion even in the absence of fire because local conifer seed sources have been eliminated. However, the study we conducted is best described as a pilot study; more extensive and systematic investigation is needed to further test this and alternative hypotheses.

3. Aspen and Wildlife

Aspen forests provide some of the most important wildlife habitats in the South Central Highlands Section (DeByle 1985). Winternitz (1980) attributed the high density and diversity of breeding birds in western aspen forests to (1) the abundance and large size of insects living in the aspen understory, and (2) the availability of nesting holes resulting from fungal infections in the trees and the activity of primary hole-nesting birds such as woodpeckers.

Cavity nesters are a major component of the bird fauna in western aspen forests. Winternitz (1980) found that 38% of breeding bird species in Colorado aspen forests are cavity nesters; Scott et al. (1980) reported 17-60%. In a study of breeding bird populations in aspen forests of Stoner Mesa, located in the western San Juan National Forest, Scott et al. (1980) found nests of 12 cavity-nesting species in live aspen trees and snags, including: woodpeckers, swallows, house wren, black-capped chickadee, western flycatcher, and western bluebird. About half of the 104 nests were in snags and half in living trees. Preferred snags averaged 50 ft (15 m) tall and 16 in (40 cm) diameter (range 5 - 25 in (12 – 62 cm)) with no preference for broken-topped snags as is the case with conifers. Peterson and Peterson (1992) reported a similar study in aspen forests of Canada, where large aspen and poplar snags (>30 cm (12 in) diameter) were preferred nesting sites for flicker, yellow-bellied sapsucker, downy woodpecker, red-breasted and white-breasted nuthatches, black-capped chickadee, mountain bluebird, tree and violet-green swallows, house wren, kestrel, saw-whet owl, bufflehead, goldeneye, and big brown bat. Westworth and Telfer (1993) found that winter birds and woodpeckers especially preferred older aspen stands with large (>15 cm (6 in) diameter) trees and snags. Both Peterson and Peterson (1992), and Westworth and Telfer (1993) stated that many of the bird species associated with mature aspen forests cannot thrive in habitats where large aspen trees and snags are not available.

B. REFERENCE CONDITIONS

The broad-scale geographic and elevational distribution of aspen during the pre-1870 reference period probably was similar to what we see today (described above). Floristic composition probably also was generally similar. However, landscape-scale patterns of stand age, structure, and composition varied through time in response to disturbance and successional processes. The most important natural agent of disturbance in aspen forests during the reference period was fire; other natural disturbances included windthrow, fungal diseases, tent caterpillars and other insects, snow damage, hail, lightning, and sunscald (Jones and DeByle 1985d, Jones et al.1985, Veblen 2000).

Jones and DeByle (1985b;77) observed that "... almost all even-aged aspen stands in the West appear to be the result of severe fire, whether or not the aspen type is climax on the site." Yet despite this widespread recognition of the importance of past fire in aspen forests, we have little specific information on aspen fire history. Baker (1925) studied fire scars in Ephraim Canyon in central Utah, and concluded that light fires had occurred every 7 - 10 years within the general region of his study area (actual extent of the study area not specified). Meineke (1929) determined that fires had occurred in every decade of the 19th century at the Great Basin Range Experiment Station in the Wasatch Range, Utah, but that the only severe fire was in 1867. Harniss and Harper (1982) found that the conifers were older in subalpine fir - aspen stands at higher elevations than in white fir - aspen stands at lower elevations, and suggested that this may reflect longer fire intervals at the higher

elevations, but their study provided no estimates of actual fire intervals in the aspen zone.

1. Effects of Fire in Aspen Forests

Aspen stems, with their thin bark, are easily killed even by relatively low-intensity fire (Peterson and Peterson 1992). Most fire-killed aspen stems die within the first year, but some may take as long as 5 years to finally die (Brown and DeByle 1987, 1989). The aspen root system usually is unharmed by fire because it is insulated by the soil. With death of the canopy trees, the hormones that were formerly produced by living apical meristems no longer suppress development of root sprouts, and the root system responds by producing prolific suckers within the first growing season after the fire. Highest densities of sprouts are seen in the first or second year, and may be as high as five times the pre-fire density of aspen sprouts (Jones and Trujillo 1975, Bartos and Mueggler 1981, Brown and DeByle 1987, 1989). Patton and Avant (1970) measured 30,000 - 35,000 sprouts/ha after a prescribed fire in the Sangre de Cristo Mountains of northern New Mexico. Sprout density usually is highly variable within a single stand, because of fine-scale variability in pre-fire vegetation, animal effects (e.g., pocket gophers), and fire severity (Brown and DeByle 1989). More intense fires generally produce greater sprout densities because they kill more of the canopy trees and elevate soil temperatures--both of which stimulate the roots' sprouting response (Hungerford 1988, Brown and DeByle 1989, Peterson and Peterson 1992,

Bailey et al. 2002). However, extremely intense fires, especially those that burn for long periods on the soil surface, may injure the root system and therefore reduce aspen sprout density (Parker and Parker 1983).

Root sprouts may grow up to 2.5 m tall in the first year, but grow more slowly thereafter (Peterson and Peterson 1992). Sprout density rapidly decreases after the second year, with as much as an 80% reduction by year 5, as smaller, less vigorous sprouts having lower canopy positions die, and few new sprouts are produced (Bartos and Mueggler 1981, Peterson and Peterson 1992). If a regenerating aspen stand is re-burned after only 2 - 6 years, the second fire usually kills all or most of the sprouts, and the second sprouting response by the root system is much weaker than the first (Perala 1974, Quintilio et al. 1991). Indeed, frequent re-burning at very short intervals may essentially eliminate aspen from a site (Buckman and Blankenship 1965).

In addition to killing the canopy and stimulating root sprouts, fire has major effects on the understory of aspen forests. Total understory production typically increases substantially after fire, especially after fires of moderate to high intensity, and may remain elevated for a decade or longer. Generally grass and forb production increases, but shrub production is reduced after fire (Brown and DeByle 1989, Bartos et al. 1994). Species richness also may be increased by fire. Anderson and Bailey (1980) compared aspen stands in Alberta's aspen parkland that had been burned annually in the spring for 24 years with other stands that had not burned. The burned stands had twice the number of herbaceous species per quadrat but slightly fewer woody species.

Understory species that have been observed to increase in cover after fire in aspen forests include the forbs *Chenopodium fremontii, Dracocephalum parviflorum, Epilobium angustifolium, Galium boreale, Iliamna rivularis,* and the grasses *Agrostis scabra, Calamagrostis canadensis,* and *Calamagrostis rubescens*. The shrub snowberry (*Symphoricarpos rotundifolius*) has been observed to increase dramatically after prescribed fire in the Pagosa District of the San Juan National Forest (J. Redders, personal communication). Other species are injured by fire and exhibit reduced cover after burning, such as the herbs *Aquilegia coerulea, Astragalus miser, Fragaria vesca, Galium boreale, Linnea borealis, Lupinus spp, Maianthemum canadense, Thalictrum fendleri, Vaccinium spp,* and all the shrubs (Brown and DeByle 1989, Quintilio et al. 1991, Bartos et al. 1994).

2. Fire History in Aspen Forests of the San Juan National Forest

Given our lack of precise information on aspen fire history in the South Central Highlands Section during the reference period, we conducted a detailed study of fire history in a portion of the San Juan National Forest where aspen is the predominant vegetation type (Romme et al. 2001). This area is located on the western flanks of the La Plata Mountains in the western portion of the San Juan National Forest (Figure I-1), adjacent to the "pine zone" study area that was analyzed in the context of ponderosa pine forest history (Chapter II).

Methodology:

Reconstructing fire history is more difficult in aspen forests than in ponderosa pine forests, because aspen are easily killed by fire and few fire-scarred trees can be found with which to date past fires. Therefore, we used a less precise method of determining fire history that was based on the statistical distribution of current stand ages; i.e., the time since the last lethal fire (Johnson and Gutsell 1994). To develop this method, we began by sampling several aspen stands in 1995 within the Lime Creek burn, an area located near Silverton, Colorado, where an extensive fire in 1879 was documented by written records. In two aspen stands, we removed an increment core from every stem, at a height of about 1 meter, within a circular plot. We also collected increment cores from 15 - 20 of the largest and oldest appearing stems in three additional stands. The cores were glued to slotted boards, air dried, sanded, and stained. The number of annual rings was counted under 20x power magnification using a dissecting microscope. Additional years were added to the estimate of stem age to compensate for cores that had missed the center of the tree; the number of years added was based on the radius of curvature of the innermost rings. Three years were also added to the age of each stem as an estimate of the time required to grow to coring height.

The age structure of aspen stems in the 1879 Lime Creek burn was constructed from all readable cores (about 20% of the sampled increment cores were rotten, lacking centers, or otherwise unreadable). Because of small sample sizes, ages from the two study sites were aggregated for this analysis (Figure IV-1). We expected that the post-fire aspen cohort, which resprouted after the 1879 burn, would be approximately 115 years old in 1995. Nearly 60 % of the aspen trees in our sample were, in fact, 105-115 years old. Thus, the oldest living aspen stems in the stand today apparently represent the initial post-fire cohort, even though many individuals of younger age classes were present due to continued recruitment of stems into the canopy for several decades following the fire. The Lime Creek data also revealed that very few aspen trees had survived the fire in 1879. The three trees in Figure IV-1 between 120-130 years old actually may have become initiated after the burn but were misdated due to difficulty in counting annual rings in aspen (Jones et al.1985).

Figure IV-2 shows the age distribution of 53 large aspen stems sampled from among the dominant stems (not all stems as in Figure IV-1) in three additional sites in the 1879 burn. The greatest number of stems are clustered around the expected age, while a small number of sampled trees, although large in diameter, were later recruits into the population. These results demonstrated that we could estimate the time since the last major fire in an aspen stand from the oldest cohort present.
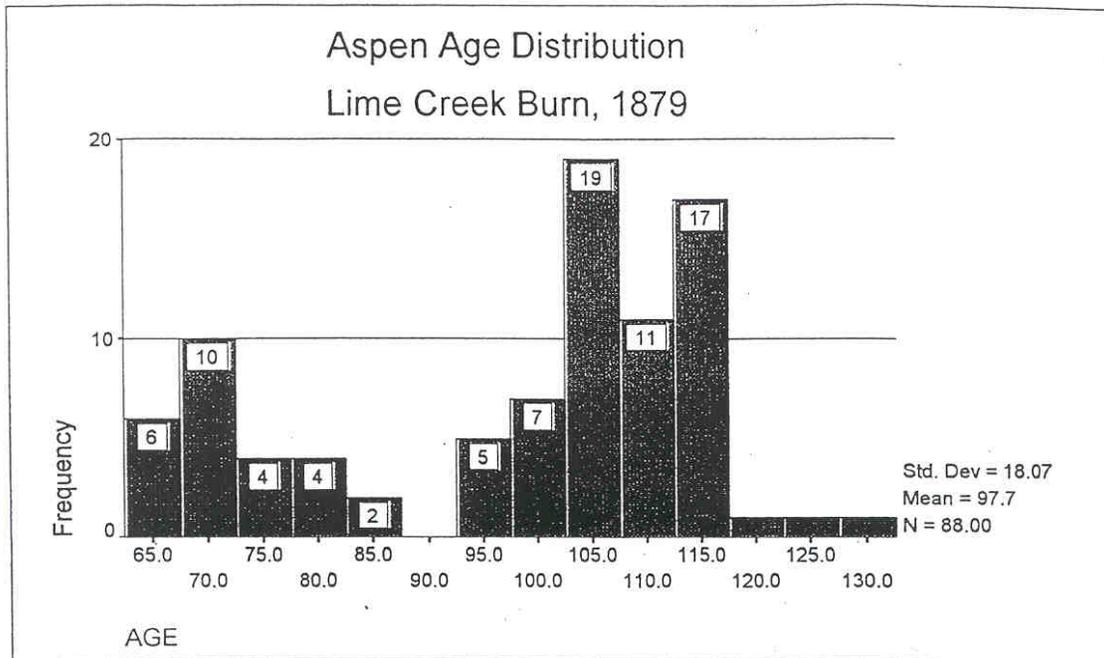
Figure IV-1. Age structure in 1995 of an aspen stand that regenerated after the 1879 Lime Creek burn. All stems > 5 cm (2 in) dbh within the plot were sampled.
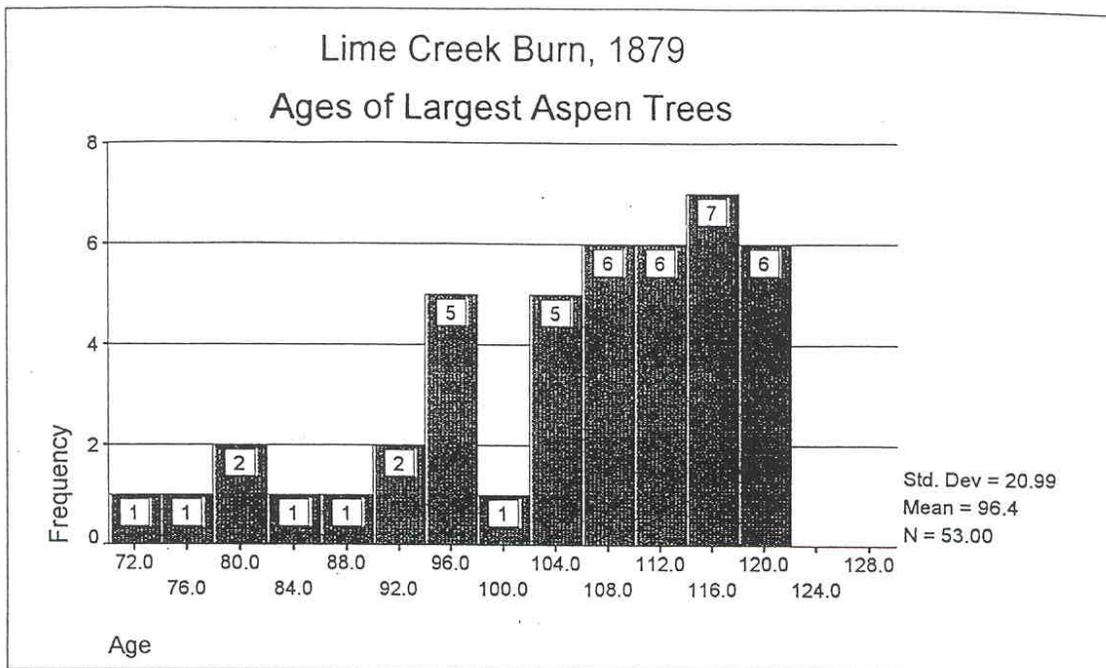
Figure IV-2. Age structure in 1995 of dominant aspen stems within three stands that regenerated after the 1879 Lime Creek burn. Smaller aspen stems were not sampled in these stands.

Landscape-Level Fire History:

Once we had verified that post-fire aspen cohorts could still be detected in aspen stands that burned >100 years ago, we applied this method to a large area of unknown fire history in the Mancos and Dolores Districts of the San Juan National Forest (Figure I-1). The study site covered 77 km^2 and spanned elevations of 2650 -3310 m. Many stands, especially at the lowest elevations, contained only aspen in the tree canopy (stable stands); while at the higher elevations, we found mostly seral stands consisting of both aspen and conifer understory, as well as some stable stands. A 1 km^2 grid was overlaid on the 7.5 minute topographic quadrangles for the study area based upon the UTM 1,000 m grid tics. A sample point was randomly chosen within each 1 km^2 grid cell as follows: a smaller grid of 100 ha grid cells was placed over each 1 km^2 cell and a random number was generated to choose the sample point from these smaller cells. In this way, each unit of the total study area had an equal probability of being sampled (Johnson and Gutsell, 1994). The sampling sites identified on the map then were located in the field. If the sampling point appeared to have recent disturbances such as logging, we randomly selected another point within the 1 km grid cell. At each sampling point we visually classified the stand as "stable," lacking conifers in the understory, or "seral," including conifers in the understory, and sampled the 20 largest, sound trees. The tree species and diameter at breast height were recorded, and an increment core was taken at breast height. Cores were glued to boards, sanded, stained, and dated as described above.

We then summarized the ages of dominant aspen stems in each stand, and estimated the decade in which the most recent lethal fire had occurred and the current stand was generated. Criteria for identifying and dating the post-fire cohorts in the stands that were sampled are summarized in Table IV-1.

Table IV-1. Method used for estimating stand age (time since last stand-replacing fire) in aspen forests on the west side of the San Juan National Forest.

1. We assumed that no fires occurred in our aspen study area after 1880. This assumption was based on the fact that the fire record ended abruptly in 1880 within the five ponderosa pine stands on the west side of the San Juan National Forest where detailed fire histories were reconstructed using fire scars. In these five ponderosa pine stands (FPC, PLT, SMI, TCK, and BCN; chapter II), there was only one fire recorded after 1880, a 1904 burn that scarred a single tree. If there were few or no fires in the ponderosa pine zone after 1880, it seems unlikely that fires would continue to occur in the more mesic aspen forests adjacent to the pine zone.
2. We assumed that in the years when fires occurred in aspen forests there also were fires in the adjacent ponderosa pine zone. We have an extensive, high-resolution fire history for the pine zone (chapter 2 in this report). Therefore, we summarized the years in each decade in which fires were detected in the ponderosa pine study

area, and assumed that these were the likely years in each decade in which fires also occurred in aspen forests. Note that we assigned aspen stand origin dates only to a decade (not to individual fire years) because the relatively imprecise aspen stem age data did not allow determination of specific fire years. Fire dates on the west side of the San Juan National Forest include the following (*asterisk indicates years of extensive fires detected in >1 ponderosa pine study area*); in brackets are the maximum ages, in 1995, of post-fire cohorts that could have become established in each decade (e.g., the oldest stems in an 1879 post-fire cohort would be 116 years old in 1995, and the oldest stems in an 1872 cohort would be 123 years old in 1995):

    -- 1904 … [**91**]
    -- none in the 1890s
    -- 1880 … [indistinguishable from 1879 cohort, so lumped with 1879]
    -- 1879* 1873 1872 … [**116 - 123**]
    -- 1868 1864 1861 1860 … [**127 - 135**]
    -- 1854 1851 … [**141 - 144**]
    -- 1847* 1842* … [**148 - 153**]
    -- 1837 1836 1831 … [**158 - 164**]
    -- 1829* 1826 1825 1824 1821 … [**166 - 174**]
    -- 1819 1818 1817 1814 1813* 1810 … [**176 - 185**]
    -- 1798 1796* 1790 … [**197 - 205**]
    -- 1789 1787 1786* 1785* 1783 1782 … [**206 - 213**]
    -- 1775 1773* 1772 … [**220 - 223**]
    -- 1768 1767 1765 1761 … [**227 - 234**]
    -- 1754* … [**241**]
    -- 1748* … [**247**]

3. We regarded the stems that established within 10 years after a fire date as a post-fire cohort. A post-fire cohort must be represented by at least 3 stems of the appropriate ages; we also recognized tentative cohorts based on only two stems of appropriate age.

4. Stands that contained no recognizable cohort were placed in one of two categories:

    a) If there were one or more stems that established before 1880, we assumed that the stand originated long ago (before the 1760s, which is the earliest date for which we have a recognizable post-fire cohort) and most of the original post-fire cohort had died through natural causes (disease, etc.).

    b) If there were no stems that established before 1880, then we assumed that the post-fire cohort had been removed by logging or other 20th century disturbances; such stands were deleted from the statistical analysis because their fire history could not be determined.

Fire History and the Landscape Mosaic during the Reference Period:

Table IV-2 summarizes the number of aspen stands that became established in each decade from the 1870s to the 1760s, as well as the number of stands that originated earlier than the 1760s, or in which stand age could not be determined. The table also distinguishes between seral and stable aspen stands. There were extensive fires in the 1870s and 1860s. Fewer stands date from the early to mid 1800s, either because there were fewer or less extensive fires during that time, or because evidence of these early fires has been destroyed by the fires of the later 1800s. We detected no stands originating from the 1770s through the first decade of the 1800s, but one stand had a clear post-fire cohort dating from the 1760s -- the oldest stand to which we could assign a specific fire decade. A substantial fraction of the sampled stands appeared to be even older -- to have last burned at some time prior to the 1760s -- but we could not assign specific fire decades to these stands, because the oldest post-fire trees apparently had disappeared through natural mortality.

What kind of a landscape mosaic existed in the aspen zone during the pre-1880 reference period? Table IV-3 summarizes the distribution of stand ages as they must have existed in the mid-1880s. These ages were determined by subtracting 110 years from the stand ages in 1995 (Table IV-2). Looking at all aspen stands combined (last column in Table IV-3), the median stand age in the 1880s was about 70 years. This means that about half of the stands in the landscape were

> 70 years old and half were < 70 years old. If half of the landscape had burned within the previous 70 years, then it would require about twice this length of time, or 140 years, for a cumulative area equal to the entire landscape to burn. Thus, our best estimate of the fire turnover time in an aspen-dominated landscape, during the period of indigenous settlement, is about 140 years.

We also compared the distribution of stand ages in the mid-1880s for seral vs. stable aspen stands (Table IV-3). The median age appears substantially older for stable stands than for seral stands, but this is in part a statistical artifact resulting from the large number of stable stands that were either >120 years old or impossible to date. Had we found only one or two additional stable stands from the early or mid-1800s, the medians would be about the same for seral and stable stands. Looking at the 25% quartiles for seral and stable stands (Table IV-3), it is apparent that the youngest quarter of the stable stands were very young (ca 10 yr old) in the mid-1880s, whereas the youngest quarter of the seral stands were 10 - 30 years old.

We conclude from this analysis that within this aspen dominated landscape during the period from the mid-1700s to the late 1800s, approximately half of the aspen forest consisted of relatively young stands developing after fires within the preceding 70 years, and that half of the stands had escaped fire for more than about 70 years. Fires occurred somewhere within the 77-km2 study area nearly every decade, but it required more than a century for a cumulative area equal to the entire study area to be

burned. Some stands burned again at relatively short intervals (< 70 years), but many others persisted for more than a century without burning. The fire turnover time in seral stands was roughly comparable to that in stable stands.

Table IV-2. Summary of decades in which the last lethal fire occurred in aspen stands of the western San Juan National Forest, Colorado.

Decade of Last Fire	Number of Seral Aspen Stands	Number of Stable Aspen Stands	Number of Aspen Stands (Seral & Stable Combined)
1870s	4	6	10
1860s	4	2	6
1850s	2	0	2
1840s	3	0	3
1830s	3	1	4
1820s	2	0	2
1810s	2	0	2
1800s	0	0	0
1790s	0	0	0
1780s	0	0	0
1770s	0	0	0
1760s	1	0	1
Pre - 1760s	16	11	27
Unknown	6	13	19
Total	43	33	76
Total, exc. unknown	37	20	57

Caveats:

Three important weakness of our fire-dating method should be acknowledged. First, we cannot say in which particular years fires occurred, because dating fires from post-fire age cohorts is inherently less precise than dendrochronological dating based on fire-scarred trees. However, because fire scars are so rare in aspen forests, the decade-level precision that we achieved is probably about the best that can be done in aspen-dominated landscapes of the Rocky Mountains. A second weakness of our method is that it cannot distinguish between two or more fires within the same decade, nor can it depict actual sizes or shapes of patches created by individual fires. Finally, it is important to note that we probably detected only the relatively large fires that occurred in the past. Many smaller fires undoubtedly occurred in places between the locations of our sample points, and were not detected. This may not be a serious error from the standpoint of interpreting past fire effects, however, because a few large fires probably were responsible for most of the burned area; this is the case today in boreal forest and several other types

of fire-dominated landscapes (Johnson 1992, Moritz 1997).

It is also important to recognize that we have depicted the structure of the aspen forest mosaic at a single time at the very end of the reference period, viz., the 1880s. We do not know just how representative this particular time was of the entire period of indigenous settlement. The landscape mosaic in 1880 probably was similar in its broad features to earlier mosaics, but there must have been substantial fluctuations over time. Thus, in earlier periods, the median stand age probably was greater or less than the 70 years that we determined for the mid-1880s. Note, for example, the period in the late 1700s and early 1800s, a time when few aspen stands were regenerated by fire in our study area (Table IV-2). This was a period of reduced fire activity throughout the Southwest (Swetnam and Baisan 1996), when the age structure of the aspen forest mosaic probably shifted towards a predominance of older stands. In contrast, the middle and later 1800s was a time of greater fire frequency in the Southwest, when the aspen landscape mosaic may have been dominated by younger stands. Therefore, in evaluating today's age structures and developing desired future conditions, the 70-year median stand age that we determined for the 1880s should be viewed only as an approximate characterization of the reference period. In fact, this estimate may be towards the younger end of the spectrum of median stand ages that existed during the period of indigenous settlement, because of the extensive fires that occurred just prior to 1880.

Table IV-3. Summary of stand ages in the mid-1880s, at the end of the period of indigenous settlement in the western San Juan National Forest.

Stand Age in 1880s	Number of Seral Aspen Stands	Number of Stable Aspen Stands	Number of Aspen Stands (Seral & Stable Combined)
10 yr	4	6 - *25% quartile*	10
20 yr	4	2	6 - *25% quartile*
30 yr	2 - *25% quartile*	0	2
40 yr	3	0	3
50 yr	3	1	4
60 yr	2	0	2
70 yr	2 - *median*	0	2 - *median*
80 yr	0	0	0
90 yr	0	0	0
100 yr	0	0	0
110 yr	0	0	0
120 yr	1	0	1
> 120 yr	16	11 - *median*	27
Total	37	20	57

3. Fire Behavior in Aspen Forests

Aspen forests in the Rocky Mountain region often do not burn readily today (Perala 1974, Alexander 1982, Jones and DeByle 1985b, Brown and Simmerman 1986, Peterson and Peterson 1992). This is because of high moisture content in the herbaceous fuels that usually dominate the ground layer, the high spreading architecture of aspen tree crowns, and the absence of flammable volatile substances in aspen wood and foliage (as compared to conifer wood). Thus, the question arises how such forests could have burned as frequently in the past as is implied by our fire history analyses. Two considerations are important.

First, extensive aspen fires in the past probably occurred only in very dry years, when fuel moisture in the herbaceous vegetation was low enough in the summer to permit ignition and fire spread. During the extremely dry year of 2002, for example, fire-fighters in the San Juan National Forest observed more active and intense fire spread in aspen stands than had been seen in the previous several decades. It is likely that most of the cumulative aspen forest area burned in half a century during the reference period was burned in just a few individual years in which very dry conditions prevailed. For example, we documented fires in aspen forests during the decade of the 1840s (above); probably all or nearly all of the area that burned in this decade was burned in the one very dry year 1842. During "ordinary" years (i.e., when climatic conditions were close to average), small fires may have occurred in aspen forests in the spring and/or early fall. These are seasons when the herbaceous vegetation is dormant and

has relatively low moisture content. However, the time period between snowmelt and greening of the herbaceous vegetation is relatively short. The time between herbaceous senescence and development of snow cover in the fall may be somewhat longer, but fall is often wet in the San Juan country and short days with low temperatures would further reduce fire spread. Consequently, fires in aspen forests during "ordinary" years probably were small.

A second consideration about fires in the aspen type prior to ca. 1880 has to do with the makeup of the herbaceous fuels. Prior to the onset of extensive and heavy grazing by livestock, the ground layer vegetation may have been dominated by different species that were more conducive to fire spread than are the species that dominate aspen forests today. Baker (1925) wrote that aspen litter was too scant to carry fire in stands grazed heavily by livestock, but that in ungrazed pockets one finds "... rank undergrowth of tall leafy grasses, such as Bromus, Agropyron, and Stipa, as well as herbs." Similarly, Tanner and Hayward (1934;213), working in Utah's in La Sal Mountains, stated that "Where grazing has been prevented for a number of years, there is a luxuriant growth of grasses and other plants among the aspens." More recently, Jones and DeByle (1985b) reported that livestock grazing can reduce herbaceous fuels such that fire cannot spread readily, and Brown and Simmerman (1986) stated that "Grazing reduced fire behavior potential by 80 to 90 percent of ungrazed conditions." Unfortunately, we simply do not know what the pre-1880 composition of aspen forests was in this area, or the extent to which

composition may have been changed by livestock grazing and other disturbances (see sections below). Therefore, we cannot determine the extent to which fire behavior in aspen forests may have been altered by human activities of the last hundred years.

In pure or nearly pure stands of aspen, where there was little or no conifer understory, fires in the past (as well as today) probably were surface fires; i.e., they burned through the fuels on the forest floor but did not burn into the tree crowns. However, even low-intensity surface fires can be stand-destroying fires in an aspen forest, because aspen is thin-barked and easily killed by fire (Jones and DeByle 1985b,

Johnson 1992). In seral aspen stands containing a large conifer component, crown fires and higher-intensity surface fires probably occurred in the past, as they do today (Peterson and Peterson 1992). Parker and Parker (1983) noted that the moderate-intensity fires that tend to occur in aspen stands also stimulate the greatest amount of aspen sprouting after fire, thus contributing to aspen's continuing dominance on the site. In contrast, the higher-intensity fires of conifer or conifer-aspen forests usually produce less aspen sprouting, and may even provide an opportunity for conifers to establish dominance on sites that could support either aspen or conifers.

C. LEGACIES OF EURO-AMERICAN SETTLEMENT AND CURRENT CONDITIONS

The aspen zone in the South Central Highlands Section has been used throughout the last century for timber production, livestock grazing, and recreation. To facilitate these activities, an extensive road network has been constructed in many portions of the aspen zone, and fires have been actively suppressed. Landscape-level impacts of roads and logging are evaluated in Chapter VII. Major impacts include fragmentation of mature forests, shifts in the age class structure of the landscape mosaic, development of successional stands that lack important structural components (e.g. large snags and fallen logs), and easy human access.

1. Stand-Level Effects of Logging in Aspen Forests

Aspen in the Rocky Mountains is usually harvested by clear-cutting

small blocks up to about 40 acres in size. Clear-cutting removes the suppressing effects of hormones produced by apical meristems, just as fire does, and usually is followed by prolific sprouting of new stems from the root system (Shepperd 1993). As with fire, maximum sprout densities are seen in year one or two. Abundance of sprouts is directly related to the number of mature stems killed; clear-cutting stimulates more sprouts than partial cutting (Schier and Smith 1979). Documented sprout densities in the first two years after clear-cutting range from 35,000 - 75,000 stems/ha (Jones 1975, Bartos and Mueggler 1982, Crouch 1981, 1983a,b). Sprouts grow rapidly, and the dominant individuals may reach average heights of 4-5 ft (1.2 – 1.5 m) by the third year (Crouch 1981). After the second year, the density of sprouts declines rapidly as smaller, less vigorous sprouts die and few new sprouts are produced. Sprouts

also die from browsing by elk, sheep and voles (*Microtus montanus*), trampling by livestock, insect damage, bacterial or fungal disease, snow damage, frost, and other causes (Crouch 1981, 1983a,b, Hinds and Shepperd 1987).

Bartos et al. (1983; cited in Bartos et al. 1991) estimated that at least 10,000 healthy aspen suckers per hectare are needed initially to ensure development of an adequately stocked new stand after clear-cutting. This density is often achieved after clear-cutting, and many stands that were cut during the past 50 years appear to be regenerating adequately. However, some clear-cuts have failed to produce sufficient sprouts to regenerate the cut stand (Crouch 1986, Johnston 2001, Rolf 2001). Failed regeneration may be most likely to occur on relatively flat sites having a high water table or heavy browsing by native or domestic ungulates (Crouch 1986; Shepperd 1993, 1996).

Clear-cutting an aspen stand often leads to increased understory production (Bartos and Mueggler 1982), although in at least one documented situation no increase was observed (Crouch 1983a,b). In contrast to the effects of fire (increased species diversity, increased forage production, decreased shrub production; see above), clear-cutting apparently does not increase native plant species diversity, though it may facilitate establishment of invasive non-native species like cheatgrass and Canada thistle (Paulson and Baker 2006). Clear-cutting also may stimulate shrub production more than herbaceous production (Bartos and Mueggler 1982).

Logging has largely replaced fire as the major disturbance process in aspen forests of the South Central Highlands Section during the past century. How similar are the effects of fire and logging? Both kinds of disturbance kill the canopy and stimulate root sprouting, but logging is very different from fire in terms of post-disturbance environmental conditions, nutrient dynamics, and responses by animals and herbaceous plants. Notably, young stands of fire origin typically contain abundant snags and fallen logs from the forest that was burned. This "biological legacy" of the previous forest provided important habitat for birds and other fauna, and played a role in nutrient cycling and soil development (see chapter VII). In the young stands developing after logging today, these biological legacies are generally lacking. Consequently, species that evolved to thrive in young aspen stands of fire origin (e.g., many cavity nesting species; Winternitz 1980, Scott et al. 1980, Hutto 1995) may be suffering for lack of quality habitat -- not only because of the paucity of recent fires, but also from the fact that young stands of logging origin are not adequate substitutes for post-fire aspen stands in terms of key habitat characteristics. For example, Westworth and Telfer (1993) conducted a census of breeding birds in aspen forests of the Canadian Rockies, including recent clear-cuts and post-fire stands up to 80 yr old. They found the lowest richness (number of species) and lowest number of territories in the clear-cut, and the highest richness and number of territories found in a 14-year old post-fire stand that contained snags and logs from the previous stand.

Comparisons of burned vs. logged aspen stands also have shown that fire stimulates increased herbaceous production and species diversity in the understory, but logging either does not

enhance the herbaceous stratum or does so minimally (Anderson and Bailey 1980, Bartos and Mueggler 1982, Crouch 1983a,b). Fire also rapidly decomposes some of the litter and duff on the forest floor, providing a pulse of nutrients, and blackens the soil surface sufficiently to increase soil temperatures. The post-fire increase in nutrients and soil temperatures is transient, lasting only a few years at most, but it may be important in stimulating herbaceous growth as well as aspen suckers.

2. Effects of Livestock Grazing and Native Ungulate Browsing

Baker (1925;17) wrote that "Mammals are the chief living enemies of aspen" and went on to say that sheep can eliminate aspen sprouts in clear-cut areas within 3 years. Similarly, Fleischner (1994) argues that livestock grazing has dramatically altered nearly every plant community throughout the west. The effects of heavy cattle and sheep grazing in the late 19th and early 20th centuries have been fairly well documented for ponderosa pine forests of the South Central Highlands Section (Chapter II), but we have little specific information on effects of livestock grazing on aspen forests of this region. Anecdotal references (e.g., Baker 1925, Tanner and Hayward 1934) suggest that the herbaceous component of aspen forests may have been altered substantially by livestock grazing (in particular, abundance of grasses may have been reduced), but reliable quantitative data on species composition during the pre-1870 reference period are not available.

In some portions of the Rocky Mountain region, native ungulates (deer and elk) have strongly affected aspen stand structure and dynamics. Deer and elk prefer aspen browse over that of any conifer (Packard 1942, Jones 1974). Continual heavy browsing on aspen sprouts and on bark of mature trees results in death of canopy trees and lack of regeneration, with eventual deterioration or loss of the stand (e.g., Gysel 1960, Krebill 1972, Mueggler 1985, Kay 1997). Heavy browsing by livestock and deer between about 1905 and 1934 is believed to have prevented establishment of new aspen stems during that period in a portion of the Fishlake National Forest in Utah, and deer apparently eliminated the aspen sprouts in a clear-cut in Arizona (Jones 1974, Mueggler and Bartos 1977). Aspen clones appear to vary greatly in their susceptibility to damage from heavy ungulate browsing, but once an aspen stand begins to deteriorate, it may lose nearly all of its canopy trees within a few decades (Schier 1975, Shields and Bockheim 1981). There is considerable controversy as to whether the effects of native ungulates on aspen forests in the west are "natural" or have been significantly altered by human activities (e.g., Romme et al. 1995, Kay 1997, Hessl 2002, Hessl and Graumlich 2002). Ungulate populations, particularly elk, are much higher now than they were in the early part of the 20th century, but they had been reduced to unnaturally low levels at the turn of century by indiscriminate hunting. We simply do not know what ungulate population densities were like during the pre-1870 reference period, nor the extent to which they naturally fluctuated in response to climatic variability and other factors.

3. Sudden Aspen Decline (SAD)

Extensive and seemingly precipitous mortality of mature aspen stems has been observed in many parts of Colorado and other western states since the early 2000s (Worrall et al. 2008). The phenomenon has been labeled "Sudden Aspen Decline" or "SAD." The cause of the mortality has not been determined. A number of insects and fungi are found on the dead and dying trees, but most are "secondary" species that feed on trees that have been killed by other agents. It is possible that the mortality is primarily a result of the severe drought that affected Colorado from the late 1990s through early 2000s. The fact that mortality appears greater on drier and lower-elevation sites supports this interpretation. Some affected stands appear to have enough healthy sprouts to regenerate the stand, but in other stands the root systems appear to be dead, and regeneration is not assured. Overall, there is great uncertainty about the causes, the spatial patterns, and the likely consequences of SAD, but research is underway to answer some of the questions.

4. Is Aspen Declining in the South Central Highlands Section?

Charles Kay published a provocative paper in 1997 entitled "Is aspen doomed?" in which he argued that western aspen stands are being destroyed by lack of fire coupled with excessive browsing by unnaturally high populations of native ungulates, mainly elk. Kay recommended more intensive management of ungulates and vegetation to alleviate aspen decline. Similarly, Johnson (1994) compared forest inventory data from New Mexico and Arizona in 1962 and 1986, and found that the total area classified as aspen had declined by 222,000 acres (46%), presumably because of increased growth of conifers (especially in smaller size classes) and poor establishment of new aspen stems. He concluded that "If such trends continue, unchanged by either prescribed fire or cutting, aspen will cease to exist as a distinct cover type in about 25 years ..." (Johnson 1994:16), and argued that we need more logging and prescribed fire to recreate more natural conditions in forests of the southwest. A recent report by Club 20 in 1998, entitled "Decline of the aspen," reiterates these same points. However, the area covered by aspen forests actually has remained stable or even increased throughout the past century in some parts of Colorado, notably around Grand Mesa (Kulakowski et al. 2004), the Gunnison Basin (Crawford et al. 1998, Manier and Laven 2002), the Uncompahgre Plateau (Jensen 2000, Smith and Smith 2005), and parts of the San Juan Mountains (Elliott and Baker 2006).

A key issue that confounds our attempts to evaluate the status of aspen in the West is the tremendous variability, at both regional and local scales, in aspen history and current trends (Romme et al. 2001, Hessl 2002). Rogers (2002) made a broad regional assessment of aspen status throughout much of Idaho, Wyoming, and Colorado, based on recently collected Forest Health Monitoring data, and determined that aspen was being replaced by other species in 61% of the surveyed plots. In the remaining 39% of plots, aspen appeared stable, with multi-layered stands and no active conifer invasion. Replacement by conifers was most

prominent in eastern parts of the study area, whereas stable stands were especially numerous in the western area, including parts of Colorado.

Another fundamental difficulty in assessing the magnitude and significance of aspen decline is the fact that investigators have used multiple, and often conflicting, concepts of "decline." Kashian et al. (2007) evaluated aspen forests in northern Colorado and determined that different definitions of aspen decline led to very different conclusions. In the sections below we evaluate the status of aspen in the South Central Highlands Section in the context of two different ecological processes that have been interpreted as "aspen decline." These processes are (i) replacement of aspen by conifers in the absence of fire or logging, and (ii) failure of aspen to regenerate because of chronic heavy browsing by native or domestic ungulates.

Replacement of Aspen by Conifers

The frequency and extent of fires in aspen forests during the 20th century have been substantially reduced in comparison with the pre-1870 reference period. As a result, the age structure of aspen stands across the landscape has shifted towards an increasing prevalence of older stands in at least some areas. Bartos et al. (1991) reported that two-thirds of the aspen stands in the Intermountain Region of the National Forests are > 95 years old.

From a silvicultural perspective, these aging stands are becoming increasingly susceptible to disease and mortality (Hinds 1985). Once aspen stems reach an age of about 60 years, the incidence of decay begins to rise in a fairly linear fashion with increasing age; in one study, 6 - 7% of 60-year old stems were classified as cull (unsuitable for timber, usually because of decay or deformity), whereas 44 - 53% of 160-year old trees were so classified (Hinds and Wengert 1977). Meineke (1929) also reported a high incidence of cull in stands greater than 90 years old, with almost 45% gross cull in the 121 - 130 age class. In the context of evaluating aspen decline, it is also important to note that conifers are becoming increasingly dominant in many successional aspen stands. For example, conifer basal area increased from 10 to 23 m^2/ha while aspen basal area decreased from 21 to 18 m^2/ha between 1979 and 2001 in aspen stands with conifer understories on the Uncompahgre Plateau (Smith and Smith 2005). Similarly, Kashian et al. (2007) reported increasing conifer dominance in a third of the stands sampled in the Colorado Front Range. If we define "aspen decline" as a decrease in aspen density, basal area, or cover over time, then aspen is declining in these portions of Colorado's Uncompahgre Plateau and Front Range.

From a different perspective, however, we can view these changes in conifer abundance as natural successional processes that have always occurred during long periods without major stand disturbance. In fact, this is how Smith and Smith (2005) interpreted the changes that they documented on the Uncompahgre Plateau. Kaye et al. (2003) evaluated 123 patches of aspen within a 340 km2 portion of Rocky Mountain National Park in northern Colorado; they found that aspen was mixed with conifers in two-thirds of the patches, but concluded that aspen was not declining when viewed over an appropriately long time span of many

decades or centuries. Kashian et al. (2007) documented a cyclic alternation of aspen dominance after fire followed by conifer dominance during long intervals between fires. In a further analysis of aspen in Rocky Mountain National Park, Kaye et al. (2005) found that a dense growth of conifers resulted in reduced establishment of new aspen stems but did not reduce growth or increase mortality rates in already-established aspen.

Our interpretation of aspen in the South Central Highlands Section is that increasing conifer abundance does not pose a serious threat to the long-term persistence or ecological function of aspen. Thus, conifers are not causing aspen to decline in this region. Even though aspen cover has declined in some areas (Johnson 1994), we view this trend as a natural successional process during a century of infrequent fire that followed the extensive fires of the second half of the 19th century. Fires have become more frequent in the past decade, and will likely become even more frequent in the coming century (Westerling et al. 2006). These fires will reduce the conifer component and once again increase aspen dominance as described by Kashian et al. (2007).

Suppression of Aspen Regeneration by Chronic Heavy Ungulate Browsing

Aspen sprouts are a favored browse of native ungulates, including deer and elk Hessl 2002, Hessl and Graumlich 2002), as well as domestic ungulates such as cattle and sheep (Baker 1925, Fleischner 1994). Chronic heavy browsing can prevent aspen root sprouts from growing into new trees. If canopy trees are dying, either gradually through natural aging processes or suddenly due to fire, insects, disease, or SAD (see above), then failure of the root sprouts to regenerate the stand can lead to loss of the stand and loss of the clone(s) represented in the stand (Kay 1997). This would represent aspen decline by almost any definition.

Declining stands of this kind are found in places throughout the West, including the South Central Highlands Section. However, we do not have a good estimate of how much of the aspen in this region is declining as a result of excessive ungulate browsing. One can find aspen stands with good regeneration following fire or clear-cutting as well as other stands where browsing has inhibited regeneration (e.g., Crouch 1981, Shepperd and Fairweather 1994, Paulson and Baker 2006). Consequently, we cannot say how extensive or ecologically significant is aspen decline due to excessive browsing in the South Central Highlands Section; this is a topic for additional research.

Although ungulate browsing appears to be a more significant process of aspen decline than replacement of aspen by conifers, and "SAD" may become of greater importance in the future (especially if we see increasing drought and warming temperatures), our overall assessment is that aspen does not appear to be in imminent danger of disappearing from landscapes of the South Central Highlands Section.

D. SUMMARY

Aspen forests form an important vegetation type between about 2,000 and 3,300 m throughout the mountainous portions of Colorado, Utah, and northern New Mexico These forests grow on a wide variety of geologic substrates and soil types, although best growth usually is seen on deep, loamy soils having high nutrient availability. Aspen is a clonal species in which a single genetic individual may cover a large area (up to several hectare) and be represented by tens, hundreds, or even thousands of individual stems. Ranging from pure aspen stands ("stable aspen") to stands that are co-dominated by aspen and conifer species (seral aspen"), aspen forests generally are characterized by a luxuriant understory of deciduous shrubs and herbs and provide some of the most important wildlife habitats in the South Central Highlands Section.

In a survey of 100 stable and seral aspen forests in the western portion of the San Juan National Forest, stable aspen stands were strongly associated with lower elevations near the ponderosa pine zone, and weakly associated with shale substrates rather than sandstones or igneous rocks, whereas seral stands were found primarily at the higher elevations. This pattern suggests that stable aspen stands in the South Central Highlands Section may have developed in response to very short fire intervals in the past, but more research is needed to fully explain the distribution of stable vs. seral aspen in this region.

The most important natural agent of disturbance in aspen forests during the reference period was fire; other natural disturbances included windthrow, fungal diseases, tent caterpillars and other insects, snow damage, hail, lightning, and sunscald. Aspen stems are easily killed even by relatively low-intensity fire. However, the aspen root system usually is unharmed by fire, and it typically produces abundant root sprouts within the first growing season after the fire. A detailed study of fire history in the western portion of the San Juan National Forest revealed that there were extensive fires in the 1870s and 1860s, but no large fires after that time. Median aspen stand age in the 1880s was about 70 years. If half of the landscape had burned within the previous 70 years, then it would require about twice this length of time, or 140 years, for a cumulative area equal to the entire landscape to burn. This is our best estimate of the fire turnover time in this aspen-dominated landscape during the period of indigenous settlement.

Aspen in the Rocky Mountains is usually harvested by clear-cutting small blocks up to about 40 acres in size. Clear-cutting usually is followed by prolific sprouting of new stems from the root system, and often leads to increased understory production as well. However, some clear-cuts have failed to produce sufficient sprouts to regenerate the cut stand, especially on relatively flat sites having a high water table or heavy browsing by native or domestic ungulates

Extensive and seemingly precipitous mortality of mature aspen stems has been observed in many parts of Colorado and other western states since about 2000. The phenomenon has been labeled "Sudden Aspen Decline" or "SAD." The cause of the mortality has not been determined.

Aspen cover has decreased in many areas since the early 1900s,

leading to concerns about aspen decline. However, the area covered by aspen forests actually has remained stable or even increased throughout the past century in other areas. Assessing the magnitude and significance of aspen decline is complicated by the fact that investigators have used multiple, and often conflicting, concepts of "decline," and different definitions of aspen decline lead to very different conclusions. We evaluated two different ecological processes that have been interpreted as "aspen decline:" (i) replacement of aspen by conifers in the absence of fire or logging, and (ii) failure of aspen to regenerate because of chronic heavy browsing by native or domestic ungulates. We view the first process—a trend of increasing conifer dominance-- as a natural successional process that has always occurred during long periods without major stand disturbance and that is not a serious threat to the long-term persistence or ecological function of aspen. Future fires will reduce the conifer component and once again increase aspen dominance. The second process—failure of stands to regenerate because of excessive browsing—is a more serious form of aspen decline. Unfortunately, we do not have a good estimate of how much of the aspen in this region is declining as a result of excessive ungulate browsing; this is an important topic for future research.

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CHAPTER V: MIXED CONIFER FORESTS

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A. VEGETATION STRUCTURE AND COMPOSITION

Mixed conifer forests are perhaps the most variable and complex of any forest type in the southwestern mountains, in terms of species composition, stand structure, and dynamics. They also have received relatively little research attention (Romme et al. 1992). Consequently, we have a relatively poor understanding of the long-term ecological dynamics and interactions that have shaped the mixed conifer landscape in the past, and that explain biotic responses to current management activities.

The mixed conifer forest type occurs at elevations ranging from about 2270 – 3030 m (7,500 – 10,000 ft). It is associated with the montane and subalpine climatic zones, the frigid and cryic soil temperature regimes, and the ustic and udic soil moisture regimes throughout the South Central Highlands Section. This diverse forest type includes the following major tree species: white fir (*Abies concolor*), Douglas-fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), Engelmann spruce (*Picea engelmannii*), subalpine/corkbark fir (*Abies lasiocarpa*/*arizonica*), and quaking aspen (*Populus tremuloides*). Blue spruce (*Picea pungens*) or southwestern white pine (*Pinus strobiformis*) also may be locally important, but are often absent or rare in mixed conifer stands. The distribution and abundance of these species within an individual stand is dependent on microclimate (as reflected by soil temperature and moisture

regimes, aspect, and topography), previous disturbance by fire or insects, and past management practices (particularly timber harvest and fire exclusion). Timber harvest has selectively removed Douglas-fir, ponderosa pine, and Engelmann spruce from many of these forests, while fire exclusion has allowed white fir to increase on many sites.

Environmental conditions, species composition, and disturbance regimes in the mixed conifer zone are intermediate between the adjacent ponderosa pine zone at lower elevations (Chapter II) and the adjacent spruce-fir zone at higher elevations (Chapter III). The high elevation spruce-fir and low elevation ponderosa pine forests have distinctive and very different structural and dynamic characteristics, and the mid-elevation mixed conifer forests represent a transition between these two distinctive types, with characteristics of both spruce-fir and ponderosa pine forest types. This is one reason why mixed conifer forests are so very complex and heterogeneous in structure, composition, and disturbance history.

To help make sense of this complexity and variability within the mixed conifer zone, either two or three general sub-types or "phases" are commonly recognized within the wide spectrum of the mixed conifer forest type in the South Central Highlands Section. The phases of mixed conifer forest are distinguished on the basis of (i) soil and climatic conditions, (ii) tree

species composition, and (iii) predominant disturbance regimes during the reference period. Unfortunately, research to date is not sufficient to permit a confident conclusion as to which of these two classification systems is most appropriate, and indeed, each is useful for different purposes. Therefore, in this chapter we briefly describe each classification, and then refer to one or the other throughout the remainder of the text, depending on which system works better for a particular question at hand. A high priority for research is to improve our understanding of the basic patterns of historical species distributions, the range of stand and landscape structures, and disturbance regimes of the mixed conifer forest zone.

The two-phase classification recognizes a **warm-dry** phase of mixed conifer forests, generally at lower elevations, and a **cool-moist** phase, generally at higher elevations (Table V-1a). Ponderosa pine is the key indicator species for the **warm-dry** phase, and many stands contain additional species from the ponderosa pine forests of lower elevations, e.g., Gambel oak (*Quercus gambelii*), *Mahonia repens*, and *Poa fendleriana.* Indeed, **warm-dry** mixed conifer forests resemble ponderosa pine forests in terms of general stand structure and disturbance regimes. However, **warm-dry** mixed conifer forests differ significantly from ponderosa pine forests in that Douglas-fir and white fir are also important components of most **warm-dry** mixed conifer forests, and historical disturbance regimes also differ, as explained below. The second sub-type of mixed conifer forest, the **cool-moist** phase, is characterized by the absence

(or near-absence) of ponderosa pine, a greater abundance of Douglas-fir and white fir, and, on some sites, the occurrence of several species more typical of the spruce-fir zone, notably subalpine/corkbark fir and Engelmann spruce. Disturbance regimes in the **cool-moist** mixed conifer zone also resemble the spruce-fir zone, as explained below.

The three-phase classification of mixed conifer forests (Table V-1b) retains the same **warm-dry** phase as in the two-phase classification, but subdivides the former cool-moist phase into two units: a **cool-moist** phase with the same name as before but a different definition*,* and a new **cold-wet** phase. The **warm-dry** and **cold-wet** phases represent the extremes in variability of stand structure, composition, and dynamics within this remarkably heterogeneous forest type, while the **cool-moist** phase is intermediate. Douglas-fir and white fir are the constant species occurring throughout the mixed conifer forest type in this three-phase classification, while ponderosa pine is the key indicator species for the **warm-dry** phase, and subalpine/corkbark fir and Engelmann spruce are the key indicator species for the **cold-wet** phase. Ponderosa pine, Engelmann spruce, and subalpine/corkbark fir are absent or only minor components in the **cool-moist** mixed conifer phase.

We can see the transition from **warm-dry** to **cool-moist** (in the two-phase classification) or **cold-wet** (in the three-phase classification) mixed conifer forests in the Sand Bench area, located in the Pagosa District of the San Juan National Forest. Sand Bench is a relatively flat ridge-top underlain by Dakota sandstone and surrounded by deep rugged canyons.

Table V-1. Distinguishing features of mixed conifer forests in the South Central Highlands Section. (a) Comparison of **warm-dry** and **cool-moist** mixed conifer forests in the two-phase classification system (see text). (b) Comparison of **warm-dry**, **cool-moist**, and **cold-wet** mixed conifer forests in the three-phase classification system (see text).

(a)

	Warm-Dry Mixed Conifer	Cool-Moist Mixed Conifer
Environments	lower elevations, mostly southerly aspects	higher elevations, mostly northerly aspects
Major Species	ponderosa pine, Douglas-fir, white fir, Gambel oak, other shrubs	white fir, subalpine fir, Douglas-fir, aspen, Engelmann spruce, snowberry, other shrubs
Disturbance Regime	recurrent, non-lethal fires (20-50 yr intervals); rare lethal fires (>100 yr. intervals)	lethal fires at long intervals (>100 yr.); occasional small non-lethal fires; landscape patch mosaic
Common Stand Structure	overstory of ponderosa pine and/or Douglas-fir; understory of white fir, and oak	even-aged or all-aged stands of variable species composition and structure
Regeneration of Canopy Trees	episodic establishment of pine and Douglas-fir, perhaps mainly after fire; adult trees survive non-lethal fire	Episodic or continual establishment of conifers between fires; aspen and possibly Douglas-fir establish primarily after fire
Regeneration of Understory Species	continual establishment of white fir during intervals between fires; both mature and juvenile fir killed by most fires	continual establishment of white fir and other shade-tolerant conifers during intervals between fires; most trees killed by most fires

(b)

	Warm-Dry Mixed Conifer	Cool-Moist Mixed Conifer	Cold-Wet Mixed Conifer
Environments	lower elevations, mostly southerly aspects	middle elevations and all aspects	higher elevations, mostly northerly aspects
Major Species	ponderosa pine, Douglas-fir, white fir, Gambel oak, other shrubs	white fir, Douglas-fir, aspen, snowberry, other shrubs	Engelmann spruce, white fir, subalpine fir, Douglas-fir, aspen, snowberry, other shrubs
Disturbance Regime	recurrent, non-lethal fires (20-50 yr intervals); rare lethal fires (>100 yr. intervals)	fires of mixed severity (lethal and non-lethal) and highly variable interval (50 – 200 yr)	lethal fires at long intervals (>200 yr.); occasional small non-lethal fires; landscape patch mosaic
Common Stand Structure	overstory of ponderosa pine, Douglas-fir and white fir; understory of white fir and oak	even-aged or all-aged stands of variable species composition and structure	even-aged or all-aged stands of variable species composition and structure
Regeneration of Canopy Trees	episodic establishment of pine and Douglas-fir, perhaps mainly after fire; adult trees survive non-lethal fire	Episodic or continual establishment of conifers between fires; aspen and possibly Douglas-fir establish primarily after fire	Episodic or continual establishment of conifers between fires; aspen and possibly Douglas-fir establish primarily after fire
Regeneration of Understory Species	continual establishment of white fir during intervals between fires; both mature and juvenile fir killed by most fires	continual establishment of white fir and other shade-tolerant conifers during intervals between fires; most trees killed by most fires	continual establishment of white fir and other shade-tolerant conifers during intervals between fires; most trees killed by most fires

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The elevation gradually increases from 2,700 m (9,000 ft) at the south end of the ridge to 2,900 m (9,600 ft) at the north end, over a distance of 2 miles. Jamieson and Romme (1991) characterized plant composition and forest structure in six unlogged stands on Sand Bench, including two stands at the lower end, two in the middle, and two at the upper end. At the lower end of Sand Bench, one finds **warm-dry** mixed conifer forest with ponderosa pine, Douglas-fir, white fir, aspen, the shrubs Gambel oak, *Arctostaphylos uva-ursi,* and *Chimaphila umbellatum,* the herbs *Delphinium nelsoni* and *Mertensia fusiformis*, and other species characteristic of lower elevations (Table V-2). In contrast, the upper end of Sand Bench supports **cool-moist** or **cold-wet** mixed conifer forest, with Douglas-fir, white fir, subalpine fir, Engelmann spruce, aspen, the shrubs *Lonicera involucrata, Sambucus microbotrys,* and *Vaccinium myrtillis,* the herbs *Arnica cordifolia, Erythronium grandiflorum, Geranium richardsonii, Pedicularis racemosa*, and other species characteristic of higher elevations (Table V-2).

Other common understory species found in **warm-dry** mixed conifer forests of the South Central Highlands Section include the shrubs *Amelanchier alnifolia, Arctostaphylos uva-ursi, Mahonia repens, Symphoricarpos rotundifolius,* and the herbs *Achillea lanulosa, Antennaria rosea, Carex geyeri, Elymus elymoides, Erigeron formosissimus, Geranium caespitosum, Koeleria macrantha, Lathyrus leucanthus, Poa fendleriana, Poa pratensis*, *Potentilla hippiana, Pseudocymopterus montanus,* and *Solidago simplex.*

Other common understory species found in **cool-moist** or **cold-wet** mixed conifer forests of the South Central Highlands Section include the shrub *Rubus parviflorus*, and the herbs *Actaea rubra, Aquilegia elegantula, Artemisia franserioides, Bromopsis canadensis, Carex geyeri, Erigeron eximius, Fragaria vesca, Goodyera oblongifolia, Lathyrus leucanthus, Ligusticum porteri, Luzula parviflora, Maianthemum stellatum, Mertensia ciliata, Oreochrysum parryi, Orthilia secunda, Osmorhiza depauperata, Pyrola minor,* and *Viola canadensis.*

As emphasized previously, much additional research is needed to clarify species distribution patterns in this remarkably complex and heterogeneous forest type. For example, an especially curious and unexplained distribution pattern is seen in white fir, which is present (often abundant) in most of the mixed conifer forests of the South Central Highlands Section, but is rare or absent in the country west of the Animas River watershed and throughout the Uncompahgre Plateau -- even though soils, topography, and climate in these latter areas do not appear to be strikingly different.

B. REFERENCE CONDITIONS

In the paragraphs below we summarize the results of empirical studies on historical fire regimes in mixed conifer forests of the South Central Highlands. We also have reconstructed historical landscape patterns and changes by means of a dynamic landscape simulation model (RMLANDS), which provides a somewhat different but complementary

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Table V-2. Density of tree species (stems/ha, all size classes > 1 m tall), determined by the point-quarter-distance method, and percent cover of selected shrub and herbaceous species, visually estimated, along an elevational gradient in the Sand Bench study area, San Juan National Forest. See Jamieson and Romme (1991) for sampling details and for complete floristic list (12 species of shrubs, 48 forbs, 10 graminoids, 1 fern, 1 lichen, and 10 bryophytes).

Species	Upper Stands (9,500 - 9,600 ft)	Middle Stands (9,200 - 9,300 ft)	Lower Stands (9,000 - 9,100 ft)
Trees:			
Abies lasiocarpa	520	76	0
Picea engelmannii	502	313	0
Populus tremuloides	82	152	45
Pseudotsuga menziesii	39	110	105
Abies concolor	18	30	92
Pinus ponderosa	0	0	53
Shrubs:			
Vaccinium myrtilus	50 - 75 %	50 - 75 %	25 - 50 %
Sambucus microbotrys	1 - 5 %	0	0
Lonicera involucrata	1 - 5 %	0	0
Symphoricarpos oreophilus	1 - 5 %	1 - 5 %	1 - 5 %
Pachystima myrsinites	1 - 5 %	1 - 5 %	1 - 5 %
Mahonia repens	0	1 - 5 %	1 - 5 %
Rosa sp.	0	1 - 5 %	1 - 5 %
Quercus gambelii	0	0	1 - 5 %
Arctostaphylos uva-ursi	0	0	1 - 5 %
Chimaphila umbellata	0	0	1 - 5 %
Herbs:			
Erythronium grandiflorum	5 - 25 %	1 - 5 %	1 - 5 %
Geranium richardsonii	5 - 25 %	1 - 5 %	0
Pedicularis racemosa	5 - 25 %	1 - 5 %	0
Arnica cordifolia	5 - 25 %	25 - 50 %	0
Carex geyeri	1 - 5 %	25 - 50 %	1 - 5 %
Bromus carinatus	1 - 5 %	5 - 25 %	1 - 5 %
Claytonia lanceolata	0	0	1 - 5 %
Delphinium nelsoni	0	0	1 - 5 %
Mertensia fusiformis	0	0	1 - 5 %
Mosses:			
Mnium arizonicum	1 - 5 %	1 - 5 %	0
Ceratodon purpureus	0	0	1 - 5 %

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picture of the spatial and temporal variability that characterized this complex vegetation type. We do not present the results of RMLANDS in this report, but refer the reader to McGarigal and Romme (2005).

1. **Warm-Dry** Mixed Conifer Forests

During the period of indigenous settlement, many of the **warm-dry** mixed conifer forests of the South Central Highlands Section appear to have been composed of relatively open stands dominated by large ponderosa pine, Douglas-fir, and white fir trees. The understory contained saplings of white fir, Douglas-fir, aspen in some places, Gambel oak (*Quercus gambelii*), and other shrubs. This kind of stand structure was maintained in part by recurrent fires of relatively low to moderate intensity, although patches of stand-replacing fire probably also were present. Research to date is not sufficient to permit any confident estimates of how large or how frequent these patches of stand-replacing fire may have been. Similar stand structures and dynamics (but different composition in places) have been described for mixed conifer forests in northern Arizona (Fule et al. 2003, Mast and Wolf 2004), western Montana (McCune 1983), the eastern Cascades of Washington and Oregon (Agee 1993), eastern California (Taylor 1993, Taylor and Skinner 1998, Urban et al. 2000, Stephens 2001) Baja California (Stephens et al. 2003, Stephens and Gill 2005), and in the southwestern region as a whole (Jones 1974).

We documented pre-1880 fire history in three warm-dry mixed conifer stands (Monument, Taylor Creek, and Burnette Canyon; see Tables II-1 and II-2 in Chapter II), as part of a broader study of fire history in ponderosa pine forests of the San Juan National Forest (Grissino-Mayer et al. 2004). Median fire intervals in these three stands during the pre-1870 reference period ranged from 18 - 28 years when all fires are considered, or from 21 - 27 years for extensive fires that scarred at least 50 % of recorder trees. (See Chapter II for a discussion of fire history methodology and statistics.) Note that there is little difference in these statistics for all fires and for extensive fires only, indicating that most fires in warm-dry mixed conifer forests were relatively extensive. This is in contrast to fire history in pure ponderosa pine forests where many fires were very small and patchy, such that median intervals for extensive fires were about twice as long as the intervals for fires of any size (Chapter II). When we considered all fire intervals (rather than the median intervals referred to above), we determined that ninety percent of fire intervals during the reference period would be expected to lie between 3 and 73 years for fires of any size, or between 4 and 82 years for extensive fires. These intervals are longer than the intervals documented for pure ponderosa pine forests (Chapter II), but substantially shorter than intervals between successive fires in spruce-fir forests (Chapter III). With respect to fire intervals, as with many other ecological characteristics, mixed conifer forests are intermediate between the extremes of low-elevation and high-elevation forests of the South Central Highlands Section.

Wu (1999) conducted a similar fire history study at three sites in the Pagosa District of the San Juan National Forest. The sites were predominantly

warm-dry mixed conifer forest, although portions of two of the sites could be classified as cool-moist mixed conifer. Median fire intervals (for fires that scarred ≥25% of recorder trees) at these three sites prior to 1880 ranged from 18-41 years (Table 3.1, p. 74 in Wu 1999). These intervals are similar to those found by Grissino-Mayer et al. (2004), but the greater range in Wu's study further illustrates the variability in fire history of mixed conifer forests.

Roughly half of the fire scars detected in the three mixed conifer stands sampled by Grissino-Mayer et al. (2004) were formed during the summer growing season, and half were formed during the dormant season, either spring or early fall (Table II-4 in Chapter II). Similarly, Wu (1999) found that most fires burned either in the dormant season or early in the growing season. These fires apparently killed few of the mature canopy trees, but probably killed most of the small understory trees and shrubs. During the longer intervals between fires, the understory species increased in density and size, and some white fir and Douglas-fir grew into the canopy. However, these species are relatively fire-sensitive, especially when small, and many individuals were killed by the next fire. In contrast, the mature ponderosa pine and Douglas-fir trees have thick bark that would enable them to survive most fires. Large, old white fir also have relatively thick bark, and some of these probably also survived low to moderate intensity fires. Occasionally, especially under conditions of severe and prolonged dry weather, the warm-dry mixed conifer stands probably were subjected to intense crown fires that killed even the mature pine and Douglas-fir (Fule et al. 2003).

It has been suggested that recruitment of ponderosa pine, and possibly also Douglas-fir, may have been at least partially dependent on periodic, low-intensity fires of the kind documented in warm-dry mixed conifer forests; the reasoning is that such fires would create suitable seed beds but not kill the seed source in the canopy. However, Wu (1999) found only weak correlations between local fire dates and tree establishment. Abundant aspen recruitment (sprouting from surviving root systems – Chapter IV) occurred consistently after fire, but ponderosa pine, Douglas-fir, and white fir established after some fires in some locations and not in others. More recent research by Brown and Wu (2005) indicates that ponderosa pine recruitment in the San Juan Mountains is more closely associated with moist climatic periods than with fire. Nevertheless, when climatic conditions are suitable, pine seedlings may become established most readily in an environment of bare mineral soil and freedom from competition. These conditions are created by a low-severity fire, but they gradually disappear within several years after fire. Thus, it has been suggested that the canopy was composed of several, relatively discrete age classes of pine and Douglas-fir that had established after fires in the past—although Wu's (1999) age structure data do not support this idea. In contrast to the shade-intolerant pine, white fir seedlings establish readily in a litter-covered soil under moderately shady conditions. Thus, there was continual recruitment of white fir during the intervals between fire, but heavy mortality of fir in all age classes whenever fire occurred (Wu 1999). Aspen and Gambel oak also are easily top-killed by fire, but both

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reproduce vegetatively via root sprouting. These two species usually produce a large cohort of new stems immediately after a fire, but also recruit new stems for at least a portion of the time between successive fires (as long as the canopy remains relatively open). A more extensive study of mixed conifer age structure and fire history is underway in the Piedra region of the San Juan National Forest (Brown and Wu, unpublished data), which will improve our understanding of relationships between tree establishment, fire, and climate.

2. **Cool-Moist** or **Cold-Wet** Mixed Conifer Forests

The **cool-moist** or **cold-wet** phase of mixed conifer forests historically had a very different stand structure and disturbance regime than the warm-dry mixed conifer forests. **Cool-moist** or **cold-wet** mixed conifer forests grow in areas that are too wet for the frequent fires that characterize the lower-elevation forest types. Late-lying snowpacks and more frequent summer rains keep fuels moist throughout most of the fire season in most years. Consequently, stands regularly may persist for many decades or centuries without fire. When fires finally do occur during prolonged dry periods, the fuel bed that has developed during the long fire-free period can support a high-intensity fire that kills most of the above-ground vegetation in the stand. Carissa Aoki (Colorado State University, unpublished data) is completing a study of fire history in a **cool-moist** mixed conifer forest near Pagosa Springs, Colorado. Pre-1900 fires in this area burned at mixed severity, with patches of lethal fire interspersed with patches of low tree mortality, resulting in a spatially complex and temporally variable landscape mosaic of stands having a spectrum of structural and compositional characteristics.

The successional process that may occur after lethal fires in **cool-moist** or **cold-wet** mixed conifer forests is extremely variable, and depends on the nature of the stand that burned as well as the details of fire intensity and behavior. If the fire is large and intense enough to kill all of the conifers and conifer seed in the canopy, then the stand may come back as an aspen forest. In a smaller or less intense fire, sufficient conifer seed may survive in the crowns of the fire-killed trees or may be dispersed into the area from nearby survivors to restock the stand at something very nearly like its pre-fire composition. Therefore, it is possible to find a wide range of variation in stand structure within **cool-moist** or **cold-wet** mixed conifer forests, as a result of variation in the time since the last fire, in the nature of that last fire, and in the nature of the forest that burned. At a landscape scale, **cool-moist** or **cold-wet** mixed conifer forest may include conifer-dominated stands, mixed stands of conifers and aspen, and aspen-dominated patches.

3. Other Agents Of Disturbance In Mixed Conifer Forests:

In addition to fire, other important agents of natural disturbance in all mixed conifer forests during the reference period included windthrow, fungal diseases, and outbreaks of various insect species. Although these disturbances were important in localized areas, they probably had far less impact on landscape-scale forest dynamics than fire – with the exception of spruce

budworm outbreaks, which may have been as important as fire.

The western spruce budworm (*Choristoneura occidentalis*) is a native insect that feeds on the new foliage of conifers, and can kill or reduce growth rates in host trees. Despite its name, the spruce budworm usually does not feed on spruce trees, but prefers instead white fir, subalpine fir, and Douglas-fir (Furniss and Carolin 1977, Hadley and Veblen 1992, Swetnam and Lynch 1994). Low-level, endemic populations of spruce budworm are nearly always present in mixed conifer and spruce-fir stands, where they have little impact on stand structure or dynamics. However, periodic outbreaks of either local or regional extent may suppress growth or kill great numbers of trees over large areas (Swetnam and Lynch 1994, Schmid and Mata 1996). The spruce budworm kills predominantly smaller fir or Douglas-fir trees and saplings. Individual Douglas-fir trees differ substantially in their resistance or susceptibility to western spruce budworm defoliation (Chen et al. 2001). Spruce budworm outbreaks occurred at intervals of 20 - 33 years in southern Colorado and northern New Mexico during the 18th and 19th centuries (Swetnam and Lynch 1994). The most recent major outbreak in the San Juan National Forest was in the late 1970s and early 1980s (personal observations).

With respect to disturbance history and landscape-scale dynamics, the mixed conifer forests probably are the least well understood of all of the forest types in the South Central Highlands Section. One reason is their inherent compositional variability and complexity. Another is the fact that – until the Missionary Ridge and Million fires in 2002 -- there were few recent fires in this forest type to provide us with direct information on fire effects and biotic responses. Post-fire studies in the areas burned in 2002 will provide important insights into fire effects and ecosystem responses to fires of variable intensity throughout a variety of mixed conifer forest stands. We also need additional research on disturbance history and successional processes in the full range of mixed conifer forest types. The Williams Creek Research Natural Area is an excellent potential location for reconstructing stand dynamics in **cool-moist** or **cold-wet** mixed conifer forests, but this potential has not yet been realized. As discussed in Chapter VII on management challenges and opportunities, we also can gain important insights into the basic ecology of this forest type by including a research component in upcoming timber harvests and other management activities.

4. Alternative Successional Trajectories in Mixed Conifer Forests

One of the key questions that should be addressed in future research is the extent to which stand composition and structure vary over time on any individual site within the mixed conifer forest zone. Jones (1974) suggested that different successional trajectories could occur in mixed conifer forests, depending on local site conditions, the nature and intensity of the disturbance, and other events (e.g., herbivory) happening after disturbance. Unfortunately, Jones' prescient ideas have not been further tested, and this question remains very uncertain. We suggest that the physical environments of the lowermost and uppermost portions of the mixed conifer zone in the South

Central Highlands Section can support only a **warm-dry** or a **cold-wet** type of mixed conifer forest, respectively. However, we hypothesize that the environment within the broad middle portion of the mixed conifer zone (the **cool-moist** mixed conifer phase in the three-phase classification) is potentially suitable for the entire suite of mixed conifer forest structures -- from warm-dry mixed conifer forest, to cold-wet mixed conifer forest, to nearly pure aspen forest. We hypothesize further that many individual sites have switched back and forth among these structural types over the last thousands of years in response to variability in disturbance frequency, intensity, and type. This hypothesis is summarized in Figure V-1, using the three-phase classification of mixed conifer forests. We emphasize that this figure is intended mainly for heuristic purposes; the model of alternative successional trajectories that it depicts has not yet been adequately tested.

We begin with a "generic" mixed conifer stand (upper center portion of Figure V-1). Such a stand may contain any of the major tree species of the mixed conifer zone, and may exhibit a wide array of structural and compositional characteristics, depending on its past history. This generic mixed conifer stand may initially undergo any one of three major successional trajectories, in response to the environmental gradients of elevation, aspect, and micro-climate (temperature and moisture).

The trajectory to the left of the generic mixed conifer forest in Figure V-1 takes one to warm, dry sites that normally occur at lower elevations and often on southerly aspects, and leads to the development of a "classic" **warm-dry mixed conifer forest** that has a relatively open canopy and that is maintained by non-lethal fires. If non-lethal fires occur at relatively short intervals (3 to 30 years), the canopy remains dominated by large ponderosa pine, Douglas-fir, and white fir, and the open stand structure provides opportunities for the shade-intolerant ponderosa pine component to regenerate when favorable climatic conditions occur. Thus, the classic **warm-dry** mixed conifer forest structure is maintained in part by recurrent, non-lethal fires, especially on relatively mesic sites at lower elevations that potentially could support a canopy of shade-tolerant species in the absence of disturbance.

If non-lethal fires occur in **warm-dry mixed conifer forests** at very long intervals (hundreds of years), or occur on the long end of the fire return interval (40 to 73 years), many of the understory individuals (including ponderosa pine) will grow into the overstory canopy, and thus perpetuate the species composition and the regeneration potential of the warm-dry mixed conifer type. This latter scenario however may reduce the competitiveness of the ponderosa pine component in the future, as the more closed overstory canopy will favor the regeneration of the more shade-tolerant white and Douglas fir trees.

The trajectory moving further down from the **warm-dry** phase on the left side of Figure V-1 results from lethal fire and produces, at least initially, a **seral Gambel oak-dominated shrubland**. Because Gambel oak re-sprouts readily and abundantly after fire, whereas the conifers have to become reestablished more slowly via seed germination, the stand is initially

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MIXED CONIFER FORESTS – SUCCESSIONAL TRAJECTORIES

KEY: Pipo (ponderosa pine), Psme (Douglas fir), Abco (white fir), Pien (E. spruce), Ablaa (subalpine fir), Potr (aspen), Quga (Gambel oak), SS = Sprouting Shrubs (Gambel oak, bitterbrush, buckbrush, snowberry, chokecherry, fendlerbush, mountain mahogany, Woods rose, serviceberry, skunkbrush, squawapple)

Natural Selection ⟶ High Intensity Stand Replacement Fire ➡

Low Intensity Fire – – – –> No Fire ·············▶

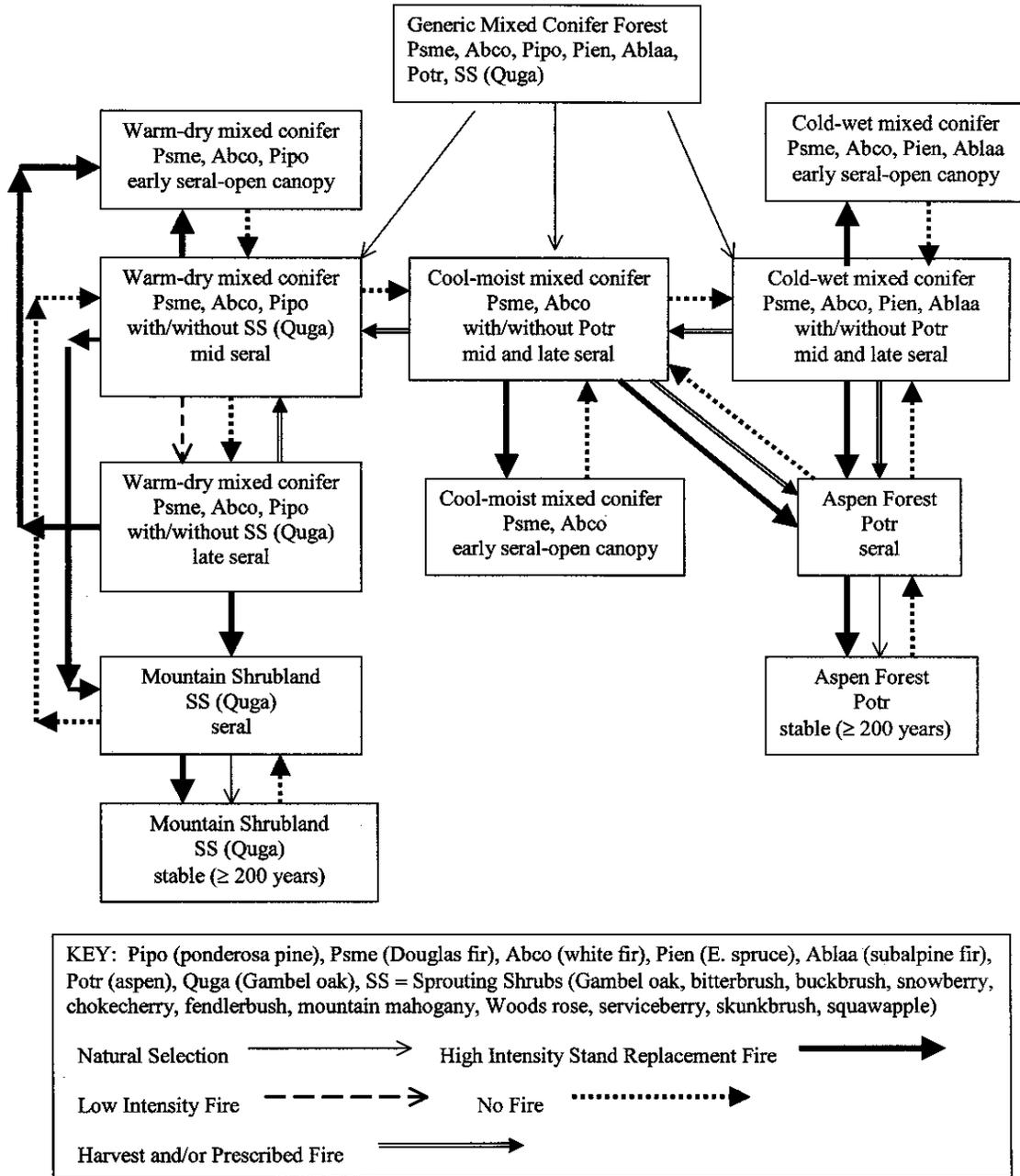Harvest and/or Prescribed Fire ⟹

Figure V-1. Hypothesized alternative successional trajectories in mixed conifer forests of the South Central Highlands Section. More research is needed to test this model.

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converted to a Gambel oak-dominated community. If another fire does not occur within the next few centuries, then succession may return the stand to something like its original mixed composition. However, if the Gambel oak stand burns again within 50 years or so (i.e., before the young conifers become reproductive), then the stand may be converted to a **stable Gambel oak-dominated shrubland** that persists for a very long time (potentially centuries) even without subsequent fires (see Chapter VI). Hanks and Dick-Peddie (1974) suggest a similar prolonged successional stage dominated by *Quercus gambelii* or *Robinia neomexicana* in the White Mountains of southeastern New Mexico.

The trajectory to the right of the generic mixed conifer forest in Figure V-1 takes one to cold, wet sites that normally occur at higher elevations and often on northerly aspects, and leads to the development of a "classic" **cold-wet mixed conifer forest** that has a closed canopy and a very long fire-free disturbance interval (hundreds of years). Stands are dominated by shade-tolerant white and subalpine firs, Engelmann spruce, and Douglas-fir in both the overstory canopy and understory strata. This forest structure may persist indefinitely in the absence of disturbance. We note that Douglas-fir is somewhat less shade-tolerant than the firs and spruce, so it may gradually decrease in abundance over extremely long time periods without major disturbance. However, because we currently have no detailed age structure data from very old stands of **cold-wet** mixed conifer forest, the trends described here are necessarily speculative. A high research priority should be to locate and sample age

structure in several old stands of undisturbed **cold-wet** mixed conifer forest, to improve our understanding of long-term successional dynamics in this little-studied forest type.

The trajectory moving straight down from the generic mixed conifer forest in Figure V-1 takes one to cool, moist sites that normally occur at elevations between those of the **warm-dry** and **cold-wet mixed conifer phases,** and leads to the development of a "classic" **cool-moist mixed conifer forest** that has a relatively closed canopy and a long fire-free disturbance interval (many decades to centuries). Stands are dominated by shade-tolerant white fir and Douglas-fir in both the overstory canopy and understory strata. This forest structure may persist indefinitely in the absence of disturbance, although Douglas-fir is somewhat less shade-tolerant than white fir, so it may gradually decrease in abundance over extremely long time periods without major disturbance.

We must stress again that the successional trajectories in Figure V-1 are partly speculative. The mixed conifer forest type is perhaps the most complex and least well understood of all the types in the South Central Highlands Section. It appears that a wide range of stand structures and dynamics is possible within this elevational zone, depending on local climatic and soils conditions and on the local disturbance history. The mixed conifer forest type contains elements of the ecology of both the ponderosa pine forests at lower elevations and the spruce-fir forests at higher elevations. A high priority for future research should be to rigorously test ideas like those embodied in Figure V-1, to improve our understanding of the long-term successional dynamics and

inherent variability of mixed conifer forests in the South Central Highlands Section.

5. Landscape Patterns in Mixed Conifer Forests of the Hermosa Unit

We can gain a glimpse into the natural patterns and processes of mid-elevation landscapes of the South Central Highlands Section by examining the Hermosa unit, a roadless area located just north of Durango in the San Juan National Forest. This 80,000-acre area is relatively pristine; no logging has occurred, except around the fringes, but fires have been suppressed, and livestock grazing and recreation are important uses. In 1993, a team of Fort Lewis College researchers sampled floristic composition, stand structure, and stand age in 85 forest stands (10x30 m) distributed throughout the Hermosa area (Romme et al. 1996). The sample stands were stratified into five general sub-regions, which represented the full range of elevation and topographic conditions within the study area.

The Hermosa unit contains mostly steep, rugged topography underlain by finely inter-bedded sandstones, shales, and limestones of the Hermosa Formation (Pennsylvanian Period). Elevations range from 2,070 m (6,850 ft) in the Jones Creek region to 3,740 m (12,340 ft) in the Monument Hill region on the north slopes of the La Plata Mountains. In this study the vegetation was classified according to habitat types – a classification system based on the species that would dominate a site following a very long period without disturbance (DeVelice et al. 1986). A great variety of mixed conifer forest types are represented in the Hermosa area: the *Abies concolor*, *Pseudotsuga menziesii*, and *Picea pungens* series at middle elevations (corresponding to **cool-moist** mixed conifer forests), the *Pinus ponderosa* series at the lowest elevations (corresponding to ponderosa pine and **warm-dry** mixed conifer forests), the *Abies lasiocarpa* and *Picea engelmannii* series at the highest elevations (corresponding to **cold-wet** mixed conifer forests or spruce-fir forests), and riparian series along perennial streams (DeVelice et al. 1986, Romme et al. 1996).

The structure, composition, and distribution of plant communities are strongly controlled by gradients in elevation and topography. At lower elevations, below about 2,700 m (9,000 ft), south-facing slopes are dominated either by open forests of ponderosa pine with Gambel oak in the understory, or by oak-dominated shrublands. North-facing slopes, at this elevation, generally support closed forests of ponderosa pine and Douglas-fir. Middle elevations (2,700 – 3,000 m (9,000 - 10,000 ft)) are mostly forested with Engelmann spruce, subalpine/corkbark fir, and Douglas-fir stands on north-facing slopes, and predominantly aspen stands on south-facing slopes. At the highest elevations (>3,000 m (10,000 ft)) the forests are mostly spruce-fir or aspen, with alpine meadows and rock fields at elevations above timberline. Narrow strips of riparian forest extend along major creeks and support a diverse assemblage of trees, shrubs, and herbs. Small meadows are found throughout the area on deep alluvial soils along streams or on shallow soils on ridgetops and south-facing slopes.

The richness of vascular plants (number of species per 10 x 30 m plot) within the Hermosa study area was

strongly influenced by elevation. The greatest richness (40 + species per plot) was seen most often in stands at lower to middle elevations (<2,700 m (9,000 ft)), whereas higher elevation stands (>2,700 m (9,000 ft)) usually had < 30 species per plot (Table V-3). Aspect, steepness of the slope, and forest habitat type did not have significant effects on species richness.

The Hermosa landscape has been shaped not only by gradients in elevation and topography, but also by past fires. Both of the major types of fire regimes that characterize the South Central Highlands Section -- the low-elevation fire regime of frequent low-severity fires, and the high-elevation fire regime of infrequent high-severity fires -- were important in the Hermosa landscape during the pre-1870 reference period. The pure ponderosa pine forests and **warm-dry** mixed conifer forests were affected primarily by frequent, low-severity fires that maintained an open stand structure with a broad range of tree sizes and ages. In contrast, the **cold-wet** mixed conifer forests, spruce-fir forests, and aspen forests, all were influenced mainly by infrequent stand-replacing fires that created a mosaic of stands of varying ages across the middle and high elevation portions of the landscape. However, it is important to recognize that low-severity fires occasionally occurred in the high elevation stands, and intense fires sometimes killed all of the trees in patches of the low elevation forests. The result is a complex and variable landscape pattern that apparently was characteristic of mixed conifer forests throughout the South Central Highlands Section during the period of indigenous settlement.

Table V-3. Richness of vascular plants (number of species per 10 x 30 m plot) in the Hermosa study area as a function of elevation zone. The numbers in the cells of the table depict the number of sample stands within each elevational zone that had the indicated number of vascular plant species.

Number of	Elevation (feet)			
Species	< 8,000	8,000 - 9,000	9,001 - 10,000	> 10,000
< 30	1	8	9	10
30 - 39	3	20	5	3
40 +	7	16	1	2

Spatial Patterns in Fire History in the Hermosa Unit:

Although the disturbance history of the Hermosa unit is complex and not completely understood, we can identify some of the major spatial patterns in fire history and fire effects from the distribution of stand ages in different parts of the landscape. We collected 5 - 10 increment cores from the dominant trees in each of the 85 sample stands, and estimated the approximate date of stand origin from the age structure of the

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dominant tree cohort. This was possible in most of the higher-elevation stands, and some lower-elevation stands, where the last disturbance was severe enough to be followed soon afterwards by establishment of a more or less even-aged cohort of trees. It was not possible in many lower-elevation stands, and some higher elevation stands, where repeated low-severity fires had maintained an all-aged tree population. In many stands, especially at the higher elevations, the last stand-replacing disturbance had occurred so long ago that the initial post-fire tree cohort was no longer intact or recognizable. We were unable to assign a precise stand age to these stands, but we did determine that they were at least as old as the oldest tree sampled (see Romme and Knight 1981, and Chapters III and IV in this report, for further discussion of the issues related to estimating stand age in Rocky Mountain forests).

Minimum, maximum, and median stand ages generally increased with increasing elevation (Table V-4), although the oldest stand actually was found at a middle elevation. This trend indicates that fire intervals gradually lengthen as elevation increases, probably because of wetter conditions at the higher elevations. This interpretation is further supported by the very different fire frequencies that we have documented for ponderosa pine forests at the lowest elevations and spruce-fir forests at the highest elevations (Chapters II and III). What is interesting here is that the mixed conifer forest type is intermediate between these two extremes of fire regimes and exhibits some characteristics of each extreme.

One reason why stand ages do not increase monotonically with elevation is because of the effects of aspect. At any elevation the north-facing slopes are relatively cooler and wetter than the south-facing slopes. Accordingly, minimum, maximum, and median stand ages are greater on mesic north-facing aspects than on xeric south-, west-, and east-facing aspects (Table V-5).

Stand ages also differed among habitat types. Minimum, maximum, and median stand ages were greatest in spruce-fir habitat types (*Abies lasiocarpa* series), somewhat less in **cool-moist** mixed conifer types (*Abies concolor, Pseudotsuga menziesii*, and *Picea pungens* series), and least in the aspen types (Table V-6). This pattern probably reflects the gradient in moisture conditions from relatively dry aspen forests (mostly on south-facing slopes in the Hermosa area) to the wettest conditions in high-elevation spruce-fir forests (mostly on north-facing slopes in the Hermosa area). Once again, the mixed conifer forests are intermediate in this gradient from wet to dry.

In addition to effects of elevation, aspect, and habitat type, landscape context (location of a stand with respect to other elements of the landscape) influences fire intervals. The minimum, maximum, and median stand ages are greater in stands located on the west side of Hermosa Creek than in stands located on the east side (Table V-7). This indicates that fire intervals have been generally longer on the west side of the creek, even though elevations are comparable on both sides. The mechanism for this difference in fire history may be related to the presence of a high alpine ridge (Indian Trail Ridge) that extends along the western border of the study area. The ridge may act as a barrier to fires spreading into the

Hermosa area from the west (the direction of prevailing winds). Thus, fires would have to be ignited directly within the forests on the west side of Hermosa Creek to burn in that area. In contrast, fires ignited almost anywhere within the Hermosa unit could be driven by the wind into the area on the east side of the creek. Subtle variation in disturbance history, similar to this example in the Hermosa area, most likely characterize mixed conifer forests throughout the South Central Highlands Section.

Table V-4. Stand ages (time since last stand-replacing disturbance) with respect to elevation in mixed conifer forests of the Hermosa study area.

Elevation (feet)	Number of Stands	Minimum Stand Age	Median Stand Age	Maximum Stand Age
< 8,000	4	85	115	160
8,001 - 9,000	22	95	183	> 322
9,001 - 10,000	8	50	138	220
> 10,000	4	195	220	> 243

Table V-5. Stand ages (time since last stand-replacing disturbance) with respect to aspect in mixed conifer forests of the Hermosa study area.

Aspect	Number of Stands	Minimum Stand Age	Median Stand Age	Maximum Stand Age
Xeric (east, southeast, south, southwest, & west)	19	50	135	> 311
Mesic (north, northwest, & northeast)	18	100	198	> 322

Table V-6. Stand ages (time since last stand-replacing disturbance) with respect to habitat type series (DeVelice et al. 1986) in mixed conifer forests of the Hermosa study area.

Habitat Type Series	Number of Stands	Minimum Stand Age	Median Stand Age	Maximum Stand Age
Populus tremuloides	3	50	95	120
Abies concolor	10	70	125	> 296
Abies lasiocarpa	23	110	200	> 322

Table V-7. Stand ages (time since last stand-replacing disturbance) with respect to landscape context (location east or west of Hermosa Creek) in mixed conifer forests of the Hermosa study area.

Landscape Context	Number of Stands	Minimum Stand Age	Median Stand Age	Maximum Stand Age
East of Hermosa Creek	22	50	158	> 320
West of Hermosa Creek	16	95	220	> 322

C. LEGACIES OF EURO-AMERICAN SETTLEMENT AND CURRENT CONDITIONS

The major Euro-American impacts on **warm-dry** mixed conifer forests are similar to impacts in ponderosa pine forests, which are detailed in Chapter II of this report. Impacts in **cool-moist** or **cold-wet** mixed conifer forests are similar to impacts in spruce-fir forests (Chapter III). A brief summary of the major effects is provided below.

1. Legacies of Grazing and Fire Exclusion

The introduction of large herds of cattle and sheep, beginning in the 1870s, led to possibly profound changes in the herbaceous component of the forest communities (Fleischner 1994). Livestock grazing also removed fine herbaceous fuels that formerly carried low-intensity surface fires over extensive areas, and resulted in a relatively sudden and dramatic drop in fire frequency in the late 19th century (Swetnam and Baisan 1996). The Forest Service also actively suppressed fires throughout most of the 20th century. As a result of heavy grazing and active fire control, many mixed conifer forest stands that formerly burned every 5 - 40 years, now have had no fire for over 100 years. Livestock use on the National Forests has been regulated since the early 1930s, but many areas at lower elevations (including ponderosa pine and **warm-dry** mixed conifer forests) may still show the effects of excessive grazing at the turn of the 20th century. These effects include altered herbaceous communities and unusually long fire intervals.

In contrast with the striking impacts of Euro-American settlement in lower elevation forests, the fire regimes of **cool-moist** or **cold-wet** mixed conifer and other high elevation forests, were not so strongly affected by early grazing and fire suppression. Fire frequency and behavior in high elevation forests were controlled primarily by weather conditions, not fuels, and most fires that covered large areas were severe, stand-replacing fires. Thus, removal of fine herbaceous fuels by livestock grazing did not have as much impact on fire spread in these crown-fire ecosystems as it did in lower elevation forests. There were few large fires during the 20th century in cool, wet mixed conifer and

other high elevation forests of the South Central Highlands Section, but this may be due as much to wet weather conditions and limited ignitions as to active fire suppression efforts. It should be noted that fire exclusion at lower elevations (e.g., ponderosa pine forests) may partially explain the lack of fires at higher elevations during the last century; in previous centuries, fires may have commonly ignited at lower elevations and then spread into higher elevations when weather conditions were suitable. This hypothesis needs further testing, however.

2. Legacies of Logging and Road Building

Extensive logging occurred throughout the lower elevation forests of the South Central Highlands Section, beginning in the late 1800s and especially during the first half of the 20th century. In nearly all of the accessible stands of ponderosa pine forest, the large trees were selectively removed. As a result, current stand structures are dominated by younger trees that became established over the first three decades of the 20th century. Much of the **warm-dry** mixed conifer forest zone also was logged in this manner. The combination of fire exclusion, selective logging, and favorable climatic conditions for young tree establishment in the early 20th century, has created an unusual stand structure in many warm-dry mixed conifer forests today. The large, old ponderosa pine and Douglas-fir trees that formerly dominated the canopy are now gone, and the stands are dominated by smaller, young individuals of pine, Douglas-fir, and white fir. White fir especially has increased in density

during the long fire-free period of the 20th century, and establishment of new ponderosa pine and Douglas-fir individuals has tapered off or stopped in many stands, probably because of the dense stand conditions (Wu 1999). A comparison of historic photos from the San Juan Mountains with the same scenes today reveals increased conifer density in many places (Zier and Baker 2006). However, the exact mechanism underlying the increase in conifer density cannot be determined from the photos alone, and it may be a combination of the factors described above. Moreover, we must stress that not all forests in the South Central Highlands Section have become more dense in the last century: Manier et al. (2005) compared forest cover in 1937 and 1994 on the Uncompahgre Plateau (Figure I-1), and found a very slight net change, as small increases in conifer cover in some areas were balanced by small decreases in other areas, with a complex spatial pattern of increases and decreases.

In contrast to lower-elevation forests, the **cool-moist** or **cold-wet** mixed conifer forests and other high elevation forest types were not subjected to extensive logging until the middle of the 20th century in many portions of the South Central Highlands Section. Nearly all of the timber harvests in mixed conifer forests have involved some form of partial cutting; little clear-cutting has occurred in this forest type. Both partial cutting and clear-cutting usually require roads for access. The history and impacts of logging and road-building on the San Juan National Forest are discussed in Chapter VII.

Many of the partial cuts in mixed conifer forests during the 1950s - 1970s emphasized selective removal of the

high-value canopy trees, with little disturbance to the smaller trees or the species of lower economic value. This kind of treatment created some unusual stand structures that do not resemble the kinds of structures resulting from natural disturbances. In **warm-dry** mixed conifer forests, for example, low-severity fires formerly removed mainly smaller tree size classes and left most of the canopy dominants intact. The fires also created suitable seedbeds for pine and Douglas-fir regeneration. Some of the selective logging operations of the 1950s - 1970s did just the reverse: they removed the canopy dominants and left the understory intact, and did not create forest floor conditions conducive to pine or Douglas-fir regeneration.

Similarly, in some **cool-moist** or **cold-wet** mixed conifer forests, selective removal of the large Douglas-fir produced stand structures that would not develop through natural successional processes. Natural fires or budworm outbreaks would kill a large proportion of small trees, as well as some large ones, and would affect both Douglas-fir and true firs. In contrast, partial timber harvests of the 1950s - 1970s generally focused on large individuals of commercially valuable species, and left most of the true firs and small stems of all species. Some of these stands are now dominated by a dense growth of small to moderate sized firs and shrubs, with little or no Douglas-fir regeneration. Douglas-fir may not become re-established in such stands for centuries, unless fire or restoration treatment is applied. See Chapter VII for some ideas on how to deal with these issues.

D. SUMMARY

The mixed conifer forest type occurs at elevations ranging from about 2270 – 3030 m (7,500 – 10,000 ft). Mixed conifer forests are perhaps the most variable and complex of any forest type in the southwestern mountains, in terms of species composition, stand structure, and dynamics. They also have received relatively little research attention Consequently, we have a relatively poor understanding of the long-term ecological dynamics and interactions that have shaped the mixed conifer landscape in the past, and that explain biotic responses to current management activities.

Environmental conditions, species composition, and disturbance regimes in the mixed conifer zone are intermediate between characteristics of the adjacent ponderosa pine zone at lower elevations and characteristics of the adjacent spruce-fir zone at higher elevations. Mixed conifer forests in the South Central Highlands Section can be divided into either two or three general sub-types or "phases." The two-phase classification recognizes a **warm-dry** phase, generally at lower elevations, and a **cool-moist** phase, generally at higher elevations. **Warm-dry** mixed conifer forests resemble ponderosa pine forests in general stand structure, but Douglas-fir and white fir are also important components of these forests. **Cool-moist** mixed conifer forests typically lack ponderosa pine, have a greater abundance of Douglas-fir and white fir, and, on some sites, include subalpine/corkbark fir and Engelmann spruce. The three-phase classification of mixed conifer forests retains the same

warm-dry phase, but subdivides the cool-moist phase into two units: a **cool-moist** phase with the same name as before but a different definition, and a new **cold-wet** phase. Ponderosa pine is the key indicator of the **warm-dry** phase, and subalpine/corkbark fir and Engelmann spruce are the key indicators of the **cold-wet** phase. Douglas-fir and white fir are the constant species occurring throughout the mixed conifer forest type in this three-phase classification, while ponderosa pine, Engelmann spruce, and subalpine/corkbark fir are absent or only minor components in the **cool-moist** mixed conifer phase of the three-phase classification.

During the period of indigenous settlement, many of the **warm-dry** mixed conifer forests of the South Central Highlands Section appear to have been composed of relatively open stands dominated by large ponderosa pine, Douglas-fir, and white fir trees. The understory contained saplings of white fir, Douglas-fir, aspen in some places, Gambel oak, and other shrubs. This kind of stand structure was maintained in part by recurrent fires of relatively low to moderate intensity, although patches of stand-replacing fire probably also were present, and relatively dense stands could be found in places. Fire intervals in **warm-dry** mixed conifer forests typically were measured in decades, and were longer than in ponderosa pine forests but shorter than in forest types at higher elevations. **Cool-moist** or **cold-wet** mixed conifer forests grow in areas that are too wet for the frequent fires that characterize the lower-elevation forest types. Late-lying snowpacks and more frequent summer rains keep fuels moist throughout most of the fire season in

most years. Consequently, stands regularly may persist for many decades or centuries without fire. When fires finally do occur during prolonged dry periods, the fuel bed that has developed during the long fire-free period can support a high-intensity fire that kills most of the above-ground vegetation in the stand. Pre-1900 fires in **cool-moist** or **cold-wet** mixed conifer forests occurred at intervals of many decades to centuries, and typically burned at mixed severity, with patches of lethal fire interspersed with patches of low tree mortality. The result was a spatially complex and temporally variable landscape mosaic of stands having a spectrum of structural and compositional characteristics.

In addition to fire, other important agents of natural disturbance in all mixed conifer forests during the reference period included windthrow, fungal diseases, and outbreaks of various insect species. Although these disturbances were important in localized areas, they probably had far less impact on landscape-scale forest dynamics than fire – with the exception of spruce budworm outbreaks, which may have been as important as fire.

The major Euro-American impacts on **warm-dry** mixed conifer forests are generally similar to impacts in ponderosa pine forests. Extensive and unregulated livestock grazing, which began in the late 1800s, removed fine herbaceous fuels that formerly carried low-intensity surface fires, and resulted in a relatively sudden and dramatic drop in fire frequency. Today, many **warm-dry** mixed conifer stands have not burned in over one hundred years, and many have greater tree densities than was typical of the reference period. Euro-American impacts on **cool-moist**

or **cold-wet** mixed conifer forests are generally similar to impacts in spruce-fir forests. Fire regimes of **cool-moist** or **cold-wet** mixed conifer were not so strongly affected by early grazing and fire suppression. Fire frequency and behavior in these and other high elevation forests were controlled primarily by weather conditions, not fuels, and most fires that covered large areas were severe, stand-replacing fires. There were few large fires during the 20th century in **cool-wet** mixed conifer and other high elevation forests of the South Central Highlands Section, but this may be due as much to wet weather conditions and limited ignitions as to active fire suppression efforts. Many **cool-moist** or **cold-wet** mixed conifer forests have high tree densities today, but this does not represent a major departure from historical conditions in these forests.

Extensive logging occurred throughout the lower elevation forests of the South Central Highlands Section, beginning in the late 1800s and especially during the first half of the 20th century. In nearly all of the accessible stands of ponderosa pine and **warm-dry** mixed conifer forest, the large trees were selectively removed. The combination of fire exclusion, selective logging, and favorable climatic conditions for young tree establishment in the early 20th century, has created an unusual stand structure in many **warm-dry** mixed conifer forests today. The large, old ponderosa pine and Douglas-fir trees that formerly dominated the canopy are gone, and the stands now are dominated by smaller, young individuals of pine, Douglas-fir, and white fir. White fir especially has increased in density during the long fire-free period of the 20th century, and establishment of new ponderosa pine and Douglas-fir individuals has tapered off or stopped in many stands, probably because of the dense stand conditions

In contrast to lower-elevation forests, the **cool-moist** or **cold-wet** mixed conifer forests and other high elevation forest types were not subjected to extensive logging until the middle of the 20th century in many portions of the South Central Highlands Section. Nearly all of the timber harvests in mixed conifer forests have involved road building and some form of partial cutting; little clear-cutting has occurred in this forest type. Selective removal of the high-value canopy trees, with little disturbance to the smaller trees or the species of lower economic value, has created some unusual stand structures that do not resemble the kinds of structures resulting from natural disturbances. Some of these stands are now dominated by a dense growth of small to moderate sized firs and shrubs, with little or no Douglas-fir regeneration. Douglas-fir may not become re-established in such stands for centuries, unless fire occurs or an active restoration treatment is applied.

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CHAPTER VI: OTHER VEGETATION TYPES

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In this chapter we examine seven types of vegetation that cover relatively small areas within the mountainous portions of the South Central Highlands Section but are nevertheless of great importance ecologically. Piñon-juniper woodlands (A) are very extensive in the Four Corners region, but occupy only the lower elevation landscapes around the margins of the mountainous area that is the main focus of this report. Mountain grasslands (B) and riparian vegetation and wetlands (C) are found over a large range of elevation throughout the region, but they tend to have patchy distributions and the total area covered by these types of vegetation is small. Alpine ecosystems (D) are found above the upper timberline, and are most extensive in the San Juan Mountains. Mountain shrublands (E) are interspersed within piñon –juniper woodlands and ponderosa pine forests, primarily at the lower elevations throughout the South Central Highlands Section. Lodgepole pine forests (F) are very extensive in the mountains to the north of our study area, but native stands cover only a small area in the northeastern portion of the South Central Highlands Section, and other small stands have been planted elsewhere. Bristlecone pine forests (G), are restricted to high elevations primarily on the east sides of high mountain masses, and cover only a small total area.

Another set of distinctive vegetation types is found in the low-elevation landscapes of the Uncompahgre Plateau, which comprises the northwestern portion of the South Central Highlands Section. These semi-arid vegetation types (e.g., salt desert and semi-desert grassland) are not treated here, because this report focuses on the higher-elevation ecosystems of the mountainous portions of the South Central Highlands Section. However, we provide a detailed treatment of the low-elevation, semi-arid vegetation types in a companion report that focuses on the Uncompahgre Plateau area (Romme, W. H. 2003, Vegetation types, condition classes, and successional trajectories in the Uncompahgre Plateau region, southwestern Colorado, unpublished report to GMUG National Forest, Delta, CO).

A: PINYON-JUNIPER WOODLANDS

1. Vegetation Structure and Composition

Piñon-juniper vegetation covers a vast area in western North America, and exhibits a wide range of stand structures and dynamics (Romme et al. 2009). Between 1650-2300 m elevations in Arizona, New Mexico, Colorado, and Utah, woodlands of *Pinus edulis* and *Juniperus osteosperma* (Colorado piñon pine and Utah juniper) form the dominant vegetative community (Floyd 2003). *Juniperus monosperma* is also important in the southernmost portions of the South Central highlands Section (Romme et al. 2009). At the lowest and

driest elevations, the woodland is represented by only scattered juniper situated on sites that minimize competition from shrubs and herbs. At this low end of the zone, annual precipitation may be less than 30 cm (12 in). Juniper becomes more abundant as mean annual precipitation increases with elevation. Piñon appears and becomes more common than juniper as the average annual precipitation approaches 35 cm (14 in) and increases to 45 cm (18 in). Piñon-juniper stands in this upper portion of the zone grow taller and closer together than in lower-elevation stands. Piñon-juniper stands continue to occupy sites on steep slopes with south exposures up to about 2600 m, especially those protected against fire by cliffs or barren areas. The Utah juniper of lower elevations is replaced by *Juniperus scopulorum* (Rocky Mountain juniper) as moisture increases and mean annual temperature declines. Piñon pine and Rocky Mountain juniper continue as subordinate elements of the Gambel oak-Ponderosa pine community up to about 2900 m.

As a rule, the distribution of piñon-juniper woodland is bounded below by the 30 cm (12 in) isohyet and above by the 45 - 50 cm (18 – 20 in) isohyet. The upper boundary, however, is not a physiological limit for any of the tree species; rather the more moist climate at higher elevations apparently permits relatively more mesophytic species, e.g., ponderosa pine, Gambel oak, and Douglas-fir, to out-compete piñon and juniper.

Precipitation patterns alone do not drive vegetation patterns. A heterogeneous vegetational mosaic also arises out of the complex dissection of the region into canyons and mesas, hills and valleys, and south-facing and north-facing slopes. Even if the land were completely flat, recurring natural disturbances such as fire and disease would create a mosaic of habitats. Differences in soils arising out of various parent rocks, depositional processes, and weathering create diverse microhabitats as well. Long-term climatic variability, as well as local anomalies such as the rain shadow in the lee of Sleeping Ute Mountain near Cortez, Colorado, prevent the establishment of a single, uniform type of vegetation over any large portion of the area. Generally, pinon and juniper form an open woodland on drier sites, but can form a closed-canopy forest on wetter sites, e.g., in parts of the Mesa Verde cuesta. Woodlands may include a prominent component of grass or shrubs, or may simply support very sparse vegetation with much exposed rock or bare soil.

Although piñon-juniper woodlands and forests are often thought of as low-diversity ecosystems, especially where the trees form a closed canopy (Huber et al 1999, Naillon et al. 1999), they can support a rich understory flora. In Mesa Verde National Park, variation in canopy density had little effect on species richness in the understory. In a comprehensive vegetation study involving nearly 400 sampling areas, the number of understory forbs and shrubs was the same in open and dense canopies (averaging 12 species per 100 m^2), with a slightly greater number (16 species per 100 m^2) in the intermediate canopies (Floyd and Colyer 2003). Fungi, mosses, and organisms that form soil crusts also are very diverse in the old piñon-juniper woodlands on the Mesa Verde cuesta (Belnap 2003, Belnap and Lindsey 2003). Similar understory diversity is reported from the Uncompahgre Plateau in western Colorado (P. Lyons, personal

communication) and northeastern Utah (Austin 1999). *Purshia tridentata* (bitterbrush) is the dominant understory shrub on much of the Mesa Verde cuesta, and other common species include Gambel oak, Utah serviceberry, big sagebrush, *Artemisia nova* (black sagebrush), *Fendlera rupicola* (fendlerbush), and *Peraphyllum ramossisimum* (squaw apple). The forb layer varies considerably across elevations and among microsites, but common forbs include *Astragalus scopulorum, Calochortus nuttallii, Commandra umbellata, Cryptantha bakerii, Cymopterus bulbosus, C. purpurescens, Eriogonum racemosa, Lupinus ammophila, Lomatium triternatum, L. grayii, Penstemon linearoides, Pedicularis centranthera, Petradoria pumila,* and *Yucca baccata.* Grasses are abundant in some places, but not in others, depending on local soil conditions and history of grazing and fire. Dense patches of the common *Poa fendleriana* (muttongrass) attain 80% cover in some areas on the Mesa Verde cuesta,, providing soil stability through the plant's intricate root system. Muttongrass may be accompanied by *Agropyron smithii* (western wheatgrass), *A. trachycaulum* (slender wheatgrass), *Koeleria cristata* (june grass), *Oryzopsis hymenoides* (Indian rice grass)*, Sitanion hystrix* (squirrel-tail grass), *Stipa comata* (needle and thread grass), and *S. viridis* (green needle grass).

2. Reference Conditions in Piñon-Juniper Woodlands

In addition to variation in structure and composition related to gradients of elevation, substrate, and topography, piñon-juniper woodlands were shaped in part by a variety of disturbance processes. Three fundamentally different kinds of piñon-juniper vegetation can be identified based on canopy structure, understory characteristics, and historical disturbance regimes (Romme et al. 2008, 2009). The three kinds--**persistent piñon-juniper woodlands**, **piñon-juniper savannas**, and **wooded shrublands**--are summarized in Table VI-A-1. It is important to distinguish among these kinds of piñon-juniper vegetation when discussing reference conditions, because disturbance dynamics (especially with regards to fire) differed greatly among them. There is great diversity within each of these general types with respect to species composition and stand structure, but this classification represents much of the variability in piñon-juniper vegetation across the western U.S. Two types, persistent woodlands and wooded shrublands, are common in the South Central Highlands Section. The third type, savannas, is uncommon in our region but becomes more prevalent in central and southern New Mexico and Arizona. In the sections below we summarize the main features of historical disturbance regimes in the different kinds of piñon-juniper vegetation. See Romme et al. (2008, 2009) for additional details and documentation.

Fire

Piñons and junipers are easily killed by even relatively low-intensity fire, because their bark is relatively thin and provides little insulation for the sensitive cambium located just below the bark. Their foliage also is very flammable, and often low-hanging, such that flames from a fire burning on the ground can sweep into the crowns of the

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Table VI-A-1. Three general types of piñon and juniper vegetation in the western U.S. A field key to the three types is provided in Romme et al. (2007).

Persistent Piñon-Juniper Woodlands

Canopy -- ranging from sparse stands of scattered small trees growing on poor substrates to relatively dense stands of large trees on more productive sites. The canopy may be dominated by either piñon or juniper or both

Understory -- variable cover of shrubs, sub-shrubs, forbs, and grasses, but often sparse with extensive areas of litter (beneath canopies) and bare soil or rock (intercanopy)

Site conditions -- associated with a wide variety of substrates and topographic settings, but most commonly found on rugged uplands with shallow, coarse-textured, and often rocky soils that support relatively sparse herbaceous cover; site conditions (soils and climate) and disturbance regimes (notably infrequent fire) are inherently favorable for tree growth

Regional Distribution -- found in appropriate upland locations throughout the West. Persistent woodlands appear to be especially prevalent on portions of the Colorado Plateau, where precipitation is bimodal with small peaks in winter and summer

Piñon-Juniper Savannas

Canopy -- low to moderate density and cover of piñon or juniper or both

Understory -- well-developed and nearly continuous grass (with forb) cover; shrubs may be present but are usually only a minor component

Site conditions -- typically found on moderately deep, coarse to fine-textured soils in gentle upland and transitional valley settings, or where local conditions are inherently favorable for grasses

Regional Distribution -- especially prevalent in basins and foothills of New Mexico and Arizona where a large proportion of annual precipitation comes during the growing season

Wooded Shrublands

Canopy -- variable tree component that may range from very sparse to relatively dense

Understory -- well developed shrub stratum with variable grass-forb cover and composition; shrubs constitute the underlying biotic community in these ecosystems

Site conditions -- associated with a wide variety of substrates and topographic settings, including shallow rocky soils on mountain slopes to deep soils of inter-montane valleys; site conditions are inherently favorable for shrub growth, thus the tree component naturally waxes and wanes over time in response to a variety of climatic and disturbance factors ("areas of potential expansion and contraction" in Romme et al. 2007)

Regional Distribution -- especially prevalent in the Great Basin, where the precipitation pattern is winter-dominated and *Artemisia tridentata* is a dominant shrub species; however, wooded shrublands can be found throughout the West where local substrates favor shrub dominance

trees and consume all of the needles and small twigs (Leopold 1924, Barney and Frischknect 1974, Koniak 1985). After severe fire, piñons and junipers can be very slow to re-establish (Erdman 1970). They do not re-sprout, so seeds must be transported by birds and mammals into the burned area and buried in suitable growing locations. The young seedlings are vulnerable to spring drought, winter freezing and thawing, and herbivory by birds and small mammals. A new piñon or juniper tree takes many decades to grow to maturity, and a stand of piñon-juniper woodland requires centuries to develop the old-growth structural characteristics (Erdman 1970, Tress and Klopatek 1987).

Spreading, low-intensity surface fires had a very limited role in molding stand structure and dynamics of many or most piñon and juniper woodlands in the historical landscape. Historical fires generally did not "thin from below," i.e., they did not kill predominantly small trees. Instead, the dominant fire effect was to kill most or all trees and to top-kill most or all shrubs within the burned area regardless of tree or shrub size. This statement also is true of most ecologically significant fires today (Romme et al. 2008, 2009).

In many piñon and juniper woodlands, stand dynamics are driven more by climatic fluctuation, insects, and disease than by fire (Eisenhart 2004, Romme et al. 2008, 2009). Although increases in piñon and juniper density have received much attention in many areas (see below), loss of piñon and juniper (especially from marginal sites) also has occurred recently and in the past. For example, a widespread and severe piñon mortality event occurred in 2002-2004 in the Four Corners region as a result of drought, high temperatures,

and bark beetle outbreaks (Breshears et al. 2005).

Historical fire rotations (i.e., the time required for the cumulative area burned to equal the size of the entire area of interest) and fire intervals at the stand level, varied from place to place in piñon and juniper woodlands but in many places were very long (generally measured in centuries). Indeed, some piñon and juniper woodlands have been stable for hundreds of years without fire, other than isolated lightning ignitions that burned only single trees or small patches and produced no significant change in stand structure. Many piñon and juniper woodlands today show no evidence of past widespread fire, although they may have burned extensively many hundreds or thousands of years ago. It appears that the longest fire rotations generally occurred in **persistent woodlands**,, and were shorter in **wooded shrublands** and **savannas**—although we have inadequate empirical data on fire histories in the latter two types of piñon and juniper vegetation (Romme et al. 2008, 2009).

Insects and Fungal Diseases

Stand structure and dynamics during the reference period also were influenced by bark beetle outbreaks and, at least in places, a black stain root disease. The bark beetles and the black stain fungus attack only piñon – insects and diseases generally have far less influence on juniper trees, which may be one reason why junipers often live longer than piñon. The two most important bark beetles in pinyon trees of the South Central Highlands Section are the pine engraver (*Ips pini*) and the piñon Ips (*Ips confusus*). Both species have a similar life cycle (Furniss and Carolin 1977). Adults bore through the

bark of piñon trees, and create "galleries" or tunnels, where they lay eggs. The larvae hatch and eat the inner bark, forming new galleries and killing the tree. Within two to several months, the larvae pupate and then emerge as adults to attack new trees. Trees that are infested with large numbers of beetles die in the same season that they are attacked, and their foliage turns brown in late summer or fall. Several generations may be produced in a single season, depending on temperatures and food supply.

The most important black stain root disease in piñon trees of southwestern Colorado is *Leptographium wageneri* var. *wageneri* (Kearns and Jacobi 2005). The fungus is transmitted from tree to tree via root contacts -- and possibly also an insect vector, but the vector has not yet been identified (James Worrall, personal communication). Because dispersal distances apparently are limited, piñon mortality from black stain root fungus tends to occur in patches. Thirty such patches were analyzed in the western portion of the San Juan National Forest (Kearns and Jacobi 2005): an average of 68% of adult piñon trees had died within the patches, which ranged from 0.07 - 0.63 ha in size and were expanding outward at an average rate of 1.1 m/year,

Part of the architectural diversity in mature and old-growth piñon-juniper forests is a result of canopy gaps formed by bark beetle outbreaks and spread of root disease. Bark beetle outbreaks commonly occur during drought conditions, when water stress reduces the host trees' ability to resist the beetle attack. Outbreaks also occur following any disturbance that creates large quantities of fresh piñon boles, e.g., windthrow or mechanical thinning operations (Furniss and Carolin 1977). In contrast, black stain root fungus produces the greatest piñon mortality

following years of well-distributed, heavy summer rain (J. Worrall, personal communication). Extensive mortality from black stain rot fungus was observed in Mesa Verde National Park following moist periods in 1932-1934 (J. Worrall, personal communication) and in the early 1980s (personal observations). In the Mesa Verde region, and perhaps elsewhere in the South Central Highlands Section, the bark beetles and root stain may interact so closely that they can be treated as a single integrated agent of piñon mortality. Individual trees infected with black stain root fungus may be more susceptible to bark beetle attack, and thus may sustain endemic bark beetle populations during moist periods when piñon trees are otherwise resistant to bark beetle attack (J. Lundquist, personal communication). These endemic beetle populations then explode into an outbreak when drought conditions again arrive.

A major mortality event occurred from 2002 - 2004 in piñon populations of southwestern Colorado, northwestern New Mexico, and adjacent parts of Arizona and Utah (Breshears et al. 2005). An outbreak of Ips beetles contributed to the mortality of piñon trees, although many trees appear to have died from direct effects of drought stress. In addition to severe drought, the past decade has been characterized by anomalously warm temperatures in the Four Corners region (and elsewhere); these high temperatures probably intensified the water stress to which the trees were subjected, and may have accelerated growth and development in the bark beetles as well (Breshears et al. 2005).

3. Legacies of Euro-American Settlement and Current Conditions

Piñon-juniper vegetation throughout its range of distribution has been heavily used, and in some places

greatly altered, during the last 120 years. The most important impacts are those related to grazing, mechanical tree removal, wood cutting, fire suppression, and exurban development (Scurlock 1998).

Grazing Impacts:

In most of the South Central Highlands Section, the first large herds of livestock were introduced in the late 1800s as Euro-American settlers moved into the region (see Chapter II for details). Because the piñon-juniper zone was readily accessible, it received relatively heavy and year-round grazing pressure by sheep and cattle throughout most of the South Central Highlands Section. Apparently almost no portion of the piñon-juniper zone escaped heavy grazing during the late 1800s and early 1900s, but grazing has been largely excluded from some areas for the last several decades, e.g., in Mesa Verde National Park (Floyd 2003). Grazing continues on most public lands and many private lands throughout the South Central Highlands Section, but ecologically appropriate and sustainable grazing practices now are being used in many areas (Knight et al. 2002, White 2008).

Where grazing continues to be heavy and chronic, substantial changes in the flora have taken place (e.g., Fleischner 1994). Even where grazing is now sustainable, there may be persistent legacies of the very heavy grazing that occurred a century ago, e.g., some palatable species may have been locally extirpated and not yet recovered. However, because we have almost no specific information about composition and structure of non-woody species during the reference period, it is impossible to determine the magnitude and locations of persistent changes induced by previous grazing. See Chapter II and the discussion of grassland ecosystems below for additional discussion of this topic.

Mechanical Tree Removal and Wood Cutting:

Extensive programs of mechanical tree reduction were carried out in piñon-juniper woodlands during the 1950s and 1960s with a primary goal of increasing forage production for livestock. A commonly used technique was "chaining" in which a large anchor chain was attached to a pair of bulldozers and pulled through a stand, knocking down the canopy trees (Paulson and Baker 2006). Chaining has been much less utilized in the last two decades, but continues in some areas. Other techniques for thinning or removing the piñon-juniper canopy also are being experimented with, including herbicide application, mowing, and "hydro-mulching."

An important consequence of chaining is fragmentation of piñon-juniper forests and woodlands (Knight et al. 2000). However, the ecological effects of this fragmentation have not received much research attention. The effects of chaining also are usually transient, because in many places the canopy becomes re-established within one to a few decades. At one site on the western side of the Unompahgre Plateau, chaining in the 1960s had removed all canopy trees, but had not removed the small piñon and juniper saplings. The saplings grew rapidly after removal of the canopy, and today the area is covered by a dense, closed canopy stand of piñon and juniper trees (personal observations). Thinning for fuel reduction and wildfire mitigation is now being conducted over extensive areas of piñon-juniper woodland in southwestern Colorado and elsewhere in the region. However, the potential impacts of this thinning on the native biota and

ecological processes such as nutrient cycling have not been examined in any detail.

Increasing Tree Densities

A striking pattern in many areas is a dramatic increase in density and canopy coverage of piñon and juniper during the last 150 years. Former grasslands and shrublands in some regions also have been converted to savanna or woodland as trees have expanded into previously non-woodland sites. However, piñon and juniper densities have not changed substantially in many other woodlands, and have even declined in others; similarly, expansion of trees into formerly non-woodland sites has been less common or nearly lacking in many regions.

Although the *pattern* of increasing tree density is well documented in many areas, the *mechanisms* driving these changes are unclear. This is an important issue, because infill and expansion often are attributed primarily to effects of fire exclusion; consequently vegetation treatments designed to reduce or eliminate piñons and junipers often are justified in part by the assumption that past and present land uses have produced "unnatural" increases in tree density. Although this assumption is probably correct in some situations, clearly it is not correct in all. For example, exclusion of low-severity surface fires during the 20th century cannot be the primary reason for infill of **persistent woodlands**, because low-severity fire was never frequent in these ecosystems even before Euro-American settlement (Romme et al. 2008, 2009). Furthermore, in many places we can explain increasing tree density as recovery from severe fire or anthropogenic clearing in the past, or as natural range expansion near the biogeographical limits of a tree species. Unfortunately, the mechanisms underlying increasing tree density in existing piñon and juniper woodlands, and expansion of piñon and juniper into grasslands and shrublands, are not well understood in most situations. Possible mechanisms include recovery from past severe disturbance, natural ongoing Holocene range expansion, livestock grazing, fire exclusion, and effects of climatic variability and rising atmospheric CO_2. In general there is much uncertainty about the ecological process(es) driving tree density increases in any particular location. The evidence for and against each of these potential mechanisms is evaluated in Romme et al. (2009).

Large Severe Fires

An upsurge of large fires (>400 ha) in forested landscapes throughout much of the western U.S began in the mid-1980s (Westerling et al. 2006). Increasing trends in large fire frequency and total area burned are particularly noticeable in some regions having extensive piñon-juniper woodlands, including the South Central Highlands Section. For example, a greater proportion of the piñon-juniper woodland on the Mesa Verde cuesta burned in the decade between 1995 and 2005 than had burned throughout the previous 200 years (Floyd et al. 2004). We know that large severe fires occurred in piñon-juniper woodlands in the past, although we have little information on spatial extent or patterns of those fires. Truncated age structures of live piñon trees and abundant charred juniper snags

document the occurrence of large fires (at least hundreds of hectares in extent) in the 1700s on the Mesa Verde cuesta. Thus, it follows that the recent occurrence of high-severity fires in piñon and juniper woodlands is not unprecedented; however, we have inadequate historical information with which to confidently evaluate how the frequency and extent of recent high-severity fires in this vegetation type compare with historical fire events.

Changes in fuel structure probably have contributed to the recent increase in large fires in some parts of the West. Invasion by highly flammable annual grasses (e.g., cheatgrass, *Bromus tectorum*) has increased horizontal fuel continuity and likelihood of extensive fire spread in many semi-arid vegetation types, including piñon-juniper woodlands and shrublands of the Great Basin and Colorado Plateau (Whisenant 1990; Knapp 1996). However, an equal or possibly more important mechanism involves the warmer temperatures, longer fire seasons, and high amplitude of wet/dry years that have occurred in recent decades (Westerling et al. 2006).

Given the very long fire rotations that naturally characterize piñon and juniper woodlands, especially **persistent woodlands**, we cannot yet determine whether the recent increase in frequency of large fires occurring in this vegetation type represents genuine directional change related to changing climate or fuel conditions, or is simply a temporary episode of increased fire activity, comparable to similar episodes in the past. In any event, the suite of current and upcoming broad-scale environmental changes--warming temperatures, increasing tree densities in some areas, and expansion of fire-promoting species such as cheatgrass— all may all interact to dramatically increase the amount of burning in piñon-juniper and other vegetation types over the next century.

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B: MOUNTAIN GRASSLANDS

1. Vegetation Structure and Composition of Mountain Grasslands

Mountain grasslands are intermixed in forest-dominated landscapes throughout the South Central Highlands Section. These herbaceous communities occur as openings in ponderosa pine, aspen, mixed-conifer, and spruce-fir forests at elevations ranging from about 2,300 – 3,500 m (7,500 - 11,600 ft). They occur on upland sites with well-drained soils in mountain and mesa landscapes. Although these grasslands are small in extent, they represent a significant component of the Section's biological diversity (Mullen 1992). They are described as Sensitive Plant Communities in the Rocky Mountain Region of the Forest Service because of their limited extent, and because they have been heavily impacted by livestock grazing and land development (Mullen 1992). Grasslands are devoid of trees and shrubs due to many factors including competition from herbaceous plants, temperatures, grazing, soil heaving, and fire (Paulsen 1975).

Within the general mountain grassland type, three more specific grassland types are dominant in the South Central Highlands Section. The **Arizona fescue type** (dominated by *Festuca arizonica*) occurs at elevations from about 2,300 – 2,700 m (7,500 – 9,000 ft), and is primarily associated with ponderosa pine and warm-dry mixed-conifer forests. The **Thurber fescue type** (dominated by *Festuca*

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thurberi) occurs at higher elevations from about 2,600 – 3,500 m (8,500 to 11,600 ft), and is primarily associated with spruce-fir, cool-moist and cold-wet mixed-conifer forests. The **Kentucky bluegrass type** (dominated by *Poa pratensis*) occurs at elevations from about 2,300 – 3,500 m (7,500 – 11,600 ft), and is associated with all forest types.

The **Arizona fescue** and **Thurber fescue types** are tall, bunchgrass communities dominated by Arizona and Thurber fescue respectively. Arizona and Thurber fescue can be described as keystone species for these types because of the pivotal role they play in these ecosystems (Noss and Cooperrider 1994. Forbs, sod-forming grasses, and sedges occupy the spaces between bunchgrasses. Sites in good ecological condition display high canopy cover of Arizona or Thurber fescue, abundant litter, minimal bare soil, and soils with thick, organic-matter-rich surface horizons. Sites where ecological conditions are not as good display lower canopy-cover of Arizona or Thurber fescue, higher cover of annuals and exotics including Kentucky bluegrass and dandelion, less litter, more bare soil, possible soil erosion and compaction problems, and higher cover of native increaser species including cinquefoil, mulesears, pussytoes, sneezeweed, strawberry, trailing fleabane, and yarrow. (See Table VI-B-1 for scientific names.)

The **Kentucky bluegrass type** is a short, sod-forming type dominated by the non-native Kentucky bluegrass and the native increaser species mentioned above. Dandelion and annual forbs are common, and noxious weeds may be present. Arizona and Thurber fescue are absent or minor components. This type commonly displays high amounts of bare soil, minimal litter, and soil erosion and compaction problems that reduce soil productivity. These grasslands can be regarded as a zootic disclimax that persists because of sustained livestock grazing (Turner and Paulsen 1976). These grasslands have undergone a process of retrogression in which the native, late-seral grass species have been replaced by exotics, forbs, and annuals (Barbour et al. 1987). The amount of litter on these sites is low while the amount of bare soil is high, which makes these grasslands highly susceptible to runoff, erosion, compaction, and weed invasions. Structural conditions reflect the open canopy associated with the great amount of bare soil and the sod-forming nature of Kentucky bluegrass, which tends to occur in a patchy pattern. The short, sparse foliage of the bluegrass and forbs provide limited litter and organic matter for nutrient cycling and soil development processes, and limited protection to the soil surface.

Some mountain grasslands in the South Central Highlands Section are dominated by other exotic species, notably crested wheatgrass, intermediate wheatgrass, orchardgrass, smooth brome, or timothy. These exotic species were purposely seeded in some grassland sites to improve forage conditions for livestock. They were also introduced throughout the region to re-vegetate roads, trails, and other disturbed sites.

Common species of the **Arizona fescue type**, in addition to Arizona fescue, include the shrubs big sagebrush and black sagebrush, and the herbs mountain muhly, Parry oatgrass, junegrass, bottlebrush squirreltail, mountain brome, blue grama, needle-

and-thread, needlegrass, pine dropseed, muttongrass, elk sedge, Kentucky bluegrass, American vetch, daisy, trailing fleabane, yarrow, dandelion, mulesears, lupine, fringed sage, cinquefoil, buckwheat, peavine, geranium, and goldenrod. Common species of the **Thurber fescue type** (Moir 1967), in addition to Thurber fescue, include the shrubs shrubby cinquefoil and snowberry, and the herbs timber oatgrass, junegrass, mountain brome, needlegrass, wild rye, sedge, Kentucky bluegrass, cinquefoil, American vetch, trailing fleabane, daisy, yarrow, sneezeweed, dandelion, harebell, meadowrue, and geranium. (See Table VI-B-1 for scientific names.)

Mountain grasslands in the region also have been described as Mountain Grassland "potential-natural-vegetation" Type (Kuchler 1964), Mountain Grassland Type (Paulsen 1975, Turner and Paulsen 1976), Mountain Grassland Ecosystem (Kingery 1998, Garrison et al. 1977), Arizona fescue, Thurber fescue, and Kentucky bluegrass series (Johnston 1987, 2001, Dick-Peddie 1993, Komarkova 1986), Arizona fescue-Mountain muhly, Arizona fescue-Danthonia parryi, and Thurber fescue-Arizona fescue plant association (Johnston 1987), and Arizona fescue and Thurber fescue types (Turner and Paulsen 1976, Currie 1975, Johnston 2001).

2. Reference Conditions in Mountain Grasslands

Descriptions such as the following give us insight and help paint a picture of what the mountain grasslands in the South Central Highlands Section looked like during the period of indigenous settlement:

-- J.T.Rothrock in 1878 (cited in Cooper 1960) wrote as he traveled through southwest Colorado, "In the beautiful valley of the Conejos River, we found luxuriant bunch-grass covering the ground as thickly as it could stand".

-- E.F.Beale in 1858 (cited in Cooper 1960) wrote as he traveled through northern Arizona, "We came to a glorious forest of lofty pines, through which we have traveled ten miles. The country was beautifully undulating, every foot being covered with the finest grass, and beautiful broad grassy vales extending in every direction."

Aside from historic anecdotal descriptions, there is little information or scientific data pertaining to the condition of mountain grasslands in the South Central Highlands Section during the reference period. Furthermore, intensive livestock grazing since the reference period has eliminated many undisturbed reference sites that could have been used to represent reference-period conditions. It is therefore difficult to accurately reconstruct the composition and structure of these plant communities prior to the arrival of domestic livestock in the late 1800s. That considered the following description is based on the best reference sites available, and extensive field reconnaissance within this Section.

During the reference period many of the mountain grasslands of the South Central Highlands Section apparently displayed high diversity and cover of herbaceous species, particularly the native bunchgrass species Arizona and Thurber fescue that displayed a

combined canopy cover greater than 50%. Forbs, sod-forming grasses, and sedges occupied the spaces between bunchgrasses. Litter amounts were high due to the large biomass associated with the robust bunchgrasses. Total vegetation cover (including litter) was about 80-90%. Bare soil occupied about 10-20% of the ground surface. Exotic species (including Kentucky bluegrass and dandelion) and noxious weeds were absent. Annual forbs and native increaser species such as yarrow, strawberry, cinquefoil, sneezeweed, and mulesears occurred to a minor extent. Soil erosion and compaction were absent or minimal. Soils had thick, organic matter-rich surface horizons formed from the consistent supply of plant material associated with the abundant and well-distributed vegetation. Structural conditions displayed a relatively closed canopy, reflecting the high density and well-distributed arrangement of the robust bunchgrasses. The tall, thick foliage of these bunchgrasses provided an abundance of litter and organic matter for energy flow, nutrient cycling, and soil development processes, and influenced hydrologic processes by protecting the soil surface from raindrop impacts, runoff, compaction, and erosion.

The function of mountain grasslands of this Section during the reference period was associated with the ecological processes of grazing by wild ungulates and fire. Intermountain grasslands of North America evolved with light grazing by wild herbivores, including elk and deer (Mack and Thompson 1982). The numbers of wild ungulates and patterns of their grazing likely varied from year to year. The animals were free to move to new areas when the abundance or palatability of desired forage species decreased. Since the plant species associated with the grasslands of this Section presumably were well adapted to the frequency, intensity, extent, and magnitude of historic grazing by wild ungulates, the composition, structure, function, and productivity of these ecosystems were maintained during the reference period.

Periodic fire was a part of the natural-disturbance regime of many North American grasslands (Sims 1988, Wright and Bailey 1982). The grasslands of the Arizona fescue type were part of the overall forest-dominated landscape associated with the fire-adapted ponderosa pine and warm-dry mixed-conifer forests (Currie 1975, Clary 1975). Since these forests were characterized by frequent, low-intensity fires throughout the reference period (Chapters II and V), we assume that frequent fires also played a significant role in maintaining the composition, structure, and function of the adjacent mountain grasslands during the reference period. The grasslands had plenty of biomass serving as fine fuel to carry a fire through them. Fire functions to reduce litter, recycle nutrients to the soil, stimulate new herbaceous growth, and restrict woody-plant establishment in grasslands. Fire probably was less important in the Thurber fescue type, since fire frequencies are much longer in the higher-elevation spruce-fir and cool-moist mixed-conifer forests with which these grasslands are associated (Chapters III and V).

3. Legacies of Euro-American Settlement and Current Conditions

Relative to the reference period, the current state of the mountain grasslands in the South Central

Highlands Section reflects impacts associated with livestock grazing, land development, and land management activities.

Heavy and unsustainable grazing occurred in grasslands throughout most of the South Central Highlands Section during the late 1800s and early 1900s (see Chapter II for details). Cattle graze extensively in mountain grasslands because these areas tend to be easily accessible, forage is typically more abundant than in forests, and many grassland plant species are highly palatable (Clary 1975, Paulsen 1975). Poorly managed or excessive livestock grazing can lead to (i) alteration of species composition, including decreases in density and biomass of individual species, and reduction of species richness; (ii) disruption of ecosystem functioning, including interference in nutrient cycling and ecological succession; (iii) alteration of ecosystem structure, including changes in vegetation stratification and community organization; (iv) a decrease in biological diversity; and (v) an increase in soil erosion (Fleischner 1994). However, it is important to recognize that livestock grazing today is being conducted in ecologically appropriate and sustainable ways in many parts of the South Central Highlands Section, where these new grazing systems are providing numerous ecological and social benefits (Knight et al. 2002, White 2008).

The excessive livestock grazing of the late 1800s and early 1900s has resulted in a legacy of significant ecological changes in many of the mountain grasslands of the South Central Highlands Section. Altered plant-community composition is the most obvious change. The dominant bunchgrasses that are so critical to the ecological integrity of these ecosystems have been extirpated in many areas, and the abundance and distribution of remaining bunchgrasses have been drastically reduced in others. Exotic species (including Kentucky bluegrass, dandelion, and noxious weeds), and native increaser species (including yarrow, cinquefoil, mulesears, sneezeweed, and annual forbs) have taken the place of bunchgrasses in many places. This change in species composition has altered the structure of many of these grasslands from tall, bunchgrass-dominated community types to short, sod-dominated or forb-dominated types. Additional livestock grazing-related changes in mountain grasslands include the reduction of litter, an increase in bare soil, increases in soil erosion and compaction resulting in a loss of soil productivity, and a decrease in native species richness. These changes in turn decreased the amount of organic matter available for nutrient cycling in the soil, changed successional pathways, and lessened the likelihood of low-intensity fires carrying through these grasslands.

Land development and recent land management activities also have introduced exotic plants and noxious weeds into the mountain grasslands of the South Central Highlands Section (see Chapter VII). Roads and trails needed for home sites, recreation, timber sales, livestock grazing, mining, pipelines, communication towers, and fire suppression have been conduits for the introduction and spread of exotic plants into the forested landscapes where the grasslands occur. Mountain grasslands that have lost native species often have much bare soil exposed, which makes them susceptible to the establishment of

exotic plants. In addition, many roads and grasslands were purposely seeded with exotic species for erosion control and to improve forage conditions for livestock (e.g., smooth brome, intermediate wheatgrass, timothy, orchardgrass, Kentucky bluegrass, and yellow sweet clover). In many areas these non-native species still dominate the plant community.

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Table VI-B-1. Plant species mentioned in the mountain grasslands section (alphabetical order by common name)

SHRUBS - big sagebrush (*Seriphidium tridentatum ssp tridentatum*), black sagebrush (*Seriphidium novum*), shrubby cinquefoil (*Pentaphylloides floribunda*), and snowberry (*Symphoricarpos rotundifolius*).

FORBS - American vetch (*Vicia americana*), beardstongue (*Penstemon barbatus*), buckwheat (*Eriogonum umbellatum*), butterweed (*Packera neomexicana*), cinquefoil (*Potentilla hippiana*), daisy (*Erigeron formosissimus*), dandelion (*Taraxacum officinale*), deervetch (*Lotus wrightii*), fringed sage (*Artemisia frigida*), geranium (*Geranium caespitosum*), goldenrod (*Solidago simplex*), hairy goldenaster (*Heterotheca villosa*), harebell *(Campanula rotundifolia*), lupine (*Lupinus argenteus*), meadowrue (*Thalictrum fendleri*), mountain parsley (*Pseudocymopterus montanus*), mulesears (*Wyethia arizonica*), owlclover (*Orthocarpus luteus*), paintbrush (*Castilleja integra*), peavine (*Lathyrus leucanthus*), pussytoes (*Antennaria rosea*), sneezeweed (*Dugaldia hoopesii*), strawberry (*Fragaria vesca* and *virginiana*), trailing fleabane (*Erigeron flagellaris*), and yarrow (*Achillea lanulosa*).

GRAMINOIDS - Arizona fescue (*Festuca arizonica*), blue grama (*Chondrosum gracile*), bottlebrush squirreltail (*Elymus elymoides*), crested wheatgrass (*Agropyron cristatum*), elk sedge (*Carex geyeri*), Indian ricegrass (*Achnatherum hymenoides*), intermediate wheatgrass (*Thinopyrum intermedium*), junegrass (*Koeleria macrantha*), Kentucky bluegrass (*Poa pratensis*), mountain muhly (*Muhlenbergia montana*), mountain brome (*Bromopsis pumpelliana),* muttongrass (*Poa fendleriana*), needle-and-thread (*Hesperostipa comata*), needlegrass (*Achnatherum nelsonii),* orchardgrass (*Dactylis glomerata*), Parry oatgrass (*Danthonia parryi*), ryegrass (*Elymus glauca*), sedge (*Carex*), smooth brome (*Bromopsis inermis*), Thurber fescue (*Festuca thurberi*), timber oatgrass (*Danthonia intermedia),* timothy (*Phleum pratense*), western wheatgrass (*Pascopyrum smithii*), and wild rye (*Elymus canadensis*).

C: RIPARIAN VEGETATION AND WETLANDS

Riparian areas and wetlands occur at all elevations throughout the South Central Highlands Section. They occur on valley floors and in other low-lying landscape positions, and are primarily associated with perennial streams. Although limited in extent, riparian and wetland ecosystems contribute a great diversity of native vegetation communities and bird, fish, and vertebrate species to southwestern landscapes (Howe and Knopf 1991, Durkin et al. 1995). Riparian areas and wetlands store water, stabilize valley floors, enhance water quality, provide recreation and esthetic values, and provide wildlife species with habitat, water, food, cover, and travel routes.

Riparian areas and wetlands are defined here together, as the interface between the riverine aquatic ecosystem and the adjacent upland ecosystem, where the water table is usually at or near the land surface (Gregory et al. 1991, Brinson et al. 1981). They are frequently flooded, or at least seasonally saturated by a fluctuating water table, and have plant species, soils, and topography that differ considerably from those of the adjacent uplands. Riparian areas and wetlands support a prevalence of hydrophytic vegetation adapted for life in saturated soil conditions (Corps of Engineers 1987).

1. Riparian Vegetation Structure and Composition

The structure and composition of riparian and wetland vegetation is typically complex and varies at fine spatial scales, especially in rugged mountainous regions like the Couth Central Highlands Section.

Consequently, there is no single, universally acceptable classification system for these ecosystems. Baker (1989) sampled 115 stands distributed along most of the major rivers in western Colorado, representing an elevational gradient of 1,700 m and a latitudinal gradient of 300 km. Four major vegetation types were identified, each of which could be subdivided into numerous subtypes as influenced primarily by interacting elements of valley morphology, elevation, and flood history. The four types were montane riparian forests (below about 2620 m) dominated by *Populus angustifolia* and *Picea pungens*; subalpine riparian forests (above about 2620 m) dominated by and *Abies lasiocarpa Picea engelmannii*; lower subalpine carrs (generally at 2620-3110 m elevation) dominated by tall willows including *Salix drummondiana*, *S. geyeri*, and *S. monticola*; and upper subalpine carrs (above 3110 m) dominated by short willows including *Salix brachycarpa*, *S. planifolia*, and *S. wolfii*.

Focusing just on montane riparian forests along the Animas River in the San Juan Mountains of southwestern Colorado, Walford and Baker (1995) developed a finer-scale classification of eight riparian vegetation types dominated by various combinations of *Populus angustifolia*, *P. tremuloides*, *Picea pungens*, *Pseudotsuga menziesii*, and various shrubs and herbs. A far more detailed classification of riparian and wetland vegetation types in the South Central Highlands Section is described in Table VI-C-1 within a hierarchy at the Subclass, Alliance, and Community Type levels (Grossman et al. 1998,

Muldavin et al. 2000, Redders 2000). See also Kittel et al. (1999) and Johnston (2001).

2. Reference Conditions in Riparian Vegetation

Specific information about riparian areas and wetlands during the reference period in the South Central Highlands Section is limited. Reference sites at low to middle elevations that could have been used to represent reference-period conditions are especially rare due to extensive human impacts since EuroAmerican settlement (Baker 1989). However, we can infer some of the important features of riparian and wetland vegetation during the reference period based on the ecological processes that probably have not been greatly altered.

It is important to recognize that riparian areas, especially those situated in rugged mountainous landscapes, are dynamic ecosystems shaped both by relatively stable features of the landscape, including valley morphology and elevation, and by contingent factors such as periodic flooding and resultant changes in channel morphology and substrate characteristics (Baker 1989). Major floods, as occurred along the Animas River in southwestern Colorado in 1911, 1927, 1941, and 1957, periodically wash out riparian vegetation, deposit new sediments, and initiate new successional processes (Baker 1990, Baker and Walford 1995). Beavers also play a large role in riparian and wetland dynamics by creating ponds, which eventually washed out leaving moist sediments ideal for establishment of willows and water-loving herbs (Wohl 2001). Thus, riparian vegetation along the rivers and streams of the South Central Highlands Section is best viewed as a continually shifting, fine-scale mosaic of patches, reflecting the dynamic interplay between stable landscape features (valley morphology and elevation) and periodic disturbance (floods and beavers).

3. Legacies of Euro-American Settlement and Current Conditions

Riparian areas and wetlands have changed dramatically during the last century and a half in the southwestern United States due to human impacts. In New Mexico, riparian vegetation has probably been impacted more by human activities than any other type of vegetation (Dick-Peddie 1993). Human impacts that have occurred in the South Central Highlands Section include urbanization, agriculture, logging, livestock grazing, mining, recreation, roads, dams, diversions, and the introduction of exotic species.

Riparian areas and wetlands in the San Juan Mountains are among the lands that were first settled and developed for townsites, agriculture, and roads. Riparian trees were cut for fuel and shelter, and to clear land for agriculture and settlement (Dahms and Geils 1997). Settlement and agriculture occurred along riparian areas near towns throughout the South Central Highlands Section.

The construction of dams, reservoirs, and diversions not only cleared the vegetation and modified the topography, but also decreased and regulated water flow, blocked movements of aquatic organisms, and changed the natural geomorphic stream processes of channel formation and erosion/deposition. The associated drop in water tables and lack of flooding have

resulted in significant changes in the abundance, distribution, and reproductive mechanisms (germination and seedling survival) of native riparian area/wetland plant species, particularly willows and cottonwoods (Glinski 1977, Brady et al. 1985). Regulated stream flow is thought to be the most important factor contributing to the decline of cottonwood and willow riparian ecosystems in the West (Brown et al. 1977, Fenner et al. 1985, Rood and Heinze-Milne 1989, Rood and Mahoney 1990, 1993). Numerous dams and reservoirs have been built throughout the South Central Highlands Section and include, e.g., Big Meadows, Electra, Lemon, McPhee, Platoro, Rio Grande, Santa Marie, Vallecito, and Williams.

Heavy and unsustainable grazing occurred in grasslands throughout most of the South Central Highlands Section during the late 1800s and early 1900s (see Chapter II for details). The effects of livestock grazing on riparian areas and wetlands are well documented in the literature (Kauffman and Krueger 1984, Platts and Raleigh 1984, Skovlin 1984, Clary and Webster 1989, Clary and Medin 1990, Schulz and Leininger 1990, Fleischner 1994). The reduction or elimination of woody riparian species by livestock is particularly detrimental to riparian areas and wetlands that are dependant on those species to stabilize banks and hold those systems together. Even when environmental conditions are conducive to cottonwood and willow regeneration, stands of cottonwood and willow can be destroyed by excessive grazing by livestock (Glinski 1977, Carothers 1977, Kauffman et al. 1983). However, it is important to recognize that livestock grazing today is being conducted in ecologically appropriate and sustainable ways in many parts of the South Central Highlands Section, where these new grazing systems are providing numerous ecological and social benefits (Knight et al. 2002, White 2008).

High-elevation mining without adequate handling of mine waste has caused severe degradation of many streams in the South Central Highlands Section, notably in portions of the western San Juan Mountains (Somers and Floyd-Hanna 1996, Paulson and Baker 2006). Logging also can affect riparian areas and wetlands by removing vegetation, which decreases the interception and infiltration of water, and increases runoff. With increased surface runoff, erosion increases and more sediment is dumped into stream systems. Roads, built and maintained for logging activities, are the dominant source of soil erosion and stream sediment in forest environments (Swank and Crossley 1988). Sediment fills gravel and cobble beds that are sites of attachment for insect eggs, larvae, macroinvertebrates, and fish eggs (Noss and Cooperrider 1994). This greatly diminishes critical fish habitat.

The reduction or elimination of native riparian and wetland plant species due to human impacts described above, has allowed exotic species such as Russian olive and salt cedar (tamarisk) to become established and highly competitive in these ecosystems. The diversity of Southwestern riparian ecosystems has been greatly reduced by these two plants (Dick-Peddie 1993). Salt cedar and Russian olive apparently outcompete native cottonwoods and willows, limiting the regeneration success of native woody plants (Finch et al. 1995).

We lack details about changes since the reference period in most

specific areas of the South Central Highlands Section. Nevertheless, we offer the following brief descriptions based on the best reference sites available and field reconnaissance within the South Central Highlands Section.

Evergreen Forests - These forests have probably been the least affected by human impacts due to their more remote locations and the limited access to them. During the reference period they looked much like they do today, with subalpine fir, Engelmann spruce, and blue spruce displaying a relatively open and patchy canopy cover along the stream channel and throughout the valley floor.

Deciduous Forests and Mixed Evergreen-Deciduous Forests - These forests have probably been the most affected by human impacts due to their easy access and their desirability for settlement, agriculture, livestock grazing, and roads (as described below). During the reference period, cottonwood trees and willows were abundant and well distributed along most perennial and intermittent streams, much more so than what is displayed in most places today. Palatable herbs, particularly graminoids, were also more abundant and well distributed than what is displayed in most places today. [An exception to this general pattern is seen in photographs of the Animas River where the town of Durango now lies. The photos show almost no trees along the river. We do not know if this is just a curious anomaly, perhaps reflecting local cutting or burning just before the photos were taken, or if there is more to the story of riparian forests during the reference period than we now recognize. Again, we emphasize the paucity of specific historical information about these ecosystems.]

Deciduous Shrublands - These shrublands have been significantly affected by human impacts due to their easy access and their desirability for settlement, agriculture, livestock grazing, and roads. During the reference period willows were abundant and well distributed along perennial and intermittent streams, much more so than what is displayed in most places today. Palatable herbs, particularly graminoids, were also more abundant and well distributed than what is displayed in most places today. The extensive removal of beavers in the 19th century also eliminated one of the key processes that formerly created moist sediments conducive to willow seedling establishment (Wohl 2001).

Perennial Forbs and Perennial Graminoids - During the reference period these wetlands looked much like they do today, but total acreage has been reduced due to human impacts associated with their desirability for settlement, agriculture, livestock grazing, and roads.

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Table VI-C-1. Riparian and wetland vegetation of the South Central Highlands Section (based on Grossman et al. 1998, Muldavin et al. 2000, Redders 2000). Community types are plant communities with definite floristic composition and uniform physiognomy, and named by species that dominate the uppermost canopy layer and a diagnostic species that occurs in the underlying strata (Grossman et al. 1998). Alliances are aggregations of community types named by species that dominate the uppermost canopy layer. Subclasses are aggregations of alliances, and based on the predominant leaf phenology of the life form in the upper canopy layer. Scientific names are listed at the end of this document. Alliances marked with an asterisk (*) have been described by the Forest Service as Sensitive Plant Communities that may need some type of special management or protection within the South Central Highlands Section (Mullen 1992). See Table VI-C-2 for a list of plant species names used in this table.

Subclass: Evergreen Forests
   Alliances: Subalpine Fir-Engelmann Spruce and Blue Spruce*.
     Community Types: Subalpine Fir-Engelmann Spruce/Alder,
             Subalpine Fir-Engelmann Spruce/ Drummond Willow,
             Subalpine Fir-Engelmann Spruce/Currant,
             Subalpine Fir-Engelmann Spruce/ Honeysuckle,

Subalpine Fir-Engelmann Spruce/Bluejoint Reedgrass,
Subalpine Fir-Engelmann Spruce/Mesic Forb,
Subalpine Fir-Engelmann Spruce/Mountain Bluebells,
Blue Spruce/Alder, and
Blue Spruce/Red-osier Dogwood.

This subclass is dominated by evergreen trees that occur throughout the South Central Highlands Section at elevations ranging from about 2,400 – 3,000 m (8,000 – 10,000 ft). Subalpine fir, Engelmann spruce, and blue spruce trees occur along the stream channel and throughout the valley floor, and commonly display a relatively open and often patchy canopy cover. Alder and Drummond willow are found along the stream channel and usually display high cover in the community types named after them. Honeysuckle, red-osier dogwood, and gooseberries occur throughout the valley floor (including along the channel) and display low to high cover in the community types named after them. Bluejoint reedgrass, mountain bluebells, and mesic forbs tend to be found in herb-dominated patches between trees in the community types named after them.

Common shrubs and herbs associated with the Evergreen Forest type include alder, Drummond willow, honeysuckle, red-osier dogwood, currents, bittercress, cow parsnip, bluebells, Fendler cowbane, brook saxifrage, twisted stalk, horsetail, arrowleaf groundsel, goldenglow, geranium, and bluejoint reedgrass.

Subclass: Deciduous Forests
  Alliances: Narrowleaf Cottonwood*,
        Box Elder,
        Box Elder-Narrowleaf Cottonwood*, and
        Rio Grande Cottonwood*.
    Community Types: Narrowleaf Cottonwood/Alder,
                Narrowleaf Cottonwood/Red-osier Dogwood,
                Narrowleaf Cottonwood/Hawthorn,
                Narrowleaf Cottonwood/Serviceberry,
                Narrowleaf Cottonwood/Skunkbrush,
                Narrowleaf Cottonwood/Willow,
                Narrowleaf Cottonwood/Coyote Willow, and
                Box Elder/Red-osier Dogwood.

This subclass is dominated by broad-leaved deciduous trees that occur at elevations ranging from about 2,000 – 2,900 m (6,500 – 9,500 ft), with the Narrowleaf Cottonwood and Box Elder alliances occurring throughout the South Central Highlands Section at elevations from about 2,000 – 2,900 m (6,500 – 9,500 ft and the Rio Grande Cottonwood alliance occurring mostly in New Mexico at elevations below about 2,100 m (7,000 ft). Cottonwood and box elder trees occur along the stream channel and throughout the valley floor, and commonly display a relatively open and often patchy canopy cover. Alder and willows are found along the stream channel and usually display high cover in the community types named after them. Red-osier dogwood, serviceberry,

skunkbrush, and hawthorn occur throughout the valley floor (including along the channel) and display low to high cover in the community types named after them.

Common shrubs and herbs associated with the Deciduous Forest type include alder, honeysuckle, red-osier dogwood, maple, rose, serviceberry, gooseberry, skunkbrush, hawthorn, coyote willow, Bebb willow, strapleaf willow, mountain willow, whiplash willow, Drummond willow, Geyer willow, false Solomon seal, bittercress, cow parsnip, bluebells, Fendler cowbane, brook saxifrage, horsetail, goldenglow, geranium, and Kentucky bluegrass.

Subclass: Mixed Evergreen-Deciduous Forests
   Alliances: Narrowleaf Cottonwood-Blue Spruce*,
       Subalpine Fir-Engelmann Spruce-Narrowleaf Cottonwood*, and
       Narrowleaf Cottonwood-Douglas-Fir.
    Community Types: Narrowleaf Cottonwood-Blue Spruce/Alder,
         Narrowleaf Cottonwood-Blue Spruce/Mixed Shrub, and
         Narrowleaf Cottonwood-Blue Spruce/Willow.

This subclass is dominated by evergreen and broad-leaved deciduous trees that occur throughout the South Central Highlands Section at elevations ranging from about 2,300 – 2,900 m (7,500 – 9,500 ft). Narrowleaf cottonwood, subalpine fir, Engelmann spruce, and blue spruce trees occur along the stream channel and throughout the valley floor, and commonly display a relatively open and often patchy canopy cover. Douglas-fir usually occurs in the valley floor away from the channel. Alder and willows are found along the channel and usually display high cover in the community types named after them. Honeysuckle, red-osier dogwood, serviceberry, snowberry, and maple occur throughout the valley floor (including along the channel) and display low to high cover in the Mixed Shrub community type. Common shrubs and herbs associated with the Mixed Evergreen-Deciduous Forest type are described above for the Deciduous and Evergreen Forests.

Subclass: Deciduous Shrublands
   Alliances: Bebb Willow*,
       Drummond Willow*,
       Geyer Willow*,
       Strapleaf Willow*,
       Mountain Willow*,
       Coyote Willow*,
       Shortfruit Willow*,
       Planeleaf Willow*,
       Wolf Willow*,
       Bluestem Willow*,
       Alder,
       Drummond Willow-Alder,
       Red osier Dogwood,
       Hawthorn, Silverberry,
       Shrubby Cinquefoil,

Bog Birch,
River Birch, and
Saltcedar.
Community Types: Alder/Horsetail,
Alder/Mesic Forb,
River Birch/Mesic Graminoid,
Shrubby Cinquefoil/Tufted Hairgrass,
Drummond Willow/Mesic Forb,
Mountain Willow/Mesic Forb,
CoyoteWillow/Mesic Graminoid,
Wolf Willow/Mesic Forb,
Coyote willow/Bare Ground,
Geyer willow/ Mesic Forb,
Shortfruit Willow/Mesic Forb,
Planeleaf Willow/Marsh Marigold,
Planeleaf Willow/ Water Sedge,
Wolf Willow/Water Sedge, and
Wolf Willow/Beaked Sedge.

This subclass is dominated by tall deciduous shrubs that occur along the channel and throughout the valley floor, and display low to high cover. Alder, river birch, saltcedar, and the willows are found along the stream channel and usually display high cover in the alliances named after them. The other shrubs and Bebb willow occur throughout the valley floor (including along the channel) and display low to high cover in the alliances named after them. The Bog Birch, Planeleaf Willow, Wolf Willow, Shortfruit Willow, and Geyer Willow alliances are wetland types that occur at elevations above about 2,900 m (9,500 ft). Drummond Willow, Mountain Willow, and Alder alliances occur at mid to high elevations above about 2,400 m (8000 ft). The Bebb, Strapleaf, and Coyote Willow alliances, and Red osier Dogwood, Hawthorn, Saltcedar, and River Birch alliances occur at elevations from about 2,100 – 2,700 m (7,000 - 9,000 ft). Graminoids and Forbs tend to be found in herb-dominated patches between shrubs in the community types named after them.

Common herbs associated with the Deciduous Shrubland type include marsh marigold, goldenglow, geranium, cow parsnip, bluebells, bedstraw, bittercress, Fendler cowbane, arrowleaf groundsel, brook saxifrage, pink stonecrop, strawberry, osha, elephant head, king's crown, water sedge, beaked sedge, tufted hairgrass, mannagrass, Kentucky bluegrass, bluejoint reedgrass, and horsetail.

Subclass: Perennial Forbs
  Alliances: Marsh Marigold and Bittercress-Bluebells

This subclass is dominated by forbs that occur at elevations above about 2,700 m (9,000 ft). Common herbs associated with this type include marsh marigold, bluebells, bittercress, Fendler cowbane, arrowleaf groundsel, brook saxifrage, pink stonecrop, tall larkspur, elephant head, star gentian, willowherb, water sedge, and tufted hairgrass.

Subclass: Perennial Graminoids
   Alliances: Water Sedge*,
        Beaked Sedge*,
        Water Sedge-Beaked Sedge*,
        Smallwing Sedge,
        Mountain Sedge-Marsh Marigold,
        Tufted Hairgrass*,
        Reed Canarygrass,
        Threesquare Bulrush,
        Softstem Bulrush,
        Broadleaf Cattail*,
        Common Spikerush*, and
        Baltic Rush.

This subclass is dominated by grasses, sedges, and rushes that occur at all elevations of the South Central Highlands Section. Common herbs associated with this type include those named in the alliances above and wooly sedge, elephant head, brook saxifrage, willowherb, arrowleaf groundsel, and bluejoint reedgrass.

Table VI-C-2. Plant names used in Table VI-C-2, classification of riparian plant communities (alphabetical order by common name).

TREES - blue spruce (*Picea pungens*), box elder (*Acer negundo*), Douglas-fir (*Pseudotsuga menziesii*), engelmann spruce (*Picea engelmannii*), narrowleaf cottonwood (*Populus angustifolia*), Rio Grande cottonwood (*Populus deltoides* ssp. *wislizeni*), and subalpine fir/corkbark fir (*Abies lasiocarpa*) – *Abies bifolia/Abies arizonica* (Weber). Subalpine fir and corkbark fir are grouped together and referred to as subalpine fir in this document, due to their ecological similarities and the difficulty of differentiating them in the field. They both grow in the South Central Highlands Section and probably interbreed as well.

SHRUBS – Alder (*Alnus incana* ssp. *tenufolia*), Bebb willow (*Salix bebbiana*), bluestem willow (*Salix irrorata*), bog birch (*Betula glandulosa*), coyote willow (*Salix exigua*), Drummond willow (*Salix drummondiana*), Geyer willow (*Salix geyeriana*), gooseberry (*Ribes montigenum*), hawthorn (*Crataegus rivularis*), honeysuckle (*Distegia involucrata*), maple (*Acer glabrum*), mountain willow (*Salix monticola*), planeleaf willow (*Salix planifolia*), red osier dogwood (*Cornus sericea*), river birch (*Betula fontinalis*), rose (*Rosa woodsii*), Russian olive (*Elaeagnus angustifolia*), saltcedar (*Tamarix ramosissima*), serviceberry (*Amelanchier alnifolia*), shortfruit willow (*Salix brachycarpa*), shrubby cinquefoil (*Pentaphylloides floribunda*), silverberry (*Shepherdia argentea*), skunkbush (*Rhus trilobata*), snowberry (*Symphoricarpos rotundifolius*), strapleaf willow (*Salix eriocephala* var. *ligulifolia*), and wolf willow (*Salix wolfii*).

FORBS – arrowleaf groundsel (*Senecio triangularis*), bedstraw (*Galium triflorum*), bittercress (*Cardamine cordifolia*), bluebells (*Mertensia ciliata* and *fransiscana*), broadleaf cattail (*Typha latifolia*), brook saxifrage (*Micranthes odontoloma*), cow parsnip (*Heracleum lanatum*), elephant head (*Pedicularis groenlandica*), false Solomon seal (*Maianthemum racemosum and stellatum*), Fendler cowbane (*Oxypolis fendleri*), geranium (*Geranium richardsonii*), goldenglow (*Rudbeckia ampla*), king's crown (*Rhodiola integrifolia*), marsh marigold (*Caltha leptosepala*), osha (*Ligusticum porteri*), pink stonecrop (*Sedum rhodanthum*), star gentian (*Swertia perennis*), strawberry (*Fragaria virginiana*), tall larkspur (*Delphinium barbeyi*), twisted stalk (*Streptopus amplexifolius*), and willowherb (*Epilobium hornmannii*).

GRAMINOIDS - baltic rush (*Juncus balticus*), beaked sedge (*Carex utriculata*), bluejoint reedgrass (*Calamagrostis canadensis*), common spikerush (*Eleocharis palustris*), horsetail (*Equisetum arvense* and *pratense*), Kentucky bluegrass (*Poa pratensis*), mannagrass (*Glyceria striata*), mountain sedge (*Carex scopulorum*), reed canarygrass (*Phalaris arundinacea*), smallwing sedge (*Carex microptera*), softstem bulrush (*Scirpus tabernaemontani*), threesquare bulrush (*Scirpus pungens*), tufted hairgrass (*Deschampsia cespitosa*), water sedge (*Carex aquatilis*), and wooly sedge (*Carex lanuginosa*).

D: ALPINE ECOSYSTEMS

The alpine zone in the South Central Highlands Section occurs above timberline at elevations above about 3,500 m (11,500 ft).   Mean annual precipitation ranges from about 75 – 125 cm (30 - 50 in). The alpine environment is characterized by short cool growing seasons, long cold winters, high wind, and intense light.   Shaped by geologic uplift, erosion, frost action, and glaciation, the alpine landscape is steep and rugged in many places but gentle and smooth in others. Rock outcrop and talus slopes are common. Soils are shallow and rocky on steep slopes and exposed ridges, and deeper and more productive on other sites.

Much of the alpine zone in the western U.S. occurs on federal land, and much of it is in Wilderness Areas. Alpine lands are important watersheds for agricultural and metropolitan uses, and provide year-round snow accumulation and water storage for summer runoff of streams. They provide summer range for livestock, primarily sheep, and many wildlife species. They are lands associated with mineral resources and mining. These "lands above the trees" also display stunning panaramic vistas and a remote solitude that attracts recreationalists.   Much of the alpine zone is unroaded and mostly undisturbed by human activities.

Wildlife species of note in the alpine include brown-capped rosy finch (*Leucosticte australis*) which nests exclusively in the alpine in deep crevices in solid rock or in cracks on looser rock surfaces, and the white-tailed ptarmigan (*Lagopus leucurus*) which is one of the only year-around residents of the alpine, spending the winter foraging in alpine tall willow communities (Kingery 1998).

Elk, lynx, marmot, pika, pocket gophers, and bighorn sheep are other common alpine occupants. The endangered Uncompahgre Fritillary butterfly (*Boloria acrochema*) also makes the alpine its home, utilizing the dwarf-willow vegetation type (Povilitis 2000).

1. <u>Alpine Vegetation Structure and Composition</u>

Alpine plants are uniquely adapted to their environment. The major adaptations include reduction in growth height, perennial life cycle, herbaceous habit, and belowground biomass (Thilenius 1975). High photosynthetic efficiency, resistance to drought, fleshy leaves, evergreen leaves, thick cuticles, epidermal hairs, and dark scales are other important adaptations. Alpine vascular plants grow slowly, an adaptation to low temperatures, high desiccation rates, and sudden microclimatic changes.

There is tremendous diversity of species and vegetation communities in the alpine zone, including mosses and lichens, which constitute a significant contribution to the total flora (Zwinger and Willard 1972, Hartman and Rottman 1985, Jamieson et al. 1996, Johnston 2001). Community types change abruptly over short distances, due to small-scale topographic changes that exert a significant influence on snow and moisture conditions and the associated vegetation.

Within the alpine zone of the South Central Highlands Section are four general and widespread vegetation types: fellfield, turf, wetland, and dwarf willow (Baker 1983, Thilenius 1975, Dick-Peddie 1993).

The <u>fellfield type</u> occurs on harsh, wind-swept sites with shallow, rocky soils. It is dominated by short cushion plants (forbs) and displays a relatively low canopy cover. Common species of this type include *Eremogone fendleri, Lidia obtusiloba, Myosotis asiatica, Paronychia pulvinata, Phlox caespitosa, Silene acaulis*, *Trifolium dasyphyllum,* and *Trifolium nanum.*

The <u>turf type</u> occurs on protected sites away from excessive wind, and tends to have deeper, moister, and more well-developed soils. It is dominated by forbs and graminoids, and usually displays a relatively high canopy cover. Within the general turf type, a number of community types can be recognized including *Carex elynoides, Festuca thurberi Geum rossii,* and *Kobresia myosuroides*. A combination turf/fellfield community (*Carex rupestris*/cushion plants) is also recognized (Baker 1983, Dick-Peddie 1993).

The <u>wetland type</u> primarily occurs on poorly-drained, low-lying sites where water accumulates. It tends to display high cover and includes the following community types: *Deschampsia cespitosa, Juncus drummondii*, *Potentilla fruiticosa,* herbaceous wetland, and tall willow. The tall willow type is composed of *Salix planifolia* and *S. glauca*, and occurs along riparian corridors that are often connected to lower elevation subalpine streams. The herbaceous wetland type is variable and displays a great diversity of species including *Caltha leptosepala*, *Carex* spp., *Clementsia rhodantha*, *Pedicularis groenlandica*, *Primula parryi*, and *Rhodiola integrifolia*.

The <u>dwarf willow type</u> is composed of *Salix arctica* and *S. reticulata,* and commonly occurs on sites

with a heavy snowpack that extends into the summer.

Krummholz, composed of dwarfed conifers (mostly *Picea engelmannii*) and alpine herbaceous species, is a transition type that occurs between spruce-fir forests of the subalpine climate zone and true alpine. A rockland or talus type is also recognized and dominated by *Senecio atratus*, *Ligularia soldanella*, and lichens (Dick-Peddie 1993

2. Reference Conditions in Alpine Vegetation

Specific information about the structure and function of alpine ecosystems during the reference period is almost nonexistant. Since the mid to late 1800s, most alpine ranges in the South Central Highlands Section have been used almost continuously for summer grazing, making it extremely difficult to reconstruct historical condition (Thilenius 1975). Because specific historical information is lacking, we cannot confidently describe vegetation composition and structure of alpine ecosystems during the reference period. We speculate that general vegetation structure and distribution resembled what we see today, but that species composition was different in places that have received heavy sheep grazing.

3. Legacies of Euro-American Settlement and Current Conditions in Alpine Ecosystems

Alpine ecosystems in the South Central Highlands Section today reflect impacts associated primarily with livestock grazing, mining, and recreation.

Since the late 1900s, sheep have been the principal domestic livestock using alpine lands. The early sheepmen moved their animals into the alpine zone for summer grazing where the sheep were grazed in tightly grouped bands and continuously bedded in the same location for several nights in a row. These practices resulted in large losses of forage through trampling and in soil damage from excessive trailing to and from the bedding ground to water (Thilenius 1975). Historic sheep driveways in the alpine also exhibit local range deterioration, including a lack forage plants and an increase in erosion and runoff. Management of sheep in the alpine zone has been better regulated andd more ecologically sound since the mid-1930s, and range conditions have improved greatly in many places, but legacies of uncontrolled grazing that occurred before that time are still with us.

Mining has been a widespread activity in alpine ecosystems of the South Central Highlands Section since the mid 1800s (Smith 1996). Old supply roads, deserted structures, settling ponds, and mine tailings are common mining features in many areas. When many of these mines were abandoned, steps to limit pollution from them were stopped or never initiated, causing major pollution problems (e.g., in the upper Animas River of the San Juan Mountains). Abandoned mine tailings often contain soils that are highly acidic and laden with toxic concentrations of heavy metals. Erosion, acid-water runoff, and sedimentation from mines often result in adverse impacts to vegetation, streams, and aquatic ecosystems (Somers and Floyd-Hanna 1996, Paulson and Baker 2006). Recreational use of high mountain areas,

including the alpine, has increased dramatically in the past half century; recreational impacts are discussed in Chapter VII.

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E: MOUNTAIN SHRUBLANDS

Interspersed among piñon-juniper and ponderosa pine forests and woodlands are extensive tracts of mountain shrubland or Petran chaparral (Erdman 1970, Keeley and Keeley 1988, Spence et al. 1995, Floyd et al. 2000, Floyd 2003). The shrublands are dominated by species such as mountain-mahogany (*Cercocarpus montanus*), Utah serviceberry (*Amelanchier utahensis*), and fendlerbush

(*Fendlera rupicola*) on drier sites and at lower elevations, and by species such as Gambel oak (*Quercus gambelii*), and snowberry (*Symphoricarpos* spp) on wetter sites and at higher elevations. Nearly all of the dominant species have extensive root systems and the capacity to re-sprout after disturbance. Old oak-dominated stands (undisturbed for 100+ years) may contain relatively large individuals up to 10 cm (4 in) in diameter and 5 m (16 ft) tall, but in most stands the stems are < 5 cm (2 in) diameter and < 2 m (7 ft) tall. These shrublands are remarkably diverse in many places; for example, Spencer and Romme (1996) documented 32 species of shrubs and subshrubs on the slopes around Fort Lewis College in Durango, CO. Mountain shrublands cover a large area in the South Central Highlands Section. They are especially prominent along the southern and western slopes of the San Juan Mountains, on the Mesa Verde cuesta, and on the Uncompahgre Plateau. However, they have received little research attention, and we know little about their historical composition and dynamics.

The mechanisms explaining why mountain shrublands occur in environments that appear suitable for forest or woodland are poorly understood (Spencer and Romme 1996). Overall floristic composition of the shrublands generally is very similar to composition of adjacent forests or woodlands – except that the tree component is absent or very sparse. The species of shrubs and herbs found in the shrublands also can be found in nearby forests and woodlands – except that the shrub species are less abundant in the latter locations. This floristic similarity, coupled with the fact that the shrublands are not strongly associated with any particular elevational or topographical setting (i.e., they can be found across a very wide range of elevations and topographic conditions), suggests that the mountain shrublands are primarily a result of disturbance. The disturbance may be low-intensity but chronic, or high-intensity and acute. For example, steep slopes underlain by unstable Mancos shale at elevations where deep snow is frequent, are usually covered with mountain shrubland -- perhaps because the sprouting shrubs with short-lived stems can tolerate the chronic, low-intensity disturbance generated by soil erosion and snow-creep, whereas long-lived trees that reproduce only by seed cannot tolerate this chronic disturbance. In contrast, stable substrates of Dakota sandstone may support either ponderosa pine forest or oak shrubland in the vicinity of Durango, Colorado, with no obvious site differences between the forest and shrubland. It is possible that a intense disturbance at some undocumented time in the past killed the ponderosa pine forest locally (e.g., a severe fire followed shortly by another severe fire that killed the young trees re-establishing after the first fire, or a severe mountain pine beetle outbreak followed by severe fire), and the trees were subsequently unable to become re-established either because of inadequate seed sources or competition from the well-established shrubs that re-sprouted after the disturbance. We stress, however, that this is speculation, and that the exact reasons for the occurrence of shrubland in many places throughout the South Central Highlands are simply not understood.

One place where we have some specific information on the history and long-term dynamics of mountain shrublands is in Mesa Verde National Park. Shrublands of Gambel oak and Utah serviceberry dominate the upper

portions of the Mesa Verde cuesta (above about 1700 m), while the lower portions of the cuesta are mostly piñon-juniper forest. Floyd et al. (2000) determined that the fire turnover time in the shrublands (the time required to burn a cumulative area equal to the total extent of shrubland) was about 100 years, whereas the turnover time for the piñon-juniper forest was about 400 years. Piñon-juniper forest requires 300+ years to recover after fire (Erdman 1970), whereas burned shrublands recover within a decade or two (Floyd et al. 2000). Thus, it appears that the shrublands on Mesa Verde are maintained in large part by periodic fire. If fire were excluded for 300+ years, then the piñon-juniper forest probably could expand into the areas now dominated by shrubland, since scattered piñon and juniper trees do grow in this area today. However, such a long period without fire is highly unlikely. Indeed, Floyd et al. (2000) found that the cumulative area burned in Mesa Verde during the second half of the 20th century (when the policy was total fire suppression) was about the same as the cumulative area burned in the second half of the 19th century (when no fire suppression was attempted). Under conditions of high temperatue, low humidy, low fuel moisture, and high winds – conditions that occur about once or twice a decade in Mesa Verde – fires in piñon-juniper forest and mountain shrublands cannot be controlled even with modern fire-fighting technology (Omi and Emrick 1980, Floyd et al. 2000).

Mountain shrublands are used primarily for grazing, wood-cutting, and recreation. Opportunities and interest in restoring or improving or otherwise modifying this type of vegetation appear limited. Prescribed burning or mechanical removal of biomass is sometimes employed to improve forage for elk, deer, and livestock, but the effectiveness of such treatment is usually short-lived because of the rapid re-sprouting and growth of the shrubs. Dense stands of Gambel oak are also recognized as a fire hazard in the wildland-urban interface because they can burn intensely under extreme fire weather conditions (Romme et al. 2006). At least some component of tall, dense shrublands should be retained in the landscape because they provide valuable nesting habitat for several bird species, including green-tailed towhee and orange-crowned warbler.

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F. LODGEPOLE PINE FORESTS

Lodgepole pine (*Pinus contorta* var. *latifolia*) is a distinctly uncommon species in the South Central Highlands Section. Natural stands of lodgepole pine are found only in the northeastern portion of the area, at mid elevations on the northern slopes of the La Garita Range. Lodgepole pine also has been planted in the Lime Creek burn area near Molas Pass and Coal Bank Pass, just south of Silverton, CO, and several places on the Uncompahgre Plateau.

Lodgepole pine extends into south-central Colorado in the Sangre de Cristo Range which lies to the east of the South Central Highlands (Peet 1978, Allen et al. 1991). However, the lodgepole pine forests that grow on the northern flank of the La Garita Range are at the southernmost limit of distribution of this species in southwestern Colorado. Lodgepole pine is a major forest type in northern Colorado and Wyoming, where it has received extensive study (e.g., Peet 1988, Knight 1994, Knight and Reiners 2000, Veblen 2000). However, we are aware of no reseaerch conducted in the stands in the South Central Highlands Section. The extent to which we can extrapolate from studies conducted farther north to the lodgepole pine forests in our area is unknown. Therefore, rather than extensively summarizing that literature in this report, we conclude only that it may be applicable here or it may not, and that this vegetation type is simply not well understood in our area.

The lodgepole pine stands that were planted near Silverton are interesting in two respects. First, they were planted in the middle decades of the 20th century because of concern over the slow rate of natural re-forestation in the Lime Creek burn which occurred in 1879 and affected ca. 25,000 acres (10,000 ha) of spruce-fir, aspen, and cold wet mixed conifer forest at elevations of approximately 9,500 – 10,500 ft (2,900 – 3,200 m) (Paulson and Baker 2006). The second interesting aspect of these lodgepole pine stands is that they appear to be thriving. This is surprising given the fact that they are growing outside the pre-1900 range of the species. Two plausible explanations can be suggested. First, the species may not have occupied all potentially suitable

habitat even before 1900, because of barriers to dispersal or simply slow dispersal into the San Juan region from sources to the north. However, there are no obvious barriers between the La Garita population and the Lime Creek area, and broad-scale migration patterns of this species during the last thousand years are unknown. The second possible explanation is that the potentially suitable range for this species has expanded in the 20th century as a result of climatic warming. At this time, we are aware of no way to critically test these two hypotheses.

Although lodgepole pine is an important timber species in northern Colorado and Wyoming, it is of limited economic importance in the South Central Highlands Section because of its very restricted distribution and abundance. Consequently, opportunities for management of lodgepole pine in our region are limited. In fact, there is concern that lodgepole pine could become an undesirable invasive species in portions of the San Juan Mountains (Paulson and Baker 2006), as has occurred in other parts of the world such as Sweden (Engelmark et al. 2001).

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G. BRISTLECONE PINE FORESTS

Bristlecone pine (*Pinus aristata*) is an uncommon species and forest type in the South Central Highlands, but adds an interesting and important component of biodiversity where it does occur. The oldest trees in Colorado are 2,400+ year old bristlecone pine (Brunstein and Yamaguchi 1992). Because bristlecone pine is relatively uncommon and unimportant commercially, it has received little research attention in the South Central Highlands Section. However, two recent studies by W. L. Baker and colleagues (below) provide baseline information about distribution and varaibility in community composition of bristlecone pine forests in the Colorado portion of the Section, while DeVelice et al. (1986) characterize some of the stands in northern New Mexico.

Bristlecone pine forests grow primarily on dry, steep, south-facing slopes at elevations of 2700-3700 m in the Front, Mosquito, Sawatch, Sangre de Cristo, and San Juan Mountain Ranges (Peet 1978, Baker 1992, Ranne et al. 1997). The environment generally is harsh: soils are poorly developed, temperatures are cool even in summer, and strong winds are common. Many trees, especially larger and older individuals, exhibit partial cambial mortality (strips of bare wood where the bark has died and sloughed off), perhaps because of wind or other damage. Most stands in the San Juan Mountains are on extrusive igneous substrates, including andesitic lavas, breccias, tuffs, and conglomerates (Ranne et al. 1997;22). Bristlecone pine in the San Juan Mountains is largely confined to the eastern and northeastern portions of the range, in the Rio Grande and Gunnison drainages (Baker 1992, Ranne et al. 1997, and *personal observations*). Ranne et al. (1997) identified two major plant associations dominated by bristlecone pine in the San Juan Mountains: a *Pinus aristata / Festuca arizonica* association at lower elevations, and a *Pinus aristata / Festuca thurberi* association at higher elevations. Bristlecone pine apparently is not a "climax" species, but requires disturbance for abundant regeneration (Baker 1992). Snow avalanches can remove large patches of bristlecone pine, and some cutting for mine timbers occurred in the late 19th century (Ranne et al. 1997;22).

Given the limited distribution and economic importance of bristlecone pine in the South Central Highlands Section, opportunities for restoration or management are limited. It does not appear to face any imminent threats. However, a potential future threat is white pine blister rust, an exotic fungal disease that has devastated five-needle pines in the the northern Rocky Mountains. White pine blister rust is rare in Colorado at this time, and has not been reported in the South Central Highlands Section. However, it exists in the Colorado Front Range and in the Sangre de Cristo Range, so could appear in this region at any time.

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CHAPTER VIII. MANAGEMENT CHALLENGES AND OPPORTUNITIES

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Many of the ecosystems of the South Central Highlands Section today are substantially different from the ecosystems that existed during the reference period. These changes have come about primarily through human activities during the last century although some key climatic events also have played a role. However, the magnitude and causes of change vary greatly among ecosystem types. Dramatic and extensive changes have occurred, for example, in mountain grasslands (due primarily to grazing), alpine ecosystems (grazing, mining, and recreation), riparian vegetation (grazing and water diversion), and ponderosa pine forests (logging, grazing, and fire exclusion). In contrast, much of the spruce-fir and cool moist mixed conifer forest in the region has been little altered, except in specific locations where logging and road-building have occurred. Many of the changes since the reference period are regarded as undesirable; for example, the ponderosa pine forests of the region are currently at risk from extensive insect outbreaks, uncontrollable wildfires, and reduced biological diversity. Long-term timber supply also is questionable in some areas.

One of the major land management activities in the twenty-first century likely will be restoration of ecosystems that were degraded during the past century. In this chapter we first review some of the major challenges facing land managers in the South Central Highlands Section, and then explore some of the opportunities for mitigation and restoration of damaged or degraded components of the ecosystems in the region.

A. Challenges: Forest Fragmentation by Roads & Logging

One of the major global challenges to biodiversity and ecological integrity is fragmentation of wildland habitats by roads, logging, agriculture, home building, and other human developments. It can be argued that habitat fragmentation by roads and logging is a less serious issue in the Rocky Mountain region because the topography and vegetation of these landscapes are naturally heterogeneous, i.e., they are "naturally fragmented" to begin with and the biota are well adapted to such fragmentation. An alternative view is that additional anthropogenic fragmentation is especially deleterious in landscapes that are already highly fragmented (Buskirk et al. 2000, Knight and Reiners 2000). We do have some information about the ecological effects of roads and forest fragmentation in the Southern Rocky Mountains (Knight et al. 2000), but new research is needed to more fully evaluate the significance of natural and anthropogenic fragmentation of habitats in this region. In this section we first examine some general impacts of roads in any kind of landscape, and

then focus specifically on effects of roads and logging on forested landscapes in the South Central Highlands Section. We selected two representative areas for detailed analysis: (i) forested portions of the San Juan National Forest as a whole and (ii) the high-elevation portion of the Pagosa District of the San Juan National Forest.

1. General Ecological Impacts of Roads:

Roads have greater impacts on plants, wildlife, and natural ecological processes than almost any other feature of the human-influenced landscape. Roads are unprecedented features in ecological history (Forman 1995, Forman et al. 2003, Baker and Knight 2000, Ritters and Wickham 2003), and can cause greater fragmentation of mature forests than clear-cutting (Reed et al. 1996a, Tinker et al. 1997, McGarigal et al. 2001). The total extent of roads in the Southwest and Southern Rocky Mountain regions, including the South Central Highlands Section, is astonishing. For example, as of 1997 the Rio Grande and San Juan National Forests contained 5,150 miles of road, for a road density of 0.88 miles of road per square mile (Baker and Knight 2000; Table 5.1). These figures include the Wilderness Areas in those National Forests, so road density outside of Wilderness is even greater. Some of the adverse ecological effects of these roads include the following (Baker and Knight 2000; Table 5.1): (i) *direct on-road effects*, e.g., habitat loss and accidental road-kill; (ii) *road-zone effects*, e.g., wildlife avoidance of the corridor along the road, and increased human access for hunting and trampling of areas adjacent to the road; (iii) *connectivity effects*, e.g., providing avenues for the spread of non-native species while limiting the movement of some sensitive native species; and (iv) *landscape effects*, e.g., dissection of high-quality interior habitat patches, and increased habitat edge.

2. Road Densities in Forests of the San Juan National Forest

Road densities within the forested portions of the San Juan National Forest were estimated as follows. We overlaid the roads contained in a GIS cartographic features file over the area covered by each of the four major forest types -- ponderosa pine, aspen, mixed conifer (referred to as Douglas-fir / white fir in the data base), and spruce-fir -- within the Arc-Info environment. All kinds of roads, e.g., improved and primitive, were aggregated in the roads layer. We determined the total number of miles of roads within each forest type, as well as, the total square miles of each forest type. We then divided miles of roads by square miles of forest type to obtain a measure of miles of road per square mile of forest. This analysis was done first for the entire San Juan National Forest, including officially designated Wilderness and roadless areas, and then separately for the entire Forest but excluding designated Wilderness Areas.

Road densities are greatest in the ponderosa pine forest type, with 2.60 miles of road per square mile for the entire Forest including Wilderness Areas, or 2.61 miles of road per square mile excluding Wilderness Areas (Table VII-1). The reason for the small difference in these numbers is that very little ponderosa pine forest is contained within officially designated Wilderness Areas, which tend to be at higher elevations. Road density is somewhat

lower in spruce-fir forests: 1.07 miles of road per square mile for the entire Forest but 1.75 miles of road per square mile for those areas outside of Wilderness Areas. Road densities are also lower in the aspen type, with 1.11 miles of road per square mile for the entire Forest or 1.21 miles of road per square mile outside of Wilderness Areas. Lowest road densities are in the Douglas-fir/white fir (mixed conifer) type (0.87 miles of road per square mile in the entire Forest), but even this forest type has nearly a mile of road per square mile outside Wilderness Areas (Table VII-1).

For comparison, Reed et al. (1996a) found 2.52 km of road per km^2 (4.1 miles per mi^2) in a subalpine landscape of the Medicine Bow - Routt National Forest in southeastern Wyoming. Logging and road-building during the last 50 years had substantially fragmented the forest in this watershed, to the extent that it was now more fragmented than the Oregon Cascades (Reed et al. 1996b). Road densities in the San Juan National Forest are lower than this extreme example from southeastern Wyoming, but nevertheless relatively high.

Table VII-1. Road densities in the four major forest types of the San Juan National Forest. Data are from RIS cover maps and cartographic feature files, overlain in Arc-Info environment. It should be noted that the road coverage is incomplete. Forest Service staff estimate that 20 - 30 % of the roads that actually exist on the ground are not included in the current GIS roads file. The roads GIS files will be updated within the next few years. However, the road densities in this table are under-estimates of actual road densities on the ground.

Forest Type	Miles of Road per Square Mile: Entire San Juan National Forest Including Wilderness Areas (Total Number of Square Miles in Parentheses)	Miles of Road per Square Mile: San Juan NF Excluding Wilderness Areas (Total Number of Square Miles in Parentheses)
Ponderosa Pine	2.60 miles/square mile (472.4 square miles)	2.61 miles/square mile (470.5 square miles)
Aspen	1.11 miles/square mile (480.6 square miles)	1.21 miles/square mile (444.3 square miles)
Douglas-Fir & White Fir	0.87 miles/square mile (381.2 square miles)	0.97 miles/square mile (340.1 square miles)
Spruce-Fir	1.07 miles/square mile (876.5 square miles)	1.75 miles/square mile (537.3 square miles)

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3. Effects of Roads and Logging on a High-Elevation Landscape of the Pagosa District

To obtain quantitative estimates of the impact of road-building and logging on high-elevation landscape structure in the South Central Highlands Section, we first reconstructed the history of road-building and logging in the area, and then measured changes in several metrics of landscape structure using the program FRAGSTATS, as explained below.

Logging History in High-Elevation Forests of the Pagosa District:

To evaluate quantitatively the effects of road-building and logging on high-elevation forests in a representative portion of the South Central Highlands Section, we needed a detailed history of logging operations during the 20th century. However, for most of the area, detailed records of this kind either do not exist, or are scattered throughout numerous informal reports, paper files, and recollections of individual foresters. To compile the records necessary for a rigorous analysis of landscape change as a result of logging and road building, we focused on the Pagosa District of the San Juan National Forest. This district is located near the approximate geographic center of the South Central Highlands Section, contains extensive high-elevation forests and other associated vegetation types, and still has most of the records of its logging history.

We reconstructed the logging history of the last half-century by tabulating the data contained in two timber atlases on file at the Pagosa Ranger Station ("Piedra timber management records" and "Pagosa

timber management records"), and supplemented this information by interviewing Stu Sarnow, recently retired silviculturalist for the Pagosa District. Logging occurred in ponderosa pine forests as early as the late 1800s, and by the middle of the 20th century most of the large, old-growth ponderosa pine trees had been removed from lower-elevation forests in the Pagosa District and throughout the San Juan National Forest (Chapter II). However, because of inaccessibility and a short summer season, the high-elevation forests of the Pagosa District (and some other portions of the South Central Highlands Section) were not logged extensively until after World War II. Virtually no written records exist of logging operations prior to 1950. However, we are confident that little activity occurred in high-elevation forests prior to this date because nearly all of the roads that now exist in high-elevation areas were documented as having been constructed during the last 50 years. Therefore, we used the year 1950 as a baseline and measured the changes that have occurred since that time.

The number of acres cut each year (in all forest types combined) from 1951 – 1992 in the Pagosa District of the San Juan National Forest is presented in Figure VII-1. The number of acres affected by logging each year increased through the 1950s and 1960s, reached a peak in the 1970s, and then declined through the 1980s and 1990s. The amount of clear-cut area was greatest in the late 1960s, and was low in the 1980s and 1990s. Much of the clear-cutting in the 1960s was in spruce-fir forests, but nearly all of the recent clear-cutting has been in aspen forests. Both clear-cutting and partial cutting have been in predominantly small cutting units, < 250

acres, although a few relatively large units (up to 3,000 acres) exist (Table VII-2).

Logging history by forest type in the Pagosa District of the San Juan National Forest is summarized in Table VII-3. The most extensive logging activity in *spruce-fir forests* occurred during the 1960s and 1970s, decreased somewhat in the 1980s, and decreased further in the 1990s. No clear-cutting has been done in spruce-fir forests since the 1970s; recent logging has emphasized partial-cut, selective harvest, salvage sale, and shelter wood methods. Largely because much of the Pagosa District is in Wilderness Areas where logging is not permitted, a half-century of logging has directly affected only a small proportion (ca. 5 %) of the total spruce-fir forest acreage on the Pagosa District. However, nearly all (85%) of the suitable timber base in spruce-fir has been entered at least once for timber removal (last column in Table VII-3).

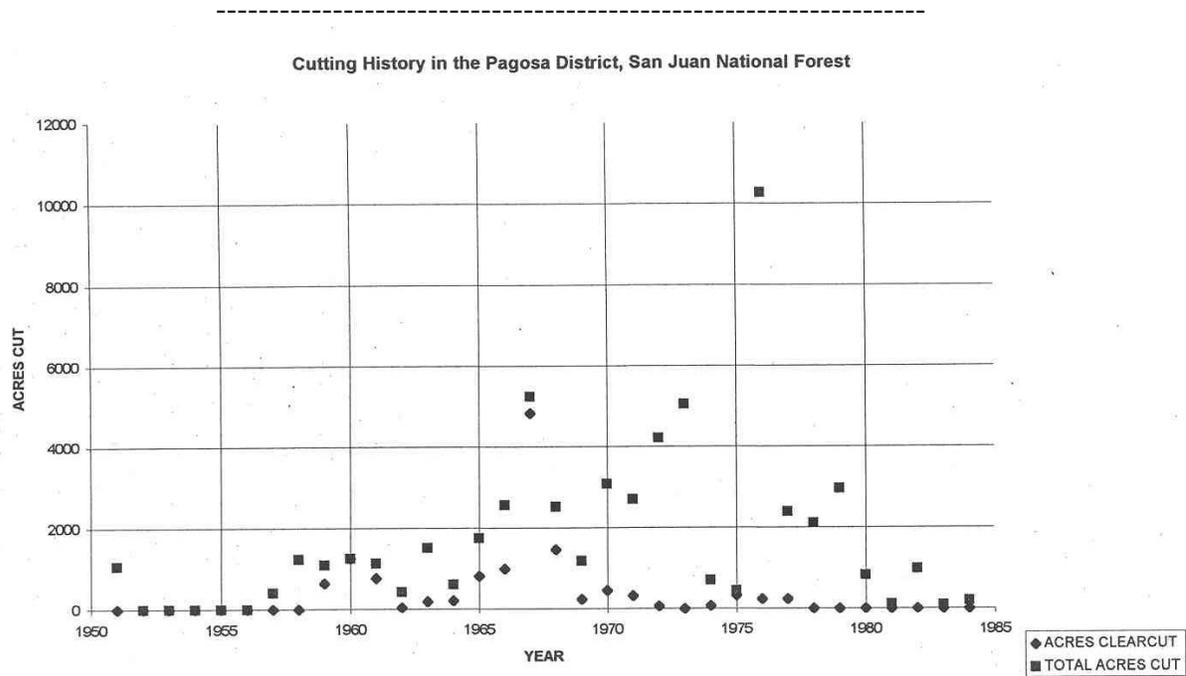---

Figure VII-1. Logging history in high-elevation forests (spruce-fir, cool moist mixed conifer, and high-elevation aspen forests) of the Pagosa District, San Juan National Forest, 1950-1992.

Since 1950, only about 20 % of the *mixed conifer forest* acreage in the Pagosa District has been entered for timber harvest. However, much of the mixed conifer type grows on steep slopes or in administratively designated Wilderness Areas where logging is not possible. Of the mixed conifer forest lands currently considered suitable for timber harvest, approximately 70 % have received at least an initial harvest entry (Table VII-3).

The most extensive logging in *aspen forests* of the Pagosa District took

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Table VII-1. Road densities in the four major forest types of the San Juan National Forest. Data are from RIS cover maps and cartographic feature files, overlain in Arc-Info environment. It should be noted that the road coverage is incomplete. Forest Service staff estimate that 20 - 30 % of the roads that actually exist on the ground are not included in the current GIS roads file. The roads GIS files will be updated within the next few years. However, the road densities in this table are under-estimates of actual road densities on the ground.

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Ponderosa Pine	2.60 miles/square mile (472.4 square miles)	2.61 miles/square mile (470.5 square miles)
Aspen	1.11 miles/square mile (480.6 square miles)	1.21 miles/square mile (444.3 square miles)
Douglas-Fir & White Fir	0.87 miles/square mile (381.2 square miles)	0.97 miles/square mile (340.1 square miles)
Spruce-Fir	1.07 miles/square mile (876.5 square miles)	1.75 miles/square mile (537.3 square miles)

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place in the 1950s through 1970s, when a total of about 600 acres were cut. Aspen timber production then was substantially lower during the 1980s and early 1990s. By 1992, about 1 % of the total extent of aspen forests in the Pagosa District, and a little over 5 % of the aspen forests currently considered suitable for timber production, had been either clear-cut or partially cut (Table VII-3). It should be noted that a much higher proportion of the aspen forest and of the suitable timber base in aspen has been logged since 1950 in the Dolores District of the San Juan National Forest (data not shown).

Caveats: In interpreting the results of our fragmentation analysis, it is important to acknowledge some of the inherent problems with the database we had to work with. First, the timber cutting records are incomplete. Stu Sarnow believes that only 85-90 % of the actual cutting activity between 1950 and 1992 is contained within the two timber atlases. Additional logged areas were identified from the Forest Service RIS database and from recent landsat and aerial photo coverage of the area. These additional areas were included in the numbers presented in Table VII-3, but there may still be other areas that

were partially logged and are not represented in our data. Secondly, land classification systems have changed during the period from 1950 - 1992. In particular, the area of suitable timber base has been reduced. For the entire San Juan National Forest, there were 1,188,000 acres designated as suitable for commercial use in 1962, but only 375,000 acres so designated in 1992. In addition, some of the clear-cut areas that lack reforestation are currently classified as shrub, barren, or grass/forb cover-types whereas they were originally spruce-fir forest, and some recent logging units included portions of previously logged units.

Table VII-2. Size distribution of timber cutting units in the Pagosa District, San Juan National Forest, from 1950 - 1992. Data were compiled from two timber atlases on file at the Pagosa Ranger Station ("Piedra timber management records" and "Pagosa timber management records"), and supplemented by personal communications from Stu Sarnow, silviculturalist for the Pagosa District. Stu Sarnow believes that 85-90 % of the actual cutting activity between 1950 and 1992 is contained within the two timber atlases. The category "acres partially cut" includes partial-cut, selective harvest, salvage sale, and shelterwood methods. This tabulation includes all timber cutting units within the Pagosa District, including cuts in spruce-fir, mixed conifer, aspen and ponderosa pine forests.

Size of Unit (acres)	Number of Partial Cut Units	Number of Clear-cut Units	Total Number of Cutting Units
1 - 250	113	41	154
251 - 500	23	4	27
501 - 750	24	0	24
751 - 1,000	10	0	10
1,001 - 1,250	15	0	15
1,251 - 1,500	2	0	2
1,501 - 1,750	4	0	4
1,751 - 2,000	2	0	2
2,001 - 2,250	2	0	2
2,251 - 2,500	2	0	2
2,501 - 2,750	0	0	0
2,751 - 3,000	0	0	0
3,001 - 3,250	1	0	1
Total, all sizes	198	45	243

Table VII-3. Logging history in high-elevation forests of the Pagosa District, San Juan National Forest, 1950 - 1992. Data were compiled from two timber atlases on file at the Pagosa Ranger Station ("Piedra timber management records" and "Pagosa timber management records"), and supplemented by personal communications from Stu Sarnow, silviculturalist for the Pagosa District. The category "acres partially cut" includes partial-cut, selective harvest, salvage sale, and shelterwood methods. It must be noted that there are several irremediable problems with this data set:

   (1) The timber cutting records are incomplete. Stu Sarnow believes that 85-90 % of the actual cutting activity between 1950 and 1992 is contained within the two timber atlases. Additional logged areas were identified from the Forest Service RIS database and from recent landsat and aerial photo coverage of the area. These additional areas are included in the numbers below.

   (2) Land classification systems have changed during the period from 1950 - 1992. In particular, the area of suitable timber base has been reduced. For the entire San Juan National Forest, there were 1,188,000 acres designated as suitable for commercial use in 1962, but only 375,000 acres so designated in 1992. In addition, some of the clear-cut areas that lack reforestation are currently classified as shrub, barren, or grass/forb cover types.

Decade	Acres Clear-cut	Acres Partially Cut	Total Acres Cut	Cumulative Total Acres Cut Since 1950	Cumulative % of Total Forest Area Cut Since 1950[1]	Cumulative % of Total Suitable Timber Base Cut Since 1950[2]
Spruce-Fir Forest						
1950-59	0	0	0	0	0	0
1960-69	2,751	158	2,909	2,909	1.7	28.4
1970-79	722	2,542	3,264	6,173	3.6	60.4
1980-89	0	2,480	2,480	8,653	5.0	84.6
1990-92	0	0	0	8,653	5.0	84.6
Mixed Conifer Forest						
1950-59	0	3,087	3,087	3,087	2.6	9.5
1960-69	148	7,072	7,220	10,307	8.6	31.6
1970-79	0	9,483	9,483	19,790	16.5	60.7
1980-89	12	1,811	1,823	21,613	18.0	66.3
1990-92	21	1,047	1,068	22,681	18.9	70.0
Aspen Forest						
1950-59	81	167	248	248	0.4	2.0
1960-69	85	88	173	421	0.7	3.4
1970-79	177	0	177	598	1.0	4.8
1980-89	9	0	9	607	1.0	4.9
1990-92	36	10	46	653	1.1	5.3

[1] Total forest area on the Pagosa District:
   Spruce-Fir forest = 173,793 acres
   Mixed conifer forest = 120,279 acres
   Aspen forest = 58,438

[2] Total suitable timber base:
   Spruce-Fir forest = 10,226 acres
   Mixed conifer forest = 32,619 acres
   Aspen forest = 12,356 acres

Given these irremediable shortcomings of the database, we cannot have complete confidence in any specific estimates of the percent of the landscape or the percent of the suitable timber base that have been affected by logging of spruce-fir forests during the last 50 years. A high priority for the future should be to improve record-keeping, e.g., to regularly incorporate all timber harvest units and other activities into the Forest's GIS and other databases. Especially as older Forest Service staff members retire or leave the area, it will become more and more difficult to compile a database of logging history of the kind we have presented here. This kind of information will be increasingly valuable to future users and forest managers.

Methods of Fragmentation Analysis:

To obtain quantitative estimates of the impact of 50 years of road-building and logging on high-elevation landscape structure in the South Central Highlands Section, we measured changes in several metrics of landscape structure using the program FRAGSTATS. The following is a brief synopsis of results; see McGarigal et al. (2001) for additional details.

For this analysis we first identified a representative area of high-elevation forests in the Pagosa District of the San Juan National Forest. Initially we located all lands above 2,420 m (8,000 ft), using the GIS. The resulting map showed a large block of high-elevation landscape in the northern and eastern portions of the Pagosa District, along the main crest of the San Juan Mountain Range. In addition, there were numerous "islands" of high-elevation terrain surrounded by lower-elevation terrain in the foothills of the southern and western portions of the District. To minimize map-edge effects in the analysis, we subjectively smoothed this initial GIS map, deleting a few small, isolated islands of high-elevation terrain, and incorporating small portions of lower-elevation terrain to connect some of the larger high-elevation islands. The resulting map (Figure VII-2) depicted an area of 225,843 ha, representing the high-elevation landscape of the Pagosa District.

Within this study area, we then created two GIS maps: the landscape of 1950 and the landscape of 1993. We regarded the 1950 map as representative of the reference condition landscape, because we knew from our analysis of logging history that very little logging or road-building had occurred in this area prior to 1950 (see above). There also were no large fires or insect outbreaks in the area during the first half of the 20th century, so mature forests may have been more extensive and continuous in 1950 than at some earlier times during the period of indigenous settlement. However, the low frequency of fires during the 20th century may not be a great departure from the reference period range of variability of high-elevation forests in this region (see above). Both maps were created at a scale of 1 ha resolution (i.e., each pixel represents 1 ha).

To produce the 1993 landscape map, we began with a 1993 vegetation map of the area that was derived from aerial photos and satellite imagery. This map was reclassified (in the GIS environment) into 14 categories (Table VII-4).
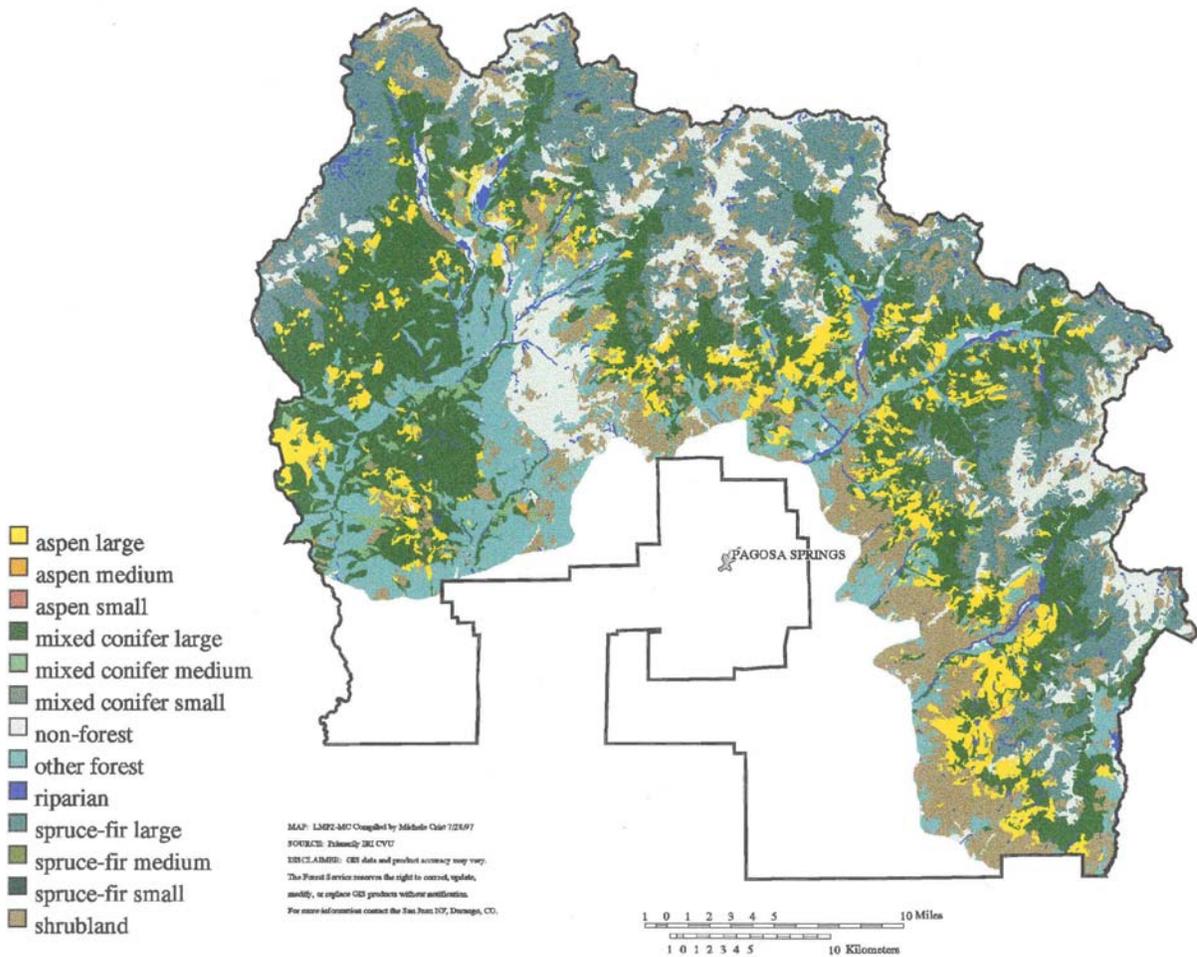
Figure VII-2. High-elevation study area in the Pagosa District, San Juan National Forest, where logging history and forest fragmentation were studied.

We actually made four versions of the 1993 landscape map. First, we simply reclassified the entire vegetation map into these 14 categories. Areas that had been logged during the last 50 years were assigned to one of the vegetation classes, based solely on the structure of the remaining vegetation as perceived on the aerial photos and satellite imagery. Clear-cut areas were assigned to the seedling & sapling class, but many partially cut areas were classified as forests with medium or large trees because a large number of such trees had been left after logging. This map depicted a minimum impact of logging; i.e., it assumed that partial cutting produced no change in stand structure unless the canopy was almost completely removed.

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Table VII-4. Vegetation categories used in the fragmentation analysis of a 225,843-ha high-elevation study area in the Pagosa District of the San Juan National Forest (Figure VII-2).

-- spruce-fir forest, large or very large trees (9 + inches dbh)
-- spruce-fir forest, medium-sized trees (5 - 8.9 inches dbh)
-- spruce-fir forest, seedlings & saplings (< 5 inches dbh)
-- mixed conifer forest, large or very large trees (9 + inches dbh)
-- mixed conifer forest, medium-sized trees (5 - 8.9 inches dbh)
-- mixed conifer forest, seedlings & saplings (< 5 inches dbh)
-- aspen forest, large or very large trees (9 + inches dbh)
-- aspen forest, medium-sized trees (5 - 8.9 inches dbh)
-- aspen forest, saplings (< 5 inches dbh)
-- other forest types (ponderosa pine, pinon-juniper)
-- shrubland (oak, snowberry, willow carrs)
-- riparian (water and riparian vegetation)
-- nonforest (meadows, grasslands, rocky slopes and scree)
-- roads

For the second 1993 landscape map, we overlaid the vegetation map described above with the timber harvest map that we had developed from historical records in the Pagosa District (see above), and classified all partially logged and clear-cut areas as separate vegetation types, regardless of the nature of the residual vegetation. This provided a maximum estimate of the impact of logging, because it assumed that any entry into a previously unlogged stand produced significant changes in stand structure or function. Both of these first two maps included roads as a land cover type. For the third and fourth versions of the 1993 map, we reclassified the road pixels as adjacent vegetation types. This effectively "removed" the roads from the maps, so that we could evaluate the relative contribution of roads to overall changes in landscape structure.

We created the 1950 landscape map from the 1993 map by reclassifying (in the GIS environment) all of the logged areas as large/very large trees of the forest type indicated by the major species of timber removed from the area according to the timber records. We thought it was reasonable to assume that the logged areas were all mature forest at the time of logging, because during this period most logging in high-elevation forests of this region was done as a first entry into mature and "over-mature" stands. Roads were then reclassified as the same cover types as surrounding pixels. In effect, we thereby "removed" all of the roads and timber-cutting patches from the landscape (with the exception of a few roads known to have existed in 1950, e.g., the main highway over Wolf Creek Pass).

Results of the Fragmentation Analysis:

A comparison of landscape metrics for the 1950 map and for the various 1993 maps reveals that the high-elevation landscape of the Pagosa District is more heterogeneous today than it was before the onset of extensive logging and road-building. Roads have had a larger impact on landscape structure than logging, even when partially cut areas are assigned their own unique cover type. FRAGSTATS determines 45 different metrics (measurements) of landscape structure. The average percentage change from 1950 - 1993, among all of these metrics, was 15 - 25 % when roads were included, but only 3 - 8 % when roads were excluded (Figure VII-3).

In what specific ways has the landscape changed since 1950? The number of distinct patches has increased -- by 2 - 14% excluding roads, or by 22 - 40 % if roads are included (Figure VII-4). Average patch size has decreased -- by 2 - 12 % without roads, or 18 - 29 % with roads (Figure VII-5). Including the effects of roads, the average size of a patch of any cover type was 55 ha in 1950, but only 35 - 45 ha in 1993. Edge density (meters of edge between two different cover types) has increased by 3 - 5 % without roads or 18 - 23 % with roads (Figure VII-6).

Of particular conservation significance is old-growth spruce-fir forest, which supports several species of plants, animals, and microbes that are thought to be sensitive to forest structural change and landscape fragmentation (Romme et al. 1992). Spruce-fir forests with large tree size classes (the cover class in our analysis

that likely represents most of the old-growth forests in the area) occupied 21.5 % of the study area in 1950, but only 20.0 % of the study area in 1993 if partially cut areas are treated as a separate cover type (figure VII-7). This is a relatively small change in overall extent of late successional forests, and the changes in patch number and patch size are more impressive. The number of patches of spruce-fir forest with large trees increased from 240 in 1950 to 280 in 1993 (including roads, and treating partial cuts as a separate cover type; Figure VII-8). The average patch size of spruce-fir forests with large trees decreased from 200 ha in 1950 to 150 ha in 1993 (Figure VII-9).

A simple measure of the size of a patch of late successional forest may be misleading, because the perimeter of the patch is actually edge environment where meteorological conditions and biotic pressures (e.g., nest predation and parasitism, human disturbance, sources of alien plant species) are quite different from the interior of the patch. Therefore, we also computed the "core" area of each patch of spruce-fir forest with large trees, by subtracting the area of each patch lying within 50 m of the edge. Mean core area decreased from 160 ha in 1950 to 120 ha in 1993 (including roads, and treating partial cuts as a separate cover type; Figure VII-10).

Evaluating the Significance of Forest Fragmentation in the Pagosa District:

In some ways, the overall landscape structure of this portion of the San Juan National Forest has been changed very little by a half century of logging and road-building.
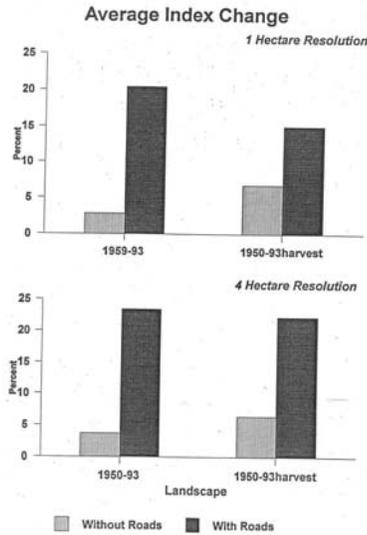
Figure VII-3. Average change over all
FRAGSTATS indices from 1950 – 1993
in the high-elevation study area of the
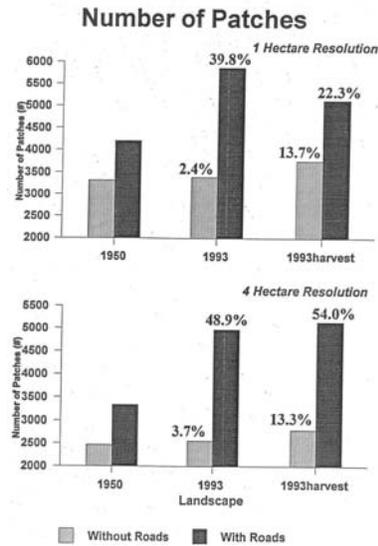Pagosa District, San Juan National Forest.

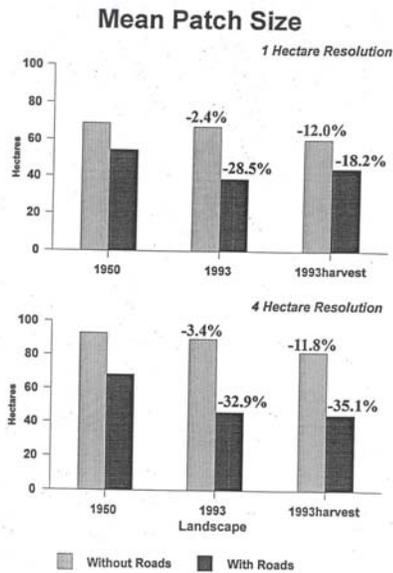Figure VII-4. Change in number
of patches from 1950 - 1993
in the high-elevation study area

Figure VII-5. Change in mean patch size
from 1950 – 1993 , in the high-elevation
study area
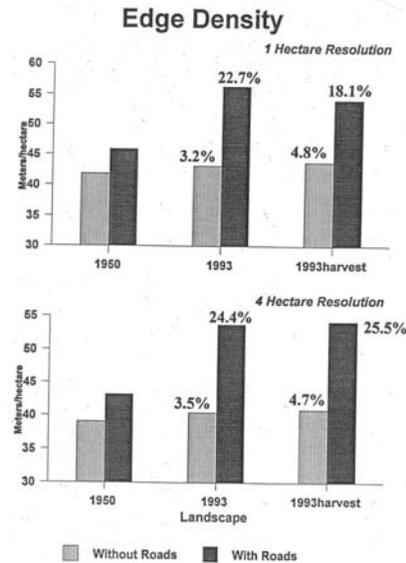
Figure VII-6. Change in edge density
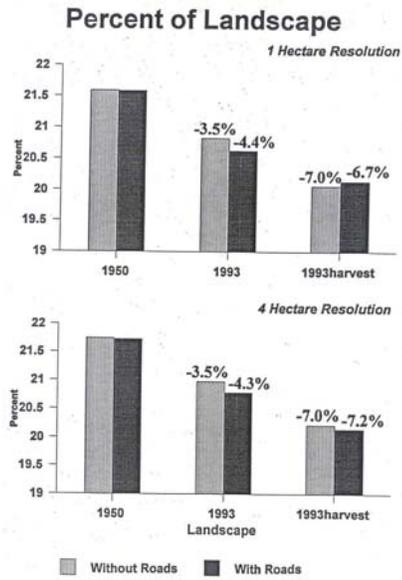from 1950 - 1993, in the high-elevation
study area

Figure VII-7. Change in percent of landscape occupied by spruce-fir forests with large or very large trees from 1950 – 1993, in the high-elevation study area of the San Juan National Forest.

Figure VII-8. Change in number of patches of spruce-fir forest with large or very large trees from 1950 – 1993. in the high-elevation study area.
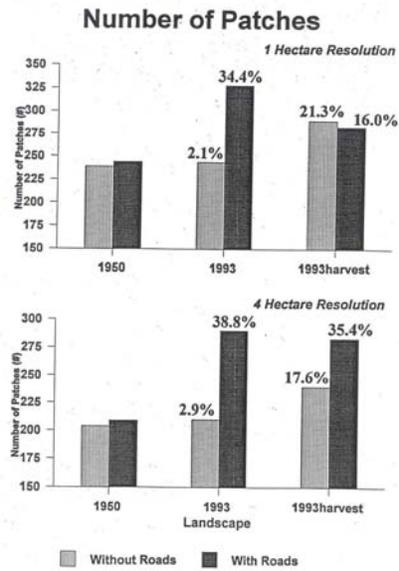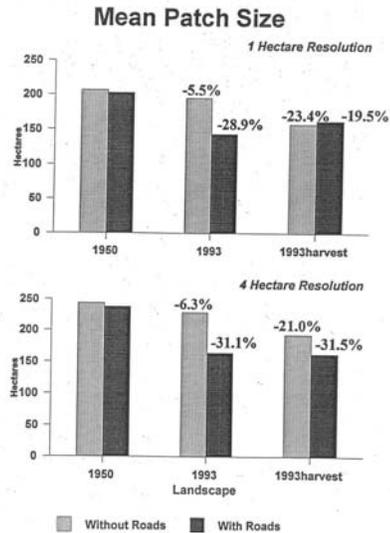

Figure VII-9. Change in mean size of patches of spruce-fir forest with large or very large trees from 1950 - 1993 in the high-elevation study area
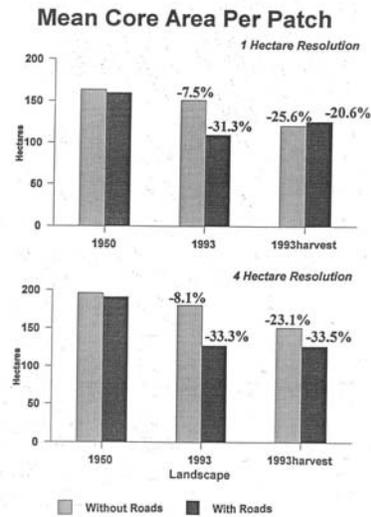

Figure VII-10. Change in mean core area per patch of spruce-fir forest with large or very large trees from 1950 - 1993

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For example, the total extent of late successional spruce-fir forest has decreased only from 21.5 % of the landscape to 20.0 %, even under the modeling assumptions that maximize the fragmenting effects of human activities (i.e., including roads and treating partially cut areas as a separate cover type; Figure VII-7). However, some of the more subtle landscape changes of the last 50 years are probably significant concerns for conservation of sensitive interior forest species, e.g., the reduction in average size of core areas within patches of late successional forest from 160 ha to only 120 ha (Figure VII-10). The magnitude of these quantitative estimates of landscape change also is buffered by the presence of a large Wilderness Area within the Pagosa District, where road-building and logging are administratively prohibited. A similar analysis for just those lands suitable for timber production would reveal much greater landscape change and fragmentation of mature forests. Additional FRAGSTATS analyses for the entire San Juan National Forest are in progress to evaluate the functional ecological significance of the changes we have detected in the high-elevation landscape structure of the Pagosa District (http://www.umass.edu/landeco/index.html).

A notable finding in our compilation of logging history was that nearly all of the suitable timber base in the Pagosa District has been entered at least once since the mid 20th century (Table VII-3). Does this matter for sensitive species or ecological processes, considering for example that ca. 90 % of spruce-fir forests lie outside the suitable timber base and therefore remain undisturbed? There are two important

implications of our finding that virtually all of the spruce-fir forests considered suitable for timber production have already sustained at least some degree of timber harvest activity. First, there simply are few opportunities left for first entries into unlogged stands, which were a predominant type of logging operation during the last half century. Future silviculture will need to shift from an emphasis on first entries to utilization or restoration of forests that have already had some of the timber taken out.

The second implication has to do with the fact that lands designated suitable for timber production tend to be the more biologically productive areas on relatively gentle terrain. Recent theoretical developments in ecology suggest that heavy utilization of the most productive forests (i.e., the suitable timber base) may potentially impair the long-term maintenance of native biodiversity. Many species of plants and animals are now thought to exist as "meta-populations" consisting of numerous small subpopulations that are more or less discrete, but periodically exchange individuals via migration (Pulliam 1988, Gilpin and Hanski 1991, Robinson et al. 1995). Small populations are vulnerable to local extinction, due to environmental disturbances, genetic processes, or random drift. Individual subpopulations within the "meta-population" probably go extinct on a regular basis, but they are continually re-established by individuals migrating in from other subpopulations. From a conservation standpoint, the most critical subpopulations are those that sustain positive population growth, i.e., in which production of new individuals exceeds mortality in most years. These are referred to as "source" populations; whereas other

subpopulations, in which mortality usually exceeds recruitment, are called "sink" populations. Sink populations persist only because of the continued flow of individuals from the source populations. If a sink population is extirpated, it can be easily re-established if the source populations are still intact. However, if a source population is wiped out -- by natural catastrophe or by human activities -- then the entire meta-population may crash, even though many sink populations are still present, because the sink populations are not self-sustaining.

The practical difficulty in all of this is that we generally do not know which subpopulations are sinks and which are sources. Therefore, the fact that a species is relatively numerous and widely distributed does not necessarily mean that it is secure. Most of the individuals seen may in fact constitute sink populations. For example, the neotropical migrant bird populations in forest fragments of the upper Midwest region are now regarded as sink populations that persist only because of continued migration from source populations in extensive forests of the Southern Appalachians and Ozark Plateaus (Robinson et al. 1995). Even though neotropical migrant birds can be seen and heard mating and laying eggs every spring in the upper Midwest, very few of the eggs survive the predators and nest parasites associated with fragmented forests of agricultural

landscapes. A top conservation priority is to maintain the source populations of these species; if habitats become unsuitable for successful breeding in the Southern Appalachians and Ozarks, then neotropical migrant populations may crash throughout eastern North America.

Where are the key source populations for various species in the South Central Highlands Section? We do not know. A better understanding of the meta-population structure of representative plants and animals of this region is a high research priority. Until specific information becomes available, it may be a good coarse-filter strategy to conserve habitats that have a high probability of supporting source populations. One such type of habitat may be mature forests growing on productive sites. The relatively high net primary productivity that is supported by these sites may also support relatively high densities of animals and allow for production of surplus individuals that can migrate to augment sink populations in marginal habitats (Hansen and Rotella 1999). Considering that nearly every spruce-fir stand growing in productive sites (i.e., the suitable timber base) has already had some level of timber cutting during the last 50 years, the few remaining old-growth spruce-fir stands within the suitable timber base probably should not be disturbed by future logging until the meta-population structure of high-elevation species in this region are better understood.

B. Challenges: Outdoor Recreation and Exurban Development

Outdoor recreation is now the principal use of most forests in the southern Rocky Mountains (Knight, 2000). Recreational use of Wilderness Areas has increased ten-fold in the last

40 years (Cole and Landres 1995, cited in Knight, 2000). Outdoor recreation is often regarded as a benign activity that has little or no adverse impact on ecological conditions or processes.

Unfortunately, this idea is far from the truth: outdoor recreation actually poses a major threat to biodiversity and ecological integrity in many wildland areas. Of all species federally listed as endangered or threatened, 133 plants and 70 animals are strongly affected by outdoor recreation—compared with 112 plant and 61 animals affected by grazing, 51 and 77 affected by logging, and 33 and 17 affected by hardrock mining. Only water development has a greater impact than recreation, with 67 plants and 189 animals so affected (Losos et al. 1995, cited in Knight 2000, Figure 7.1). In addition to traditional forms of recreation, a new type of "recreation" has exploded in the last 20 years as increasing numbers of people build homes in the forests and other wildlands of the Mountain West. Roads, trails, and homes all fragment natural landscapes, affect native plants and wildlife, and constrain our options for managing natural disturbance processes such as fire.

1. Ecological Impacts of Outdoor Recreation:

Outdoor recreation affects plant and wildlife species via direct harvest (hunting, fishing, and collecting), disturbance (either accidental or intentional), habitat modification, and pollution (Knight 2000). Recreationists often are unaware that their activities are agitating or displacing wildlife, but these effects can be substantial. For example, a study of songbird distribution in ponderosa pine forests managed by the City of Boulder, Colorado, revealed that several species (e.g., pygmy nuthatch, chipping sparrow, and Townsend's solitaire) were far less abundant within 100 meters of a trail, whereas the cosmopolitan robin was most abundant in close proximity to trails (Miller et al. 1998). Other studies have demonstrated that human use of recreational trails disturbs bears, deer, and elk, and alters their distribution in comparison with otherwise similar areas that lack regularly used trails (Knight 2000). Trails also can function as conduits for dispersal of weeds and other alien plant species (Benninger-Traux et al. 1992).

Even more destructive impacts can result from off-road recreational vehicle use, especially in sensitive areas like the alpine zone. The "Alpine Loop" in the San Juan Mountains receives a great deal of traffic each summer, most of it conducted responsibly with vehicles remaining on the rough dirt roads that traverse this spectacular route that lies largely above timberline. However, even a single vehicle driving off the road can destroy the fragile alpine vegetation, leading to severe erosion and sedimentation which can become worse with the passing of time. High-visibility vehicle scars and erosion channels of this kind are very difficult to rehabilitate, and may remain on the landscape for decades.

Knight (2000) made a preliminary estimate that about 6 % of Forest Service lands in Colorado are directly affected by recreational trails. Trail density in the San Juan - Rio Grande National Forest was estimated to be 0.26 km of trail per square kilometer of land, which is relatively low compared with some other national forests in Colorado (Knight 2000). However, trail densities should be added to road densities and off-road or off-trail activities (unmeasured) to evaluate the total impact of recreational activities on native plant and animal communities.

Recreational impacts can be reduced in a number of ways. When new roads and trails are being developed, they can be consciously placed in less sensitive locations. Many older trails run along streams or lakes— visually appealing places but also sites of high biodiversity and wildlife use— but new roads and trails can be built in alternative locations and old roads and trails that were placed in especially sensitive habitats can be closed (see Baker and Knight 2000; Table 5.3).

2. Ecological Impacts of Exurban Development:

The human population in the Rocky Mountain region is one of the fastest growing in the nation. For example, growth rates in this region from 1990-1996 were two to three times the national average (Theobald 2000). Although much of this growth is occurring in urban and suburban areas, a recent trend has been accelerated expansion into rural and previously undeveloped landscapes. Many former ranches and farms are being converted to exurban housing developments, as are small private inholdings within tracts of public land (Knight 1997). The new developments in these areas cannot be classified as traditional rural developments, for typically the new residents do not make their living on the local landscape. Rather they telecommute to urban centers or live on investment income. The most common pattern is widely dispersed houses on large lots, all connected by an extensive road system. Especially popular are home sites nestled in forests adjacent to public land, where aesthetic and recreational values are superb. Being not quite rural but not quite suburban,

this development pattern is referred to as *exurban*.

Exurban development poses several significant challenges for land managers and residents of the South Central Highlands Section. First, the houses and access roads fragment natural habitats and displace sensitive wildlife. For example, if recent rates and patterns of exurban development in La Plata County, Colorado, continue until ca. 2020, a quarter of waterfowl breeding areas, a third of waterfowl winter concentration areas, and a third of elk winter concentration areas will lie within or adjacent to rural subdivisions (Romme 1997). Second, exurban development often occurs in ponderosa pine forests and other vegetation types where fire was formerly an important ecological process. Not only are these houses vulnerable to loss in wildfire, but restoration of a natural fire regime is severely constrained by the presence of vulnerable structures that must be protected from fire. Finally, the costs to local governments of maintaining roads and providing services (fire protection, police, and schools) to these scattered residents usually exceed the revenues generated by property taxes; in effect, residents in town are subsidizing the exurban life style.

The ecological and social costs of exurban development can be reduced in a number of ways. For example, houses can be clustered rather than widely dispersed (Romme 1997, Paulson and Baker 2006). A clustered development pattern reduces the amount of habitat fragmentation and allows for more efficient provision of social services. Another strategy for protecting agricultural land from subdivision is the use of conservation easements that provide land owners with a financial

reward for keeping their land in production rather than selling it for development.

C. Challenges: Climate Change

It has become increasingly apparent that the earth's climate is changing, that global temperatures are increasing, that shifts in precipitation patterns are likely to become pronounced in the next century, and that weather related disturbances (e.g., hurricanes and droughts) likely become more frequent and severe (IPCC 2007). The potential for large, severe fire also is increasing as snowpacks melt earlier in the spring, leading to longer fire seasons. An upsurge in the frequency of large fires began in the mid-1980s and is expected to continue (Westerling et al. 2006).

Climate change complicates all of our efforts in land management. One of the greatest challenges relates to the great uncertainty about how climate will change in any particular place. For example, although the global trend of increasing temperature is well documented and likely to increase, some localities may see no temperature increase or even a cooling trend in future decades. Similarly, precipitation is expected to decrease in some areas but remain the same or increase in other areas (IPCC 2007).

Climate research continues, and our forecasting ability will undoubtedly increase substantially in coming years. In the meantime, Millar et al. (2007) recommend that managers "accept uncertainty yet certain change as premises for decision-making." The Millar et al. (2007) article and Millar's website provide a "toolbox" for managers, with a variety of potentially useful adaptation and mitigation strategies.

D. Opportunities: Principles of Ecological Restoration

One of the major land management activities in the twenty-first century likely will be restoration of ecosystems that were degraded during the past century. The Society for Ecological Restoration has defined and described restoration as follows (http://www.ser.org/content/ecological_restoration_primer.asp):

-- "Ecological restoration is an intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability. Frequently, the ecosystem that requires restoration has been degraded, damaged, transformed or entirely destroyed as the direct or indirect result of human activities. In some cases, these impacts to ecosystems have been caused or aggravated by natural agencies such as wildfire, floods, storms, or volcanic eruption, to the point at which the ecosystem cannot recover its predisturbance state or its historic developmental trajectory.

-- "Restoration attempts to return an ecosystem to its historic trajectory. Historic conditions are therefore the ideal starting point for restoration design. The restored ecosystem will not necessarily recover its former state, since contemporary constraints and conditions

may cause it to develop along an altered trajectory. The historic trajectory of a severely impacted ecosystem may be difficult or impossible to determine with accuracy. Nevertheless, the general direction and boundaries of that trajectory can be established through a combination of knowledge of the damaged ecosystem's pre-existing structure, composition and functioning, studies on comparable intact ecosystems, information about regional environmental conditions, and analysis of other ecological, cultural and historical reference information. These combined sources allow the historic trajectory or reference conditions to be charted from baseline ecological data and predictive models, and its emulation in the restoration process should aid in piloting the ecosystem towards improved health and integrity.

-- "Restoration represents an indefinitely long-term commitment of land and resources, and a proposal to restore an ecosystem requires thoughtful deliberation. Collective decisions are more likely to be honored and implemented than are those that are made unilaterally. For that reason, it behooves all stakeholders to arrive at the decision to initiate a restoration project by consensus. Once the decision to restore is made, the project requires careful and systematic planning and a monitored approach towards ecosystem recovery. The need for planning intensifies when the unit of restoration is a complex landscape of contiguous ecosystems.

-- "Interventions employed in restoration vary widely among projects, depending on the extent and duration of past disturbances, cultural conditions that have shaped the landscape, and contemporary constraints and opportunities. In the simplest circumstances, restoration consists of removing or modifying a specific disturbance, thereby allowing ecological processes to bring about an independent recovery. For example, removing a dam allows the return of an historical flooding regime. In more complex circumstances, restoration may also require the deliberate reintroduction of native species that have been lost, and the elimination or control of harmful, invasive exotic species to the greatest practicable extent. Often, ecosystem degradation or transformation has multiple, protracted sources, and the historical constituents of an ecosystem are substantially lost. Sometimes the developmental trajectory of a degraded ecosystem is blocked altogether, and its recovery through natural processes appears to be delayed indefinitely. In all of these cases, however, ecological restoration aims to initiate or facilitate the resumption of those processes which will return the ecosystem to its intended trajectory.

-- "When the desired trajectory is realized, the ecosystem under manipulation may no longer require external assistance to ensure its future health and integrity, in which case restoration can be considered complete. Nevertheless, the restored ecosystem often requires continuing management to counteract the invasion of opportunist species, the impacts of various human activities, climate change, and other unforeseeable events. In this respect, a restored ecosystem is no different from an undamaged ecosystem of the same kind, and both are likely to require some level of ecosystem management.

Although ecosystem restoration and ecosystem management form a continuum and often employ similar sorts of intervention, ecological restoration aims at assisting or initiating recovery, whereas ecosystem management is intended to guarantee the continued well-being of the restored ecosystem thereafter."

Two broad approaches to ecological restoration are active restoration and passive restoration. *Passive restoration* involves removal of the stressors that have degraded an ecosystem. Possible methods include regulating livestock grazing, firewood gathering, and recreational activity, and allowing lightning-ignited fires to burn without interference. *Active restoration* involves conscious manipulation and modification of an ecosystem to achieve a desired result. Commonly used methods of active restoration in our region include selective logging and manager-ignited prescribed fire. It is important to emphasize that active restoration is not needed (nor is it feasible) in every location throughout the South Central Highlands Section. Many of the forests in the region, especially unlogged stands at higher elevations (spruce-fir, cool-moist mixed conifer, and aspen forests) but also including some ponderosa pine forests, are relatively unaltered and still retain many or most of their pre-1870 characteristics. In these kinds of areas, a *passive restoration* approach may be sufficient to maintain and augment the desirable ecological characteristics that already exist.

Even where active restoration is needed, only certain components of the ecosystem may require intensive treatment. For example, in many persistent pinyon-juniper woodlands and wooded shrublands of the region, overstory structure and composition are not significantly departed from reference conditions (see Chapter VI-A), but the herbaceous understory has been dramatically altered by heavy livestock grazing. Especially where cheatgrass has become a major component of the understory, its influence on the fire regime may lead eventually to significant functional changes in the system as well, viz., a shift from infrequent to frequent fires. Because piñon and juniper are not fire-resistant, a new fire regime of frequent surface fires carried by cheatgrass could potentially destroy the overstory and convert the woodland to a shrub or herb-dominated community composed largely of non-native species. Thus, an active program of understory restoration is needed to eliminate or at least reduce the cheatgrass component and augment the native graminoid and forb components of the herbaceous community.

An important component of all restoration programs is protecting and maintaining those areas that have been little altered, to serve as reference areas for restoration treatments in degraded areas. A number of established Research Natural Areas (RNAs) is available for this purpose in the South Central Highlands Section (e.g., Narraguinnep and Williams Creek RNAs on the San Juan National Forest), but other potential high-quality RNAs exist that have not received formal designation. Examples in the San Juan National Forest include (i) an intact bunchgrass park along the Piedra Road (Paulson and Baker 2006;209), (ii) ponderosa pine forests on Archuleta Mesa which have largely escaped logging and grazing because the site is

inaccessible (Brown and Wu 2005), and (iii) mixed conifer forests in the extensive Hermosa and Piedra roadless areas.

E. Opportunities: Passive Restoration of Fire and Sustainable Grazing

Passive restoration involves removal of the stressors that have degraded an ecosystem. Two major stressors in the South Central Highlands Section are excessive fire suppression and poorly conducted livestock grazing. Note that neither fire suppression nor grazing is necessarily harmful; on the contrary, we must protect life, property, and resources from certain kinds of fire damage, and livestock grazing when well conducted actually can enhance ecological values (Knight et al. 2002, White 2008). The issue is not whether these activities are conducted, but how they are conducted. In the paragraphs below we explore opportunities for modifying fire suppression and livestock grazing programs to reduce their potential for adversely affecting ecosystems of the South Central Highlands Section.

1. Restoring Natural Fire Regimes

Throughout most of the 20th century wildland fire was viewed by most people and agencies in the U.S. as simply bad—destructive, usually human-caused, and largely unnatural (Pyne 1982). However, there is increasing recognition that fire is an essential ecological process in many ecosystems, including ecosystems of the South Central Highlands Section. Consequently, efforts are being made to return fire to the places where it is most needed and where its long absence has resulted in undesirable ecological changes. Ponderosa pine forest (Chapter II) may be the vegetation type most

altered by fire exclusion in this region, but all vegetation types have been affected to at least some degree, and changes related to the absence of natural fire will only become greater in the future if fire continues to be excluded from these ecosystems.

Land management agencies have opportunities to allow fires or portions of fires to burn naturally with minimal human interference (the "2009 Implementation Guidelines for Fire Management on Federal Lands"). Fires burning under this policy always are monitored closely, and suppression actions can be initiated if they begin to threaten human life, property, or significant resources. In the past, a given fire was designated as either a "wildfire" to be suppressed or a "wildland fire use fire" to be allowed to burn, and this same management was applied to the entire area within the fire perimeter. However, the "2009 Implementation Guidelines for Fire Management on Federal Lands" recognizes that some areas burning within a single fire perimeter are accomplishing management goals while other parts of the same fire threaten life or property or resources; thus it is appropriate to manage different parts of the fire differently. For example, a fire in Yellowstone National Park was ignited by a falling powerline in July, 2008. One flank of the fire was burning towards a Park housing facility, and this flank was suppressed and contained. However, the head of the fire was moving into a large wilderness area, and this portion of the fire was allowed to

burn. Thus, two different management approaches were applied to the same fire, resulting in both protection of vulnerable structures in one area and ecological benefits of fire in another area. This fire (the LeHardy fire) was viewed by many as a test case of the new "appropriate management response" policy, and it was deemed successful.

Allowing fires to burn under conditions in which they achieve resource management goals can be regarded as a form of passive restoration, even though agencies energetically monitor and continually re-evaluate such fires. However, allowing fires to burn does not entail actively modifying ecosystem structure in the way that mechanical thinning or planting do, and allowing fires to burn naturally represents removal of the stressor that is excessive active fire suppression. In addition to the ecological benefits of returning natural fire to fire-dependant ecosystems, there is an economic benefit: the cost of monitoring a "wildland fire use fire" usually are far less than the costs of actively suppressing the fire (Dale et al. 2005).

2. Innovative Management of Livestock Grazing in Mountain Grasslands

Some of the greatest ecological changes since the reference period have been seen in mountain grasslands of the South Central Highlands Section (Paulson and Baker 2006). Many of these changes are clearly undesirable: ecological composition, structure, and function have been profoundly altered, and overall biological diversity has been reduced. Many of these undesirable changes are a legacy of poor livestock grazing practices over the past century, including continuous grazing, overstocking, and overutilization.

Updated resource information, innovative new management practices, and new livestock grazing methods are available to help us manage and passively restore these grasslands for long-term viability and sustainability. Especially noteworthy and encouraging in this regard is the philosophy of harmonizing human and natural communities and the adaptive experiments now being conducted by the Quivira Coalition, headquartered in Santa Fe, New Mexico (White 2008).

New approaches and closely monitored experiments like those of the Quivira Coalition are urgently needed in many of the region's mountain grasslands. Unfortunately, simple removal of livestock from grasslands that are in poor ecological condition may not improve those conditions. When grazing pressure is reduced or removed from a site, the progression of the vegetation back to a previous condition depends on the extent to which soils, seed banks, and vegetative regeneration potential of remaining plants has been modified. It cannot be assumed that the pathway of succession following reduction in grazing intensity will be a simple reversal of the pathway of retrogression. In fact, some grasslands may require an active restoration component, e.g., planting of formerly dominant native bunchgrass species that have disappeared altogether since the reference period. Nevertheless, passive restoration in the form of good livestock management will also be required. In fact, certain kinds of grazing actually can enhance the process of ecological recovery (White 2008).

F. Opportunities: Active Restoration in Ponderosa Pine Forests

Perhaps the clearest opportunities and methods for active ecological restoration in the South Central Highlands Section are seen in ponderosa pine forests. As explained in Chapter II, many of the region's ponderosa pine forests differ substantially from reference conditions, with generally higher tree density and basal area, lower herbaceous ground cover, and more homogeneous structure in today's forests. These changes are generally regarded as undesirable, for they make the forests more vulnerable to high-severity fire and insect outbreaks, and biodiversity at both stand and landscape levels is probably lower than in the past. If the management approaches of the recent past are continued into the future, it is unlikely that conditions will improve in many ponderosa pine forests, since these past actions are in part responsible for the present condition. Similarly, if no action is taken, conditions probably will not improve for a long time. The dominant trees in many dense stands are all near the midpoint of their life spans and will not experience much natural thinning or mortality for several decades, and the deep organic litter layers and fuels will decompose only very slowly. In fact, with no action or a continuation of current management, there is a real possibility that an unusually large, uncontrollable fire or bark beetle outbreak will produce a disturbance event far outside the historical range of variability, thereby introducing additional undesirable ecological changes.

1. The Restoration Approach in Ponderosa Pine Forests

Table VII-5 summarizes some of the strategies that can be employed to restore a more open forest structure and reduce the risk of destructive fire and insect outbreaks in some ponderosa pine forests of the South Central Highlands Section. These techniques have been recently implemented on an experimental basis in the westside pine zone of the San Juan National Forest, and the results have been generally successful (Lynch et al. 2000, Berry et al. 2001, Romme et al. 2003). We also can learn much from the ongoing experiments in ponderosa pine forest restoration being conducted by the Ecological Restoration Institute at Northern Arizona University. For example, Wallin et al. (2004) recently reported variety of beneficial effects of restoration treatments on carbon, water, and nitrogen relations, as well as insect resistance, in the older (pre-1900) pines that the treatments were designed to preserve -- though responses were sensitive to the details of the restoration treatment (Skov et al. 2005). The basic approach involves mechanical thinning to reduce tree density and basal area and to restore a clumped pattern of tree dispersion in the stand, followed by a low-severity prescribed fire. Monitoring is implemented before treatment, repeated just after treatment, and then continued for several years to detect short-term and long-term effects.

Table VII-5. Summary of major changes in ponderosa pine forest ecosystems of the South Central Highlands Section, and opportunities and challenges for restoration of desired ecological components and processes that have been lost.

Component or Process	Reference Conditions	Current Conditions	Restoration Opportunities	Pitfalls & Cautions
Predominant canopy structure	- low density, diversity of tree size and age classes, in most stands	- high density, little diversity of size and age classes, few large trees, in most stands	- restore diversity of tree sizes by thinning mostly small trees, with retention of larger trees and some smaller trees in a clumped pattern	- not all stands had open structure during reference period; need to retain some dense stands also ... risk of dwarf mistletoe spread
Ground layer vegetation	- diverse herb stratum co-dominant with shrubs & tree saplings	- depauperate herb layer; some species (e.g., Arizona fescue, mountain muhly) greatly reduced or locally extirpated	- stimulate herbaceous growth and diversity through canopy thinning, grazing control, and prescribed fire	- risk of alien weeds increasing or invading burned areas ... severely degraded communities may require planting
Fire regime	- frequent, low-intensity surface fire, causing little canopy mortality	- infrequent, high-intensity crown fire, destroying the canopy	- restore frequent, low-intensity fire regime through prescribed fire coupled with thinning and fuel reduction	- disproportionate mortality of remaining old-growth trees ... potential nutrient loss & reduced site productivity
Organic litter layer on the forest floor and overall fuel loads	- relatively shallow litter and low fuels in most areas, due to frequent fire, but patches of locally high fuels and deep litter	- continuous, relatively deep litter layer, and moderate to high fuel loads in most areas	- reduce litter layers and fuels with prescribed fire coupled with thinning	- a single prescribed burn unlikely to be adequate; requires long-term burning program
Vertebrate communities	- apparently diverse; poorly documented	- probably less diverse, lower densities of species dependent on large trees, snags, & logs	- restore large trees and snags through thinning of smaller trees and prescribed fire	- some species (e.g., deer, elk, turkey) are thriving in current stand conditions
Soil organisms and soil processes	- poorly understood	- probably altered by altered stand structure & fire regimes	- restore more heterogeneous stand structure and fire regime	- lack of information ... hard to assess significance

Mechanical Thinning

To restore a more open canopy containing greater diversity of tree sizes (including large trees), we can employ selective harvesting to remove predominantly smaller trees from a stand that is overly dense and homogeneous. The approach, recently developed in the westside pine zone project (Lynch et al. 2000, Romme et al. 2003), is to begin by identifying and marking clumps of the larger trees in the stand that resemble the clumps that existed in the forest prior to high-grade logging. There are no rigid criteria for identifying these clumps, because they apparently varied in size and shape even in the forests of the reference period. Stumps of the pre-1900 trees, if present, may give some clues to the former clumping patterns, or it may be necessary to rely on general familiarity with the pre-1900 forest structure. After marking the clumps, the loggers then remove most of the smaller trees plus a few larger trees from the spaces between clumps, and leave the trees of all sizes within the clumps. Total tree density and basal area are reduced to something a little higher than estimated pre-1900 density and basal area (see Chapter II) -- the extra is provided to compensate for subsequent tree mortality resulting from windthrow after logging, or from prescribed fire (below).

Prescribed Fire

Selective logging is followed by prescribed fire, to reduce slash and litter, top-kill the shrubs, expose mineral soil for pine seedling establishment, and stimulate sprouting and growth of suppressed herbaceous plants. It may be appropriate in some stands to burn before thinning and then again afterwards, or to burn twice within a few years after the thinning operations, to significantly reduce organic litter and woody debris. Harrington (1987) recommends midsummer burning at 2-year intervals initially, to reduce Gambel oak cover and density and create mineral seed beds for establishment of new ponderosa pine seedlings. Harrington (1981) and others have offered burning prescriptions for reducing fuel loads in southwestern ponderosa pine forests.

Once the stand has been thinned and the fuel loads reduced in this way, it will be necessary to continue prescribed burning into the indefinite future—otherwise the undesirable stand conditions will re-develop over the next several decades. Two general approaches to designing a prescribed fire program are as follows. First, managers could schedule average intervals between fires, the variability in fire intervals, and the seasonality of burning, to mimic the pre-1870 fire regime of the area (Chapter II). Such an approach is problematic, however, for at least three reasons. First, it is important to emphasize that fire interval data are strongly influenced by details of sampling design, sampling intensity, and size of study area. For example, more historical fires will be detected in a larger study area and with a greater number of fire-scar samples than in a smaller study area or with a smaller sample size. Consequently, the average fire interval computed for an area of interest will be shorter with the former study design and longer with the latter. The data presented in Chapter II are for study areas of ca. 50 ha with intensive sampling; but the numbers would be different for a larger or smaller study area. The second problem with using the

numbers in Chapter II as an explicit guide to the appropriate frequency of prescribed burned is that historical fires did not burn uniformly; e.g., a 10-year fire interval does not necessarily mean that the fire at the beginning and at the end of that interval both burned every square meter of a stand. Composite fire intervals, of the kind reported in Chapter II, reflect fires that occurred somewhere within a stand, but do not tell us how much of the stand actually burned. Thus, the interval between fires at a single point on the ground (a statistic that is extremely difficult to calculate) actually is several times longer than the composite fire interval for the stand as a whole (Baker and Ehle 2001). Finally, attempting to replicate exactly the fire frequencies of the 1600s - 1800s is problematic because the climate and the overall ecological context (e.g., presence of invasive non-native plants) will be different in the 21st century.

Because of the problems associated with attempting to exactly replicate the reference period fire regime (described in the previous paragraph), a second approach to designing a prescribed fire program for restoration of ponderosa pine forests would entail a less precise and more flexible protocol for burning (e.g., Allen et al. 2002, Paulson and Baker 2006). Prescribed burning would be conducted somewhat opportunistically, whenever weather and fuel conditions and availability of personnel are become suitable for conducting burns. The reference period fire intervals in Chapter II would be used only as a very general guide, not as an explicit prescription. For example, the data in these tables indicate that prescribed fires could be somewhat less frequent at higher elevations. The key idea underlying this approach is that

prescribed burning (perhaps coupled with selective logging) should set the ecosystem on a trajectory towards a more sustainable state. That state would resemble the pre-1870 state (e.g., lower tree density than today, periodic low-severity fire), but would not exactly replicate the pre-1870 state. Gradually the system would develop new characteristics that are similar to the 1870 characteristics but also congruent with 21st century climate and environmental context.

In any event, it is critical to maintain a natural range of variability in fire intervals, intensities, and seasons. For example, some intervals between successive fires in a stand probably should be as short as 2 - 3 years and as long as 15 - 20 years. There should be some summer burns as well as spring and fall burns. Two closely spaced fires during the growing season will knock back the shrubs, consume much of the litter layer, and create good seed beds for pine seedlings and native herbaceous plants. (Such fires may kill a few canopy trees also, but the resulting snags may benefit some animals.) During long intervals between fires, newly germinated pine seedlings and herbs will be able to grow large enough to survive the next fire. In any prescribed fire, there should be portions of a stand that burn relatively hot, others that burn only lightly, and still other places that do not burn at all. The shrubs are expected to be relatively resilient to fires of almost any frequency and intensity (within the historic range of variation) because most re-sprout readily from extensive root systems. Eventually it may be possible to allow most lightning-ignited fires to burn in treated forests, rather than relying entirely on manager-ignited fires.

In a recent analysis of the 2002 Rodeo and Chediski fires in Arizona ponderosa pine forests, Finney et al. (2005) found that fire severity was generally reduced in areas that had been prescribed burned within nine years prior to the 2000 fires -- even during the extreme fire weather that characterized the Rodeo-Chediski fire complex. Similarly, the 2002 Hayman fire in ponderosa pine forests of Colorado's Front Range essentially stopped spreading when it reached areas that had burned within the previous 12 months (Finney et al. 2003), although older prescribed burns in the Hayman area had substantially less effect on fire spread and severity.

These studies by Finney et al. (2003, 2005) demonstrate the potential effectiveness of prescribed burning in reducing the extent and severity of wildfires -- even when wildfires occur during extreme fire weather conditions. Nevertheless, a single prescribed fire probably will not be sufficient to reduce abnormal fuel loads and litter layers or to re-invigorate the long suppressed herbaceous community in some stands. A single prescribed fire did increase herbaceous cover and richness in the pilot studies conducted in ponderosa pine forests near Dolores, CO, but did not reduce woody fuel loads on the forest floor (Lynch et al. 2000, Romme et al. 2003). Abrahamson and Abrahamson (1996) examined responses to prescribed burning in pine forests of northern Florida, where fire history and community structure are similar in many respects to ponderosa pine forests of the South Central Highlands Section. Some of the stands were long unburned (>35 years) but others had burned relatively recently (<20 years before the prescribed burn of the current study). In all stands, the prescribed fire stimulated the sprouting and growth of long-lived woody species. However, the single prescribed fire did not re-invigorate the growth of suppressed herbaceous plants in the long unburned stands, though it did stimulate at least some of the herbs in the recently burned stands. The authors concluded that unnaturally long periods without fire now threaten the persistence of several fire-adapted species in Florida pine forests, and that a single prescribed fire was inadequate to recover those species. They recommended a fire management program that would mimic the historic fire regime in terms of variability in fire intervals, intensities, and seasons -- much like we recommend for ponderosa pine forests of the South Central Highlands Section.

In conjunction with the prescribed burning program, it is necessary to protect recently treated stands from excessive livestock grazing. Both domestic and native ungulates will be attracted by the increased quantity and quality of forage in thinned and burned forests, and potentially may wipe out the gains in the herbaceous vegetation that have been obtained through treatment. It also is critical that invasive, non-native plants be dealt with, as described below.

2. Potential Pitfalls and Hazards of Active Restoration of Ponderosa Pine Forests

Several important caveats and considerations must be dealt with before adopting a strategy like that described above at a landscape scale (Table VII-5). These include the potential hazards of prescribed fire, economic issues, weeds, and mistletoe.

Hazards of Prescribed Fire:

There are numerous potential pitfalls associated with a prescribed burning program of the magnitude called for above. For one thing, prescribed fire has the potential to actually hasten the demise of the few old-growth ponderosa pine that still remain in dense stands. Following experimental burns in Crater Lake National Park, Oregon, fire-induced mortality occurred up to several years after the fire, as weakened trees succumbed to bark beetles, other pathogens, and drought (Swezy and Agee 1991). Mortality of large, old ponderosa pine trees was significantly greater in the burned areas than in unburned areas, and where early spring burning had been conducted, the mortality of these trees was > 30%. One reason for high mortality with early-season burns was that fine root biomass near the soil surface was especially high during this season, and fire-caused loss of fine roots weakened the trees and made them susceptible to water stress and pathogens. Raking away the litter around the bases of the large old trees before the burn is a technique that has been used in northern Arizona to reduce injury to old-growth trees (W. Covington, personal communication). However, this technique did not reduce mortality in the Crater Lake study, possibly because this removal of the litter allowed the duff to dry and then burn more thoroughly than usual, generating a high heat flux into the fine root zone in the upper soil layers. Swezy and Agee (1991) concluded that current fuel loads in some ponderosa pine stands may be too great to allow spring burning (when the trees appear to be most susceptible to injury), and

suggested that late-season burning be coupled with mechanical fuel reduction to reduce damage to the old-growth trees.

Prescribed fire also may cause significant nutrient loss, especially of nitrogen. The immediate, short-term effect of burning usually is an increase in available soil nitrogen (Kovacic et al. 1986, Monleon et al. 1997). However, over the next several years or decades, there may be changes in mineralization rate, quality of soil organic matter, and soil microbial activity, that result in reduced amounts of total nitrogen in the soil (Klemmedson 1976, Vance and Henderson 1984, Monleon et al. 1997, Kaye and Hart 1998, Neary et al. 1999, Tiedemann et al. 2000). In ponderosa pine forests in central Oregon, stands burned 4 months previously had increased soil inorganic nitrogen concentration, but there was reduced soil nitrogen concentration and net nitrogen mineralization in stands burned 5 and 12 years previously (Monleon et al. 1997). Loss of nitrogen, due to volatilization during the fire and then reduced substrate quality and quantity for several years after the fire, may lead to long-term decreases in site productivity and tree growth -- especially on nutrient poor volcanic soils like those in central Oregon. Many of the ponderosa pine forests in the South Central Highlands Section are growing on sedimentary substrates that may be inherently more fertile than the poor soils described in the Oregon study, and therefore the forests in our area may be less susceptible to nutrient loss and reduced site productivity after prescribed fire. However, no research has been conducted to address this question in the South Central Highlands, so managers should be aware of the possibility of

significant nutrient changes over the long term under the prescribed fire program recommended above. Another potential and very undesirable effect of prescribed burning relates to alien weeds, as discussed below.

A final concern related to prescribed burning is the risk of property damage and liability on private lands adjacent to or interspersed within public lands. The current trend of exurban development in many parts of the South Central Highlands Section is resulting in large numbers of expensive homes within fire-adapted ecosystems like ponderosa pine (Romme 1997, Theobald 2000, Theobald and Romme 2007). Many of the homes are extremely vulnerable to destruction by fire and cannot be easily defended from fire (Cohen 2000, Cohen and Stratton 2003). It may be tempting to conduct fuel reduction and prescribed burning programs in remote areas far from exurban development, to reduce these liability risks -- yet the areas around homes are some of the places most in need of treatment to reduce the hazards of uncontrollable wildfire. Education of new home buyers is needed, as is cooperation with local home-owners associations to reduce fire risk. The Falls Creek Ranch homeowners association, north of Durango, recently implemented a prescribed fire / fuels reduction program, in consultation with the San Juan National Forest and the Colorado State Forest Service. This collaboration could serve as a model for future cooperative efforts between federal land management agencies and private land owners to manage fire and reduce fire risks.

Despite all of these potential pitfalls, the prescribed fire program is a critical component of the overall restoration effort. Logging alone will not reduce fuel loads, create mineral soil and reduce shrub competition for new pine seedlings, or stimulate the growth of suppressed herbaceous plants. Grazing must be controlled also, especially in the first few years after treatment, because livestock will be attracted to the increased production and quality of forage in the burned areas, and may counteract the positive effects of burning on the herb layer.

Issues related to Economics, Weeds, and Mistletoe

Thinning operations may be difficult to implement in some areas because of economic factors. An economic assessment of three pilot projects conducted according to the prescription described above in the westside pine zone of the San Juan National Forest in 1995 and 1996 found that the loggers generally lost money on the smaller trees (products other than logs -- POL) because of poor markets, low financial value, and the cost of hauling the material to a distant mill that was willing to accept it (Lynch and Jones 1996, Lynch et al. 2000, Romme et al. 2003). However, they did make money on the sawlogs that were included in the harvest prescription. It was concluded that of the material cut in a stand (i.e., the trees *outside* the clumps being retained for restoration purposes), approximately 40% needed to be of sawtimber size for the operation to break even financially. They also noted that this percentage would vary depending on current market conditions and fuel costs. Nevertheless, this study indicates that the best candidates for restoration of open, diverse stand structure generally are stands that still have at least a small

component of larger trees. Stands having no or almost no sawtimber-size trees (which may include many that are in the greatest need of restoration) cannot be treated in this manner except in a below-cost mode of operation. If local timber industries are able to develop new technologies and markets to make better use of the smaller diameter material, then this prescription may be more broadly applicable. Development of such markets and technologies should be seen as a major challenge (and a potential opportunity) for the timber industry.

A second caution about thinning and prescribed fire has to do with the potential invasion of non-native plants following treatment. The conditions created by these treatments (increased light, nutrients, and bare mineral soil) are ideal for establishment of weedy species like cheatgrass (*Anisantha* (formerly *Bromus) tectorum*), Canada thistle (*Breea* (formerly *Cirsium) arvense*), muskthistle (*Carduus nutans*), and oxeye daisy (*Chrysanthemum leucanthemum*). These species are widely distributed, especially along roads and trails and in meadows and disturbed areas. If a stand is opened up and sources of these invasive species are close by, the weeds can become abundant in the area via sprouting from rhizomes (e.g., Canada thistle and oxeye daisy) or seed dispersal (e.g., cheatgrass and muskthistle). These aggressive species may dominate the stand for many decades, to the detriment of the native understory flora. Therefore, it is important to identify local weed populations and reduce or eradicate them before thinning or burning a stand. Discussions of restoration of ponderosa pine forests often focus on restoration of the canopy—but restoration (or protection) of the understory is equally important. Indeed, Keeley (2006) warns that the improvements gained through ecological restoration and wildfire mitigation, via thinning and prescribed burning, may be largely nullified by post-treatment invasion of non-native weed species. He concludes that treatments are necessary in many places, because the current forest conditions are unsustainable -- but we must not naively assume that all we need to do is thin forest canopies and re-introduce fire to achieve sustainability.

A third general caution involves dwarf mistletoe, a parasitic plant that usually does not kill its host tree directly but weakens the tree and distorts its growth (Hawksworth and Wiens 1995). This native plant generally is not a major threat to ecological integrity, but it can be a serious problem in areas devoted to wood fiber production. At present, mistletoe is not a big problem in the pine zone on the west side of the San Juan National Forest (Angwin and Raimo 1994:16). Some 2,100 acres of the study area (3% of the lands that were inventoried) received a moderate or high dwarf mistletoe rating; 22,000 acres (28%) were rated as low; and 55,000 acres (69%) had no mistletoe evident. However, where mistletoe is already present, thinning and prescribed fire can facilitate its spread into previously uninfected trees. Therefore, mistletoe needs to be explicitly considered in designing specific restoration treatments for ponderosa pine stands.

Additional Considerations:

We cannot be certain that the treatment program outlined above will necessarily accomplish all of our objectives. For example, even if we

create good seed beds by means of thinning and prescribed fire, a new cohort of pine seedlings will not become established if there is not an adequate seed crop produced soon after treatment or if severe drought occurs during the first few years (Brown and Wu 2005). The herbaceous community in many stands has been so severely degraded by past grazing and competition with woody plants that it may not recover simply from improved soil and light conditions. Many formerly important herbaceous species may have been locally extirpated, and will require planting or seeding from nearby surviving populations if they are to become reestablished.

Given these and other considerations, it is important to acknowledge that the ponderosa pine restoration forest program outlined above is still in its formative stages. Its ultimate effectiveness and unexpected side-effects are not yet known. Therefore, careful monitoring needs to be conducted to document successes and detect any undesirable outcomes or surprises early on. In effect, restoration of ponderosa pine forests is an experiment, and it should be linked to a research program built around explicit, testable hypotheses and questions. The results of the associated research will aid in assessing and modifying the treatment techniques, as well as provide valuable insights into the functioning of ponderosa pine forest ecosystems (Jordan et al. 1987).

Restoration of an open stand structure should be applied only to some, not all, ponderosa pine forests throughout the South Central Highlands Section. During the reference period, there apparently were some dense stands of small ponderosa pine trees, similar to the stands that dominate the landscape today (Shinneman and Baker 1997, Ehle and Baker 2003). These dense stands provide excellent habitat for many highly desirable species, e.g., elk, deer, and turkey, and so they should not be eliminated altogether. The main problem to be addressed through restoration is not that the dense, homogeneous stands we have today are undesirable in all respects, but that an excessive proportion of the landscape appears to be covered by stands of this kind -- a far higher proportion than occurred during the reference period.

Restoration efforts should be concentrated, at least initially, in areas that already have extensive road systems and a long history of logging. The few remaining roadless areas and tracts of unlogged ponderosa pine forest are not "pristine" -- they have the same legacies of early grazing and fire exclusion that characterize nearly all ponderosa pine forests in the region, and they may benefit in some ways from programs of thinning and prescribed fire. However, by virtue of their lack of roads and lack of logging history, they still retain certain characteristics of the reference period forests -- characteristics that are now gone from more heavily exploited stands, e.g., complete age structures in the tree populations and absence or very low abundance of alien plant species. These special, relict areas should not be subjected to experimental restoration efforts until the restoration techniques have been tested and improved in other areas already subjected to intensive human alterations.

A final consideration before embarking on an extensive program of ponderosa pine forest restoration is how these restored stands will be managed and used over the very long term (Allen

et al. 2002). The initial logging treatment can be done in one year, and a prescribed burning regimen can be established within a decade (then to be continued indefinitely). What next? What are the management objectives over the next century or two? Within several decades after initial treatment, the objectives of restoring large trees and snags, a re-invigorated herb layer, and a more appropriate fire regime, may be accomplished. At that time it may be desirable to enter the stands again to remove some of the new wood fiber that has been produced. However, it is not clear at this time just what kind of future timber management strategy should be employed in the restored ponderosa pine forests. It obviously is premature to try to make definitive plans, since so much is still unknown about the future trajectories of stand development in treated stands. Nevertheless, some general thinking and planning for the distant future may help clarify our specific goals for the present management of ponderosa pine forests in the South Central Highlands Section.

G. Opportunities: Emulating Natural Disturbance Processes in High-Elevation Forest Landscapes

In many places the management emphasis is not on ecological restoration per se, but on sustainably harvesting an economic product in such a way that ecosystem services are not impaired or degraded. New and innovative approaches to silviculture are being developed in the Pacific Northwest, Canada, and elsewhere to address this goal. These approaches are based on mimicking the kinds of natural disturbances that have shaped these forests for hundreds or thousands of years (Kohm and Franklin 1997, Romme et al. 2000, Perera et al. 2004). The idea is that, because the native biota is already adapted to these kinds of disturbances, anthropogenic disturbances of similar kind and intensity are less likely to produce unexpected or undesirable ecological surprises than are human disturbances that do not resemble the natural disturbance regime.

The biota of the South Central Highlands Section are well adapted to the wide range of natural disturbances that have affected them during their evolutionary history, e.g., fires, insect outbreaks, wind storms, and climatic variability, as described in the chapters above. However, some recent anthropogenic disturbances probably are unprecedented; i.e., they are different in type and/or intensity from any of the natural disturbances to which the organisms are adapted. Therefore, a coarse-filter strategy for reducing the impacts of anthropogenic disturbance is to design the human-caused disturbances to mimic natural disturbances to the greatest extent possible (Romme et al. 2000, Fischer et al. 2006). Two types of forest in the South Central Highlands Section that appear especially well suited to this idea of silviculture that emulates natural disturbance processes are high-elevation spruce-fir and cool moist mixed conifer forests.

Strategies for making anthropogenic disturbances mimic more closely the natural disturbance regime in high-elevation landscapes involve three primary considerations: (1) revising the size, shape, and spatial arrangement of logging units, (2) providing biological legacies of the former stand in logged

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areas, and (3) minimizing roads (Table VII-6).

Design of logging units since the 1970s has emphasized small cutting units widely dispersed across the landscape. The rationale has been reduction of aesthetic impacts of clear-cuts, enhancement of tree regeneration, and buffering of undesirable effects such as soil erosion and stream sedimentation. However, recent empirical and theoretical research has demonstrated clearly that this strategy maximizes fragmentation of mature and old-growth forest (Franklin and Forman 1987, Li et al. 1993, Forman 1995, Franklin et al. 1997). Moreover, the natural fire regime was one of infrequent but large disturbances rather than frequent, small, spatially dispersed burns. Native biota have numerous adaptations enabling them to become reestablished readily in large, as well as, small burned patches (Turner et al. 1997). Many of the problems of poor regeneration and erosion in large clear-cuts are related to the lack of biological legacies in clear-cuts rather than the initial disturbance itself (see below). Therefore, many ecologists are now recommending that logging units should be larger in size, and should be aggregated within a small portion of the landscape with patch sizes and shapes that resemble fire-created patches (Li et al. 1993, Crow and Gustafson 1997, Franklin et al. 1997). This design minimizes fragmentation of mature forest and assures at least some large patches of mature forest. It also mimics the natural disturbance regime with respect to spatial patterns of disturbance. This recommendation does not imply that a greater total area should be logged; rather it deals with the optimal spatial patterning of cutting units given the target for timber production that has been established through the broader planning process.

Recently logged areas in subalpine forests of the central Rocky Mountains usually bear little resemblance to areas of recent fires or insect outbreaks. Most conspicuous is the paucity of large standing and fallen dead trees in logged areas (Spies et al. 1988, Hutto 2008, Wei et al. 1997). These and other "biological legacies" are critical structural elements for numerous species and ecological processes involved in recovering from the disturbance (Franklin et al. 1997, Kaila et al. 1997, Spies 1997). Therefore, Franklin et al. (1997) and others are now recommending variable retention harvest systems, based on retaining structural elements of a harvested stand (e.g., large trees and snags) for at least the next rotation, to achieve three objectives: (i) "lifeboating" species and processes immediately after logging, (ii) "enriching" the new forest stand with structures that otherwise would be lacking, and (iii) "enhancing connectivity" between patches of unlogged forest habitat (Franklin et al. 1997;115). An additional technique that has been suggested, but not yet much studied or experimented with, is to follow a variable retention harvest with an extensive prescribed fire (not just burning of piles of slash) to rapidly decompose some of the smaller-sized organic matter, create blackened or mineral soil surfaces, and temporarily reduce herbaceous cover. These kinds of fire effects may be as important in natural forest recovery processes as the maintaining of biological legacies, but they obviously are not produced by a variable retention harvest alone.

Table VII-6. Opportunities for mitigating the adverse effects of past and present human activity by emulating natural disturbance processes in high-elevation forested landscapes of the South Central Highlands Section.

Objective	Possible Mitigating Action(s)	Ecological Rationale
Minimize fragmentation of mature forest by timber cutting operations	Cut timber in a few large patches rather than in many small patches, disturbing the same total area but with different spatial pattern	Most of the area disturbed by fire in pre-1860 period was burned in a few large fires rather than many small fires
Minimize fragmentation of mature forest by roads	Minimize new road building; close existing roads wherever possible; implement logging techniques that do not require roads, if possible	Roads are anthropogenic landscape features that have no ecological precedence, but have negative effects on native biota
Provide heterogeneous stand structure within logged areas and softer boundaries between cutting units and uncut forest	Design logging units for less uniform cutting intensity, e.g., mix partial cuts & clear-cuts, leave patches of uncut or partially cut forest within and along edges of clear-cuts	Fire severity in large fires tends to be patchy, and creates a heterogeneous post-fire environment
Provide large snags, coarse woody debris, and other biological legacies of the previous stand in logged areas	Leave all large snags and fallen logs within logged areas, as well as some large & small canopy trees that will eventually become snags & fallen logs	Fires leave great quantities of snags & coarse woody debris in all size classes; this material is important in post-fire succession, wildlife habitat, and long-term soil development
Prevent the spread of alien plant species along roads and in logged areas	Avoid planting abandoned roads & logged areas, or plant with native genotypes; minimize road building	Native plant species are capable of rapidly re-vegetating bare soil areas created by fire, but the natives may be less competitive than the aliens in roads and logged areas
Restore early successional communities threatened by fire suppression	Introduce prescribed fire on a landscape scale	Young stands currently developing after logging lack key structural components of recently burned habitats, so are not adequate replacement
Provide old-growth forest stands on productive sites as well as less productive sites	retain existing old-growth & restore previously cut stands on productive sites	Pre-1860 landscape mosaic contained some old-growth on all kinds of sites, not just unproductive

Another important biological legacy that is often missing after conventional forest harvest operations is coarse wood (dead material larger than about 7.5 cm (3 in) in diameter; it is sometimes referred to as "coarse woody debris" but we prefer the less pejorative term "coarse wood"). Dillon et al. (2003) have discussed the importance of this material in forests of the Southern Rocky Mountains:

-- "Natural disturbances do not remove large pieces of wood from forests. Even after an intensive fire, most of the wood remains in the form of dead standing trees (snags), which becomes [coarse wood] within two or three decades … The leaves, branches, and smaller wood are consumed by fires, but Tinker and Knight (2000) found that only about 8% of the wood >7.5 cm was burned after an intense fire in Yellowstone National Park [YNP]. Other estimates for the proportion of CWD consumed by canopy fires range from 12 to 65% (Tinker 1999). With time, the downed wood becomes incorporated into the forest floor … creating the impression that many 100+-yr-old stands have very little [coarse wood]. However, raking the forest floor reveals the remains of many decomposing logs. New [coarse wood] is added as the larger trees die and fall, one by one during stand development or in large numbers during a windstorm or after the next fire.

-- "Tinker and Knight (2000), using YNP and Medicine Bow National Forest data, also concluded that the amount of CWD remaining after several clearcuts is less than after the same number of fires on comparable sites. Moreover, and as might be expected, they found that the time required for [coarse wood] to cover 100% of the ground surface area after a fire was much less than after a clearcut, and that this time interval became similar for a clearcut and fire only when slash amounts left on the ground were doubled from average amounts left using current management practices (in other words, 200% of current [coarse wood] amounts in slash could keep practices within the HRV on some sites)."

Finally, attention should be given to reducing the extent and impact of roads, which have little or no precedent in the evolutionary history of the native biota, and which strongly facilitate the fragmentation of interior forest habitats, dispersal of alien species, and human-caused disturbance of wildlife. Some roads obviously are necessary for management purposes, but the current road system in the national forests -- containing more miles than the interstate highway system -- probably is more extensive than is really needed (Riebesame 1997). Another advantage of logging large, aggregated patches rather than small dispersed patches is that the former spatial pattern requires a less extensive road system. Necessary roads also can be placed in areas that minimize their fragmenting effects, e.g., running around a large patch of mature forest rather than bisecting it (Reed et al. 1996a), and unnecessary roads can be permanently closed. Implementing logging techniques that do not require roads and skid trails -- e.g., removing logs with horses in winter -- should also be considered. Economic limitations may preclude such techniques in many areas, but they should be explored and used where possible to reduce the heavy impact of roads and mechanized equipment in forest ecosystems.

In addition to modifying timber production strategies to minimize the adverse effects of anthropogenic fragmentation, managers should re-introduce fire as a natural process in high-elevation forests of the central Rocky Mountains. Some of the ecological effects of fire can never be entirely simulated by logging, even with variable retention techniques, and some native species like the black-backed woodpecker are almost completely dependent upon recently burned subalpine forests (Hutto 2008). Manager-ignited fire probably is not appropriate in most high-elevation forests, but many lightning-ignited fires can be allowed to burn without interference in order to provide unique early successional habitats that cannot be produced in any other way.

H. Summary

Many of the ecosystems of the South Central Highlands Section today are substantially different from the ecosystems that existed during the reference period, and many of these changes are regarded as undesirable. In this chapter we first review some of the major challenges facing land managers in the South Central Highlands Section, and then explore some of the opportunities for mitigation and restoration of damaged or degraded components of the ecosystems in the region.

A. **Challenges: Forest Fragmentation by Roads & Logging**: One of the major global challenges to biodiversity and ecological integrity is fragmentation of wildland habitats by roads, logging, agriculture, home building, and other human developments. Roads are unprecedented features in ecological history, and have greater impacts on plants, wildlife, and natural ecological processes than almost any other feature of the human-influenced landscape. Road densities are surprisingly high throughout the South Central Highlands Section. For example ponderosa pine forests in the San Juan National Forest in southwestern Colorado contain 2.60 miles of road per square mile.

To evaluate quantitatively the effects of road-building and logging on high-elevation forests, we reconstructed logging history in the Pagosa District of the San Juan National Forest. Although logging began in the late 1800s in most of the ponderosa pine forests of the region, very little logging occurred in high-elevation forests of this district prior to 1950. The number of acres affected by logging each year increased through the 1950s and 1960s, reached a peak in the 1970s, and then declined through the 1980s and 1990s. This half-century of logging has directly affected only a small proportion (ca. 5 %) of the total spruce-fir forest acreage on the Pagosa District. However, nearly all (85%) of the suitable timber base in spruce-fir has been entered at least once for timber removal. We also measured changes in several metrics of landscape structure between 1950 and 1992 using the program FRAGSTATS. For all 45 landscape metrics reported by FRAGSTATS, the average percentage change from 1950 - 1993 was 15 - 25 % when roads were included, but only 3 - 8 % when roads were excluded. The number of distinct patches has increased,

average patch size has decreased, edge density (meters of edge between two different cover types) has increased, and mean core area (patch area >50 m from an edge) has decreased

B. Challenges: Outdoor Recreation, Roads, and Exurban Development: Outdoor recreation is now the principal use of most forests in the southern Rocky Mountains. Although often regarded as a benign activity that has little or no adverse impact on ecological conditions or processes, outdoor recreation actually poses a major threat to biodiversity and ecological integrity. Of all species federally listed as endangered or threatened, outdoor recreation affects more of these species than any other activity except water development. Outdoor recreation affects plant and wildlife species via direct harvest (hunting, fishing, and collecting), disturbance (either accidental or intentional), habitat modification, and pollution.

The human population in the Rocky Mountain region is one of the fastest growing in the nation, and much of the new development is occurring in formerly rural or undeveloped areas. Exurban development poses several important ecological and social challenges in the South Central Highlands Section. First, the houses and access roads fragment natural habitats and displace sensitive wildlife. Second, exurban development often occurs in ponderosa pine forests and other vegetation types where fire was formerly an important ecological process. Not only are these houses vulnerable to loss in wildfire, but restoration of a natural fire regime is severely constrained by the presence of vulnerable structures that must be protected from fire. Finally, the

costs to local governments of maintaining roads and providing services (fire protection, police, and schools) to these scattered residents usually exceed the revenues generated by property taxes; in effect, residents in town are subsidizing the exurban life style. The ecological and social costs of exurban development can be reduced in a number of ways, e.g., by clustering developments and protecting agricultural land through the use of conservation easements.

C. Challenges: Climate Change: The earth's climate is changing, global temperatures are increasing, shifts in precipitation patterns are likely to become pronounced in the next century, and weather related disturbances (e.g., hurricanes and droughts) are likely become more frequent and severe. The potential for large, severe fire also is increasing. An upsurge in the frequency of large fires began in the mid-1980s and is expected to continue. Climate change complicates all of our efforts in land management.

D. Opportunities: Principles of Ecological Restoration: One of the major land management activities in the twenty-first century likely will be restoration of ecosystems that were degraded during the past century. *Passive restoration* involves removal of the stressors that have degraded an ecosystem, e.g., regulating livestock grazing and recreational activity, and allowing lightning-ignited fires to burn without interference. *Active restoration* involves conscious modification of an ecosystem to achieve a desired result, e.g., selective logging and manager-ignited prescribed fire. Active restoration is not needed (nor is it feasible) in every location throughout

the South Central Highlands Section. An important component of all restoration programs is protecting and maintaining those areas that have been little altered, to serve as reference areas for restoration treatments in degraded areas.

E. Opportunities: Passive Restoration: Two major stressors in the South Central Highlands Section are excessive fire suppression and poorly conducted livestock grazing. Land management agencies now have opportunities to allow fires or portions of fires to burn naturally with minimal human interference (2009 Implementation Guidelines for Fire Management on Federal Lands). Innovative new grazing management practices, such as those being conducted by the Quivira Coalition, headquartered in Santa Fe, New Mexico, are available to help manage and passively restore these grasslands for long-term viability and sustainability.

F. Opportunities: Active Restoration of Ponderosa Pine Forests: Perhaps the clearest opportunities and methods for active ecological restoration in the South Central Highlands Section are seen in ponderosa pine forests. The basic approach involves mechanical thinning to reduce tree density and basal area and to restore a clumped pattern of tree dispersion in the stand, followed by a low-severity prescribed fire. In conjunction with the prescribed burning program, it is necessary to protect recently treated stands from excessive livestock grazing and to mitigate invasive, non-native plants. Monitoring is implemented before treatment, repeated just after treatment, and then continued for several years to detect short-term and long-term effects. It is important to emphasize that restoration of an open stand structure should be applied only to some, not all, ponderosa pine forests throughout the South Central Highlands Section.

G. Opportunities: Emulating Natural Disturbance Processes in High-Elevation Forest Landscapes: Where the management emphasis is not on ecological restoration per se, but on sustainably harvesting an economic product in such a way that ecosystem services are not impaired or degraded, innovative new approaches to silviculture are being developed, based on mimicking the kinds of natural disturbances that have shaped these forests for hundreds or thousands of years. Because the native biota are already adapted to these kinds of disturbances, anthropogenic disturbances of similar kind and intensity are less likely to produce unexpected or undesirable ecological surprises than are human disturbances that do not resemble the natural disturbance regime. Possible strategies for making anthropogenic disturbances mimic more closely the natural disturbance regime in high-elevation landscapes involve: (i) revising the size, shape, and spatial arrangement of logging units, (ii) providing "biological legacies" of the former stand in logged areas by means of variable retention harvest systems and retaining coarse wood, and (iii) reducing the extent and impact of roads.

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