HISTORICAL RANGE OF VARIABILITY ASSESSMENT FOR FOREST VEGETATION OF THE NATIONAL FORESTS OF THE COLORADO FRONT RANGE

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PREFACE AND ACKNOWLEDGMENTS

This report is a response to the widespread recognition that forest resource planning and decision-making would benefit from a series of assessments of the historical range of variability (HRV) of the ecosystems which comprise the National Forest lands of the Rocky Mountain region. Although the work on the current report formally began under a Cooperative Agreement between the Regional Office of the Forest Service and the University of Colorado in 1999, prior events contributed significantly to this report. Most significantly, the staff of several National Forests in the region (e.g. Rio Grande, White River, Routt) conducted their own assessments of historic range of variability of Forest lands in the mid-1990s. These documents provided useful starting points, and were critically important in initiating the still evolving process of how to conduct the assessment of HRV in Region 2. In the late 1990s, the senior author of the current report participated in an external review of several of the HRV reports produced by the Forest Service staff. This external review team included among others the leaders of the current round of HRV reports in the Region (Dennis Knight of University of Wyoming, William Romme of Colorado State University, and Thomas Veblen of the University of Colorado). While lauding many aspects of the initial HRV reports, this external review team made a number of suggestions for their improvements. In particular, the external team raised questions about the validity of conclusions that were based exclusively on professional judgements or hypotheses that had not been examined through any systematic research process. The team stressed the need for greater reliance on peer-reviewed research publications, and more critical evaluation of some of the widespread perceptions about changes in these forest ecosystems which had been assumed to be related to past resource management practices. That initial experience as a critic of HRV reports hopefully has improved the quality of the current report, and with certainty it has contributed a high degree of caution and humility in the recognition of the limitations and challenges involved in conducting an assessment of HRV.

This report is the result of a recognition by the leadership of the Regional Office of Region 2 of both the importance of the HRV process and the desirability of participation in it by researchers who have published extensively on Rocky Mountain forest ecosystems in the peer-reviewed literature. The current round of HRV reports was initiated in 1999 by Claudia Regan, Regional Ecologist for Region 2. This report has benefited from her insights into both the science and management aspects of the HRV process. Her leadership and perseverance in this long process are deeply appreciated.

While the interpretations in this report are the responsibility of its authors, we have endeavored to address critiques and alternate interpretations from a wide variety of sources. Over the past five years we have benefitted from numerous discussions with many Forest Service personnel of topics directly related to the HRV assessment. In the current report we have addressed issues and alternative interpretations which have become apparent to us from these discussions with individuals and in numerous workshops. We also have addressed issues which were raised by Forest Service personnel in written reviews of the initial drafts of the report. Likewise, the current report addresses and has benefitted from written reviews of drafts of the report from researchers in the Region and from five anonymous reviewers selected by the Ecological Society of America.

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"Awareness and understanding of disturbance ecology and the role disturbance plays in ecosystem dynamics...is essential in understanding the consequences of management choices. The more we attempt to maintain an ecosystem in a static condition, the less likely we are to achieve what we intended."

---------- Averill et al. 1995.

1. INTRODUCTION

Modern concepts of resource management that emphasize maintenance of ecosystem integrity while also providing commodities and services to society are encompassed under the paradigm of "ecosystem management". Ecosystem management (Christensen et al. 1996) has been defined 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." Under the paradigm of ecosystem-based management, the focus of forest management is placed on sustaining ecosystems, which are part of a larger functioning ecosystem, while also sustaining yields of particular forest products and uses. The emphasis in on-the-ground management is then placed on utilizing forests in accord with a thorough understanding of the processes that maintain and have maintained ecosystem integrity (Crossley 1996). An important component of the ecosystem management paradigm is explicit recognition of the dynamic character of ecosystems. Ecosystem management is not intended to provide maintenance of any *status quo* in ecosystem conditions but rather accepts that change is an inherent characteristic of ecosystems across both space and time.

For resource managers, it is important to know the range of critical ecological processes and conditions that have characterized particular ecosystems over specified time periods and under varying degrees of human influences. An understanding of how ecosystems functioned and sustained themselves in the absence of major human modification of ecological patterns and processes provides a concrete model of ecosystem integrity (Crossley 1996, DeLeo and

Levin 1997). As applied to the management of forested ecosystems in the western U.S., an ecosystem management paradigm emphasizes knowledge of the range of ecosystem conditions prior to significant changes brought on by intensive Euro-American (people primarily of European ethnicity but also of Hispanic, Asian, and African origins) settlement and how these conditions have continued to change during the 20th century (Kaufmann et al. 1994, Morgan et al. 1994, Landres et al. 1999, Swetnam et al. 1999). The timing of major impact of Euro-American settlement varies in the West from the middle 18th to early 20th centuries, but generally begins in the latter half of the 19th century for most areas. Understanding the interactions of humans with natural variation in determining the current and future conditions of ecosystems is a primary goal of research that supports ecosystem management.

The aim of this report is to assess current knowledge of the historical range of variability (HRV) of forested ecosystems in the Arapaho and Roosevelt National Forests (AR) and in the Pike-San Isabel National Forests (PSI). These national forests comprise a large proportion of the public lands of the Colorado Front Range (FR). The current report is an integration of two draft reports which dealt separately with the HRV of the AR and PSI. The two separate reports are integrated here because of the similarity in forest ecosystems and land-use histories of all of these Front Range forests. While many generalizations are valid for all the National Forest land in the Colorado Front Range (FR), some interpretations will be identified to one or the other area of National Forests (i.e. AR or PSI). Our focus is to understand how natural processes have determined the composition, structure, and function of forested ecosystems in the Front Range of northern central Colorado over the past several centuries, and how these processes and patterns may have been altered by humans during the 19th and 20th centuries. This report is organized as follows: Chapter 1 is an overview of the goals and conceptual basis for this assessment. Chapter 2 discusses methodology and sources of information used for determining the HRV in this region and some limitations of different types of historical data. Chapter 3 describes the general setting of the FR, including the geology, climate, vegetation patterns, and a brief summary of human settlement and land use. Chapter 4 provides a brief background on long-term natural variability in vegetation and climate patterns, and on recent (post-1500 A.D.) climatic variability. Chapter 5 reviews the literature on successional dynamics and disturbance patterns for the major forest ecosystems represented in the FR; non-forest vegetation types are treated more briefly. Chapter 6 discusses how the major land uses of the FR (i.e. livestock raising, logging, road construction, and fire suppression) since the mid-19th century have affected vegetation conditions and patterns of ecological disturbance. Chapter 7 summarizes for the principal forest types the major findings of this assessment in relation to their potential management implications. Chapter 8 is a brief overview and discussion of needed research.

Whenever possible, the discussion stresses data collected in the FR. However, for many topics literature is available only for areas outside the FR or not at all. Furthermore, even for studies conducted within the FR there are limitations to the application of information collected at specific spatial and temporal scales. For example, case studies may reconstruct patterns at a stand or site level, but may not be applicable at a landscape scale. Finally, the amount of detail that is available and that can be incorporated into this report is limited. We expect that this report will serve as a general synthesis of HRV of forest vegetation for the FR and as an *entrée* into the literature for more detail on specific subjects.

1.1. Concepts of Ecosystem Management and Historical Range of Variability

The modern perspective of ecosystem management recognizes that ecosystems are not static and that sources of change include both humans and natural processes (Swanson et al. 1993, Morgan et al. 1994). Holling and Meffe (1996) argue that past natural resource management practices often have resulted in a loss in the natural variability of ecosystem processes and components. This, in turn, has led to reductions in ecosystem resilience, or the ability of an ecosystem to persist in response to shifts in driving factors or system processes. Holling and Meffe (1996) recognize two definitions for resilience in the ecological literature, and differences in these definitions have the potential to profoundly influence the conceptual basis for natural resource management actions. The first and more traditional definition for resilience they term *equilibrium resilience*, which is the ability for an ecosystem to return to some steady-state equilibrium condition after a disturbance. This concept of resilience is based on constancy and predictability, attributes that they argue are at the core of a “pathology” of natural resource management. In their view, this pathological approach to resource management seeks to reduce variability in ecosystems to make them more predictable and thus more reliable for societal and economic needs. Holling and Meffe (1996) further argue that this pathology often has reduced the resilience of ecosystems and led to unforseen and usually undesirable ecological surprises and crises.

Holling and Meffe (1996) term the second definition for resilience *ecosystem resilience*, which is characterized by the amount of instability an ecosystem can absorb before it changes to a new regime of relatively stable behavior. This definition differs from the more traditional concept of resilience in that it emphasizes system dynamics that are inherently unpredictable and may only become apparent for larger systems over longer time periods. The conceptual focus here for natural resource management is to identify actions that adversely impact ecosystem structure or function through changes in the variables and processes that control ecosystem behavior. As long as a range of variability in system behavior is retained, Holling and Meffe (1996) argue that ecosystem resiliency is maximized and ecological surprises or crises can be minimized.

The emphasis on the dynamic character of forested ecosystems in ecosystem management represents a major shift in attitudes over the past several decades among scientists and resource managers towards the concepts of change and instability in ecosystems (Pickett and White 1985, Botkin 1990). Formerly, there was a widespread expectation of a “balance of nature” that was reflected in concepts that stressed stability, such as the climax concept or homeostatic self-regulation of ecosystem properties. Today, ecosystem change is regarded as the norm, and periods of relatively rapid *versus* slow change should be expected and accommodated in management practices.

As a practical matter, understanding of natural variability in ecosystem conditions and processes also provides operational flexibility for management actions and protocols (Landres et al. 1999). Incorporating historical ecosystem patterns into management goals provides a coarse-filter strategy for dealing with sustainability of diverse and often unknown species requirements. Managing within boundaries of site variability and history is also probably easier and less expensive to achieve than trying to manage outside of constraints imposed by driving factors of the system (Allen and Hoeskstra 1992, Landres et al. 1999). Historical patterns of ecosystem conditions provide what may be the only viable model for how ecosystems have evolved and perpetuated themselves in the absence of significant human impacts.

1.2. Disturbance and Ecosystem Management

Due to the important role of disturbances such as fire in shaping current forest structures and compositions, resource managers in western North America often seek to restore processes that shaped ecosystems by reinstating select characteristics of disturbance regimes (Kaufmann et al. 1994). Natural disturbances such as fires and insect outbreaks clearly played the dominant role in shaping forest ecosystems in western North America, long before humans attained the technological capacity or sufficient densities to be a major determinant of vegetation conditions. A useful starting point in understanding the natural processes that shaped the modern landscape is to consider forest conditions from descriptions of forest structure and composition and of disturbance regimes that existed prior to significant land-use changes brought on by Euro-American settlement. This does not deny the possibility that Native Americans also affected ecosystem conditions in the West but it assumes that such influences were relatively localized or of low severity in comparison with the forest changes associated with massive Euro-American settlement in the 19th century. Disruption of historical patterns of structure and processes occurred primarily because of grazing, agriculture, logging, and alteration of fire regimes. Although studies of historical range of variability often focus on the time of initial large-scale and permanent Euro-American settlement (the late 1850s for the AR), ecosystem conditions were in flux prior to this time as well, probably due in part to impacts of Native Americans and certainly due to natural processes such as climatic variations or natural dispersal of species. Opinions vary widely over whether the ecological effects of fires set by Native Americans were severe and ubiquitous (Denevan 1992, Kay 1997), localized (Barrett and Arno 1999), or relatively negligible (Baker 2002). Such divergent opinions can only be resolved by having specific historical information on human occupation, land use, and fire history for the locale and ecosystem type of interest, as stressed in the current report. Given the flux of FR landscapes, whether it be due solely to climatic variability or with some contribution fromNative American activities, during the several centuries prior to massive Euro-American settlement, we stress that landscape conditions described for any particular date may not represent the full range of conditions that occurred over the last several centuries.

The conditions that may be described for key dates such as immediately prior to massive Euro-American settlement (i.e. 1850s) or the establishment of national forest reserves are only *snapshots* of changing landscapes. While such a reference point is helpful in understanding how ecosystems have evolved and perpetuated, it is still only a single configuration of a naturally fluctuating system and must be used cautiously (White and Walker 1997). The selection of an atypical reference period could lead to a misunderstanding of the range of ecosystem conditions that have occurred over longer time periods. Because of the fallacy of taking a single snapshot as representative of fluctuating reference conditions, the goal of this assessment is not to quantitatively reconstruct the percentage of the vegetation that may have been in some type of equilibrium (e.g. "climax") condition versus seral stage. Given that this issue which has been raised numerous times in discussion with resource managers, we emphasize that our objective is not to quantify past areas of seral and non-seral vegetation. Instead, we stress an understanding of the relationships of fluctuating vegetation conditions in relation to variability in the drivers (climate, natural disturbance, human activities) of these vegetation changes.

Analogously, the goal of an HRV assessment is not to provide a blueprint or template for restoration of ecosystems to some precisely determined past conditions. These conditions have usually varied significantly over time and may have been shaped by climatic conditions significantly different from the current or future climate. There may be a tendency to assume that the purpose of an HRV assessment is to reconstruct a snapshot of the pre-settlement landscape to be used as a target in ecological restoration. However, restoration is often not feasible or desirable (Wagner et al. 2000). Instead, an HRV assessment provides an understanding of the processes that have shaped the current landscape, which is essential for detecting trends in landscape conditions that may be attributed to management practices as opposed to natural causes such as climatic variability. The understanding of processes provided by an HRV assessment is meant to serve a wide range of potential management objectives including sustained production of forest products and services (Averill et al. 1995). Thus, an HRV assessment is a key part of the resource planning and decision-making process in the context of ecosystem-based management.

Understanding the influences of humans on landscape patterns requires an appreciation of the roles of both natural and anthropogenic disturbances in vegetation dynamics (Oliver and Larson 1990, Glenn-Lewin et al. 1992). Disturbance has been defined as: “any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment” (White and Picket 1985:7). Disturbances remove biomass that creates space for new individuals and releases resources to new and surviving ones. Holling (1992) describes disturbances as mesoscale ecosystem processes that “entrain” microscale vegetative processes responsible for regeneration, growth, and mortality of individual plants to create distinct community and landscape architectures. These architectures are the result of differences in temporal and spatial patterns inherent in disturbance regimes. Historical legacies in vegetation patterns, established by spatial and temporal variability in disturbance occurrences, can persist for long periods until new vegetation patterns are restructured by either climate changes or subsequent intense disturbances. An understanding of vegetation change that results from disturbances is of critical importance for assessing the possible ecological impacts of management activities aimed at suppressing disturbances such as fire or insect outbreaks.

A conceptual framework for analyzing the characteristics and consequences of disturbance is that of the *disturbance regime*, or the combination of spatial and temporal characteristics of disturbances in a particular landscape (Paine and Levin 1981, White and Pickett 1985). The key potential descriptors of a disturbance regime are: 1) spatial distribution; 2) frequency; 3) size of the area disturbed; 4) mean return interval; 5) predictability; 6) rotation period (time required to disturb an area equivalent to the study area once); 7) magnitude or severity; and 8) the synergistic interactions of different kinds of disturbances and their driving factors (e.g., climate, human ignition sources). Variations in these parameters are major determinants of landscape heterogeneity. Although there are numerous case studies of vegetation and ecosystem response to disturbance for the southern Wyoming to southern Colorado sector of the southern Rocky Mountain floristic province, relatively little work has been done on disturbance regimes *per se*. Fire has been the most frequently studied disturbance type, but even for fire, available data are insufficient for a comprehensive quantification of fire regimes across a full range of forest and grassland ecosystem types. Inevitably, assessment of historical ranges of variability of disturbances and their ecological consequences must depend on interpretation of scattered and fragmentary evidence of both a quantitative and qualitative nature.

1.3. Objectives and Questions Addressed in This Assessment

The overall objective of this report is to synthesize and assess the state of knowledge on how ecological disturbances and forest conditions have varied in the Colorado Front Range over the past several centuries. Knowledge of this historic range of variability of forest ecosystem conditions is essential for understanding how current forest conditions may be related to past and current management practices and climatic variability. The knowledge of past forest conditions is not meant to serve as a default target for forest management, but rather it will inform decisions about forest management which must consider a variety of socio-economic issues that are beyond the scope of an HRV assessment. We stress that knowledge of past forest conditions would be a simplistic and incomplete basis for forest management decisions, but we also agree with the argument that ignorance of how current forest structures and compositions have been created during the past several centuries of varying natural disturbances, management practices, and climatic variability would endanger the future success of current management conditions (Averill et al. 1995).

The foci of this assessment are changes in disturbance patterns and processes as they influence forest conditions such as structure (tree ages, sizes), successional status and relative dominance by different tree species, extent of forest cover types, and tree health. Thus, this assessment is clearly oriented towards an improved understanding of disturbance ecology and forest stand dynamics (*sensu* Oliver and Larson 1990). This emphasis does not imply that lesser importance is given to ecosystem parameters such as productivity and biogeochemical cycling. Rather, it reflects the fact that there are no historical data on those aspects of ecosystem function, and that such information is exceedingly scarce even for the modern forest ecosystems of the FR. However, the strategy of this assessment recognizes that modern studies may reveal relationships between forest structure and composition and the ecosystem functional properties of productivity and biogeochemical cycling that will allow inference about past ecosystem function. Analogously, due to a lack of data we cannot directly compare past and present biodiversity, but some inferences on how biodiversity may have changed in relation to land-use practices (e.g. grazing) can be suggested on the basis of studies of contemporary impacts of land-use on biodiversity. For example, presence of a large number of non-native species in grasslands and riparian communities is a clear indicator that those communities are outside their historic range of variability.

This assessment of historical range of variability will address the following four key questions that are critical to forest management in the context of an ecosystem management paradigm (Veblen 2000):

1. How do disturbance regimes vary spatially? In particular, how does the occurrence of disturbances such as fire and insect outbreak vary along environmental gradients (e.g., low *versus* high elevation) and for different forest types?

2. How have humans altered natural disturbance regimes and how has this varied spatially? For example, fire frequency (i.e., number of fires per time period) in low elevation forests clearly has declined during the modern fire exclusion period but has it been significantly altered for high elevation forests?

3. How do disturbance interactions affect vegetation responses as well as the occurrence and spread of subsequent disturbances? For example, how has modern fire exclusion altered the probability of the occurrence or severity of insect outbreaks?

4. How does climatic variability affect disturbance regimes and vegetation response to disturbances? To what extent might some of the forest health problems usually attributed to management practices (e.g., fire exclusion) be due to climatic variability?

The four questions must be examined for specific forest ecosystem types across the FR to provide scientific background on which resource planning and management decisions can be based. To address these questions, we will draw on a mixture of quantitative data and qualitative observations of forest ecosystems in the Front Range. We acknowledge that for some disturbance types and some vegetation types, the information base is inadequate for definitive answers to our four central questions. Thus, we will explicitly identify levels of certainty in our conclusions, and often will be tentative in our assessments. Furthermore, we will identify the types of future research needed to improve the certainty of our assessments.

The four questions above are directly relevant to national policy being implemented under the Healthy Forests Restoration Act of 2003 (H.R. 1904). There are a number of premises underlying this act that require critical evaluation for specific forest ecosystems. The key premises that garnered public and political support for President Bush's Healthy Forests Initiative and the Healthy Forests Restoration Act of 2003 include the belief that fire suppression during the 20th century has resulted in unnatural fuel buildups and unnaturally high tree densities, which in turn are resulting in fire severities and outbreaks of forest pests beyond the historic range of variability in the western U.S. (White House 2002, USDA Forest Service 2002c). National policy identifies unnatural fuel buildup as a widespread risk across the West: "Today, the forests and rangelands of the West have become unnaturally dense, and ecosystem health has suffered significantly. When coupled with seasonal droughts, these unhealthy forests, overloaded with fuels, are vulnerable to unnaturally severe wildfires. Currently, 190 million acres [77 million ha] of public land are at increased risk of catastrophic wildfires" (White House 2002). Despite broad generalizations about the effects of fire suppression on current forest structures and recent fire behavior in the West (e.g. Covington 2000, Malakoff 2002), fire history has varied substantially across the range of forest ecosystem types so that what may be a valid generalization for a low elevation, xeric pine woodland may not be true in higher elevation, more mesic forests (Veblen 2003b, Schoennagel et al. 2004). Thus, rather than applying research published for other forest ecosystems, even for the same cover type, resource managers and planners must critically evaluate the relevance of that research to their particular management units.

2. METHODOLOGY

This assessment of HRV in the FR is based on a variety of sources. When possible, we emphasize published sources of information that have been subjected to critical examination in the peer-review process. Information was obtained from scientific and other journals, books, unpublished government reports, and unpublished data sets from the Forest Service or from individual researchers. Some types of information will not have been through peer review, but we attempt to place these data into the context of this assessment using our own evaluations. When there is a choice we cite data sources that are available to the public (e.g., publications, dissertations, and theses) rather than individually held data sets (e.g., works in progress).

HRV is assessed mainly for the forested areas of the FR because much of the available data, especially tree-ring data, are from those ecosystems. However, some of the general patterns can be extrapolated to other non-forest vegetation types based on their locations near areas of reconstructed variability. Our interpretations of data and the literature incorporate our personal field experiences in the FR and nearby areas and also benefit from feedback from Forest Service personnel and other people with firsthand knowledge of the FR.

For the purposes of this report, we roughly define several scales of analysis. Stand level refers to historical patterns that occur over spatial scales of 0.01 to a few km^2 in vegetation units that are relatively homogeneous. Landscape scale refers to spatial scales of a few to c. 100 km^2 and includes a substantial range of community types and structures. Regional patterns are those that occur over areas much greater than 100 km^2, and may include historical patterns that are known or inferred to have occurred over much of central and southern Colorado. Knowledge of past patterns is highly uneven for different spatial scales. For example, historical patterns may be well known at a stand scale (e.g., when inferences are made from tree age-structures) but less certain at landscape or regional scales.

Information on historical variability in ecosystem conditions comes from a variety of sources, both quantitative and qualitative in nature, each with its own limitations that need careful interpretation before use in guiding management decisions. This assessment relies on direct sources of information (e.g., eyewitness descriptions of past events and landscapes) as well as proxy data on the historical range of variability. In the absence of direct measurements of ecosystem variability, proxy data are used to reconstruct historical ecosystem variability from indirect evidence. Well-known examples of proxy data include the use of fossil pollen to reconstruct past vegetation (e.g., Fall 1997) and the use of tree-ring widths to describe pre-historical climatic variation (e.g., Woodhouse 1993). Although proxy data are quantitative, they are censored in the sense that the full range of data may not be available because we can only sample what has survived to the present. Proxy records are filtered by past environmental processes and loss of evidence through time, often resulting in missing, patchy, or altered records. The accuracy of any environmental reconstruction from proxy records depends on how well this filtering process is understood and modeled (Swetnam et al. 1999).

Information sources utilized in this HRV assessment are broadly classified as: 1) historical records and studies; 2) macro- and micro-fossil records; 3) tree rings; and 4) modern conditions of ecosystem parameters. The limitations and potential biases of each of these sources has been discussed in detail in methodological reviews (e.g., Fritts 1976, Vale 1982, Rogers et al. 1984, Prentice 1988, Fritts and Swetnam 1989, Johnson and Gutsell 1994, Swetnam et al. 1999, Baker and Ehle 2001, Veblen 2003a). Below, we briefly comment on the key sources specifically used in this assessment.

2.1. Early Historical Records, Photographs, and Reports

The earliest survey of vegetation conditions of much of the area now encompassed by the PSI is Jack's (1900) report on the Pike's Peak, Plum Creek, and South Platte Timber Reserves conducted for the U.S. Geological Survey. This report combines Jack's personal observations with information obtained from the early white settlers of the area to describe the physical landscape, vegetation, logging, fires, livestock use, settlement and general land use. Although this is the most valuable early description of Pike N.F., Jack did not use any sampling procedure that yielded quantitative descriptions of past landscape conditions. Nevertheless, his narrative and numerous photographs indicate that he conducted fieldwork over a large part of the Reserves. He mapped areas of burned and unburned forest, but in the absence of aerial photography or details of any survey methods employed, we are uncertain of the accuracy of the map. Jack's report contains over 90 photographs many of which show large expanses of landscape and he specifically claims that the photographs are a fair representation of the forest conditions of the Reserves. Although the historic photographs cannot be used to quantitatively assess past landscape conditions, the abundance of the photos and their scattered locations imply that they represent widespread landscape conditions.

We have cautiously interpreted Jack's (1900) findings in the context of other sources of information such as independent descriptions of the landscape, fire-scar dates, and age structures of the modern forests. Other key historical studies, commissioned by the U.S.D.A. Forest Service, are Agee and Cuenin's (1924) history of much of the area that now comprises San Isabel N.F. and Ingwall's (1923) history of Pike N.F. and surrounding areas. Both histories focus on changes in resource conditions, and are based on compilations of information and interviews with early settlers who observed the landscape as early as the 1850s. These three reports explicitly consider possible biases of informants, and check their information against independent observations (including their own). All three historical studies present consistent descriptions and interpretations of important impacts (e.g., logging, burning) and landscape changes that occurred in the late 19th century.

For the AR there are no early comprehensive reports by U.S. government agents on forest conditions comparable to those for Pike and White River National Forests (Jack 1900, Sudworth 1900). The earliest, useful report produced by a Forest Service employee is Ingwall's (1923) history of Pike N.F. that includes observations on surrounding areas, including the southern sectors of the AR. Ingwall's history focuses on changes in resource conditions, and is based on compilations of information and interviews with early settlers who observed the landscape as early as the late 1800s. Early forest conditions also are inferred from a wide variety of historical sources many of which are summarized in the comprehensive history of landscape change in Colorado by Wyckoff (1999).

The documentary record of fires made by the Forest Service for the FR begins in c. 1909, which approximately coincides with the beginning of the fire suppression policy. Thus, this record is not useful for comparison of fire occurrence during the fire suppression era *versus* the pre-fire suppression era. Furthermore, even within the time period of the documentary record, limitations of this record result from: 1) probable improvement in the accuracy of estimates of areas burned based on later use of aerial reconnaissance and aerial photographs; and 2) significant changes in the boundaries of the AR and PSI, and therefore in the areas to which the fire observations pertain. Thus, we make only limited and cautious use of the documentary record of fire in the AR and PSI. Although temporal trends in the record must be interpreted cautiously, it is a useful source on the relative frequency of fires in different cover types. We also made limited use of the fire record beginning in 1915 for Rocky Mountain

National Park because the AR surrounds the Park.

2.2. Macro- and Micro-Fossil Records

Macro and micro-fossil records of plant and insect remains are available for sites in or near the FR in the southern Rocky Mountains (e.g., Elias 1996, Fall 1997). These studies yield quantitative evidence of past environmental conditions; however, they are censored samples of past environmental conditions that require assumptions in reconstructing vegetation and/or climate. Nevertheless, the broad patterns interpreted from fossil data are useful for describing general patterns of vegetation and climate during the past c. 12,000 years. Although the temporal scales of environments reconstructed from micro- and macro-fossil are often too coarse for understanding specific changes in recent centuries (Kaufmann et al. 1998), we include these studies here to document some of the long-term changes in vegetation patterns that may be continuing during the 20th century.

2.3. Tree-Ring Evidence of Disturbance Regimes and Past Environmental Conditions

Tree rings are used to describe the histories of fire, climatic variation, and outbreaks of some insects over the past several hundred years as well as in the evaluation of stand development patterns as influenced by human activities over the past c. 150 years. For assessing climatic variation during the pre-instrumental time period (prior to c. 1880s in the AR), tree rings can provide annually resolved information on climate (e.g. Graybill 1989, Woodhouse 1993). Tree rings document primarily year-to-year variation in moisture availability and its potential impact on fire occurrence (e.g. Cook et al. 1998, Veblen et al. 2000).

Tree rings also are used to date past fires by dating fire scars and ages of post-fire cohorts, from which many of the parameters of an historic fire regime can be at least partially resolved (Veblen 2003b). We follow standard terminology for distinguishing between low-intensity *surface fires* and high-intensity *crown fires* (Van Wagner 1977). Surface fires burn on the surface in litter and mostly herbaceous vegetation. In the FR, they can kill small tree seedlings but would rarely kill mature trees. Crown fires kill high percentages of the canopy trees, and may spread passively (torching trees from the surface), actively (spreading jointly with the surface fire), or independently of the surface fire. Tree-ring data usually cannot distinguish among the types of crown fire spread, but can be used to distinguish between low-intensity surface fires and intense crown fires.

Tree-ring data can be used to describe a variety of the descriptors of a fire regime, but usually cannot fully describe all parameters. The key fire regime parameters used in this report are fire frequency, interval, extent and severity. *Fire frequency* is the number of fires occurring within a specified area during a specified period (Romme et al. 2003a). The number of years between successive fires within a specified area or at a single point is the *fire interval* (or return interval). Fire *extent* (or size) refers to the total area burned in a single fire event; the fire event may be one continuous period of burning or may refer to the aggregated area of burns that occurred in a single year. *Fire severity* refers to the effects of fire on organisms, and is used as a rough approximation of fire intensity (amount of energy released during a fire). Terminology for describing fire severity generally follows Romme et al. (2003a). *High-severity* and *stand-replacing* are used synonymously to describe fires that kill all or most canopy trees. There is no agreed upon threshold level of canopy tree mortality required for the usage of these terms, but in most cases well over half the basal area of a stand would be killed in a stand-replacing fire. To stress the fact that not all trees are killed in some high-severity fires, we occasionally use the term *partially stand-replacing*. In the FR, most high-severity forest fires create opportunities for recruitment of a new cohort of canopy trees. However, there may be significant lags and variations in the rate of post-fire tree establishment related to harsh conditions created by exceptionally severe fires, lack of seed trees, and/or climatic variability. *Low-severity fires* kill few or no canopy trees, and generally do not result in a distinct cohort of post-fire tree establishment. *Mixed severity fires* are events that include patches of high-severity fires as well as areas of low-severity, non-lethal burns. Thus, mixed severity fires create a mosaic of heterogeneous fire severity across the landscape. Each severity type (i.e. high-severity versus low-severity fires) should reach a minimum threshold of c. 20% of the area burned to justify describing the event as mixed severity; otherwise, such events should be described as mainly high- or low-severity fires.

There are a number of limitations on how accurately or completely an historic fire regime can be described from tree-ring evidence. Fire history in forested areas can be described quantitatively on the basis of two types of tree-ring evidence: dates of fire scars (fire-interval approach) or age of stands that presumably regenerated following stand-replacing fires (stand-origin approach). The fire-scar based approach usually provides annual (or even seasonal) resolution of the dating of past fire events but is limited in its ability to determine the spatial extent of past fires. In contrast, the stand-origin approach utilizes the extent of even-aged post-fire stands to delimit the boundaries of the most recent fire at a site (Johnson and Gutsell 1994) but often does not provide annual resolution of the fire date (Kipfmueller and

Baker 1998a). Important limitations of the fire-scar based method include (McBride 1983, Agee 1993, Lertzman et al. 1998, Swetnam et al. 1999, Baker and Ehle 2001, Veblen 2003a): 1) possible elimination of part of the fire record due to logging or extensive tree mortality caused by insect outbreaks or recent intense fires; 2) uncertainty over how representative fire-scar dates are for the larger landscape when scarce old trees are subjectively sampled for fire scars; and 3) incomplete recording of fires due either to fire behavior (e.g. fast moving fires) or lack of fire-scar-susceptible trees at the time of the fire. Thus, fire histories based on fire scars rarely provide a complete and spatially precise record of all past fires within a sample area or a larger landscape. Despite these limitations, a fire history based on an adequate sample size of cross-dated fire-scar dates provides a useful quantitative, but filtered, index of past trends in fire occurrence.

Fire history studies can document trends and can describe dominant fire severity (high vs. low) but do not necessarily provide precise fire frequency targets; for these and other reasons, summary statistics of fire regimes should not be mimicked in an effort to re-incorporate fire at a “natural” frequency (Veblen 2003a). The degree to which such studies can be used to understand an actual fire regime depends on how well the filtering process is understood (Lertzman et al. 1998, Swetnam et al. 1999, Baker and Ehle 2001). Although useful in analyzing fire regimes and testing hypotheses, summary statistics of fire history, such as mean fire interval or fire rotation, are incomplete descriptors of past fire behavior and the ecological effects of fires (Veblen 2003a). Because of these uncertainties, we make only limited and cautious comparative use of mean fire intervals for characterizing fire occurrence in different time periods or different ecosystems. Instead, we stress the importance of considering trends in regional annual indices of fire occurrence. We also stress the interpretation of fire-scar evidence in conjunction with other lines of evidence such as tree population age structures, changes in growth patterns of individual trees, and historical landscape photographs, all of which may provide some indication of the ecological impact of past fires.

In contrast to the application of the fire-interval approach to forests dominated by fire-resistant pines and Douglas-fir, fire history studies conducted in dense lodgepole pine and spruce-fir forests, the most important forest types of the FR, are usually derived from dating post-fire stands, sometimes in combination with fire-scar dating (Romme and Knight 1981, Kipfmueller and Baker 1998). Potential sources of error in the stand-origin approach include (Veblen 1992, 2003a, Goldblum and Veblen 1992, Kipfmueller and Baker 1998): 1) undetermined time lags in tree establishment after stand-replacing fires; 2) erroneous determination of total tree age due to tree core samples that do not intercept the pith right at the root/shoot interface to determine actual germination date; 3) erroneous ring counts due to missing or false rings if samples are not crossdated; 4) possible confusion of a post-fire cohort with tree establishment following other types of disturbance or climatic variation; and 5) destruction of evidence of earlier fires by more recent burning. Due to these and other limitations (Huggard and Arsenault 1999) we cautiously interpret fire histories from the stand-origin approach. Again, summary statistics such as fire rotation have their appropriate use in analyzing fire regimes, but are not sufficiently robust to serve as models or targets for management.

2.4. Inference from Modern Conditions of Ecosystem Parameters

This assessment makes use of modern (e.g., post 1950) descriptions of ecosystem parameters in the FR such as data on current forest structure and composition. The fundamental limitation of these data sources is the lack of any comparable pre-20th century databases for comparison. However, tree growth and age data from living as well as dead trees collected during the modern period can be used to reconstruct past forest structures (e.g., Donnegan and Rebertus 1999, Kaufmann et al. 2000). Such studies can provide valuable insights into past forest composition and structure as well as disturbance histories. The principal limitations of such intensive stand reconstructions include: 1) limited spatial extent; 2) incomplete preservation of evidence with increasing time before present; and 3) at some sites, incomplete or no information on potentially important past influences of livestock on stand structure. Decay and disappearance of dead trees results in approximate and partial reconstructions of past forest conditions. Consequently, the quantitative limitation to their precision needs to be considered when interpreting their implications for ecological restoration. The most serious limitation of such studies is the great uncertainty of the spatial applicability of their results. For example, to what percentage of the distribution of a forest type do these localized stand reconstructions apply?

At a broad spatial scale, tree age data from the U.S.D.A. Forest Service's Resource Information System (RIS) provide the potential for interpreting general features of past forest structure. However, intensive age structure studies of the same species in the southern Rocky Mountains indicate serious limitations of the FR RIS data for forest reconstruction: 1) cores collected at breast height (as in the RIS database) may miss many decades of tree age, especially for slow-growing trees in the subalpine zone (Veblen 1986a); 2) the small number of trees sampled in each stand do not constitute an objective sample, and may not include the oldest tree in the stand (Goldblum and Veblen 1992, Kipfmueller and Baker 1998a); and 3) core samples that did not intercept the pith do not accurately date the tree's age at coring height (Veblen 1992). Furthermore, our personal experiences indicate that ring counts done in the field are often highly inaccurate. Given these problems, we make only limited and cautious use of tree age data from the RIS database. Thus, we avoid overly detailed interpretations because of the imprecision in determining tree establishment dates, but because of the large size of the data sets, covering tens of thousands of hectares, the general patterns of stand ages are regarded as representative of the FR landscape.

Given the limitations to all sources of information on HRV, we cannot rely on only one line of evidence to reconstruct historical variability. Instead, we use inferences and evidence from multiple sources to draw conclusions about general to specific patterns that resulted from past processes. Where specific information on past forest conditions is lacking, we sometimes draw inferences from known past processes. For example, although we lack quantitative forest inventory data for the reference period (c. 17th to middle 19th centuries), extraction of large diameter trees of certain species in the late 1800s clearly indicates that the forests of the reference period contained trees of such sizes. Some interpretations are presented as hypotheses, and new data and studies may contradict or support interpretations presented here.

Finally, a critical issue in assessing the range of historic variability in the FR is the extent to which conclusions based on studies conducted elsewhere can be applied to the FR. For example, there is abundant literature on fire history in ponderosa pine ecosystems in the Southwest. However, differences in climate, vegetation structure, and human settlement mean that the conclusions from these studies cannot be uncritically accepted for the FR. For example, although fire history has been studied extensively in ponderosa pine forests in Arizona and New Mexico, there is abundant evidence that the findings of those studies have only limited application to the ponderosa pine cover type of the FR (Brown et al. 1999, Kaufmann et al. 2000, Veblen et al. 2000). In this report, whenever possible, we rely on literature based on studies conducted in the FR or nearby areas. For topics that lack on-site studies, we necessarily must refer to studies conducted elsewhere. We will make explicit what we consider to be general patterns for ecosystems similar to those found on the FR versus patterns that have been documented specifically in the FR. We consider findings from elsewhere as hypotheses that need to be critically examined for the FR. These hypotheses can be evaluated according to similarity of sites in terms of ecosystem attributes (e.g., habitat types), human history, and regional climates. Even for studies conducted within FR, there are important limitations to applicability of the data over the entire FR. Throughout this report we compare present ecosystem conditions with those of the probable range of historical variation and categorize those conditions as being “inside” or “outside” of HRV.

3. ENVIRONMENTAL AND CULTURAL SETTING OF THE STUDY AREA

3.1. Geology and soils

3.1.1. Arapaho-Roosevelt N.F.

The AR includes an area of about 526,000 ha (1.3 million acres), administratively divided into five ranger districts: Boulder, Sulphur, Clear Creek, Estes-Poudre, and Redfeather (Fig. 3.1). The Estes-Poudre and Redfeather Districts recently were combined into the Canyon Lakes District. In elevation the AR ranges from the margins of the western Great Plains grasslands at around 1550 m (5084 ft) to alpine environments on peaks such as Mt. Evans at over 4200 m (13,776 ft). Forest boundaries extend from the Wyoming border to the north to Mt. Evans in the South. The dominant physiographic feature of the AR is the northern half of the Colorado Front Range, the summits of which form the Continental Divide in this area. Arapaho NF includes lands on both sides of the Continental Divide, while Roosevelt NF is confined to the eastern side of the Divide.

Individual mountain ranges in the AR include, from the north, the southern end of the Medicine Bow Mountains, the Rawah Range, Mummy Range, Never Summer Mountains, Indian Peaks, Vasquez Mountains, and the Mt. Evans area. The AR surrounds Rocky Mountain National Park. The Forest also surrounds and includes portions of Middle Park west of the Continental Divide and forms the eastern boundary of North Park on the Wyoming border. These Parks are high elevation (ca. 2200 to 2800 m) grasslands.

The Front Range is an extended mountain chain dating from the Laramide Orogeny (c. 66 million years ago) that runs south from Wyoming to approximately Cañon City, Colorado. A Precambrian core of gneiss, granite, and schist uplifted and eroded to the east the Tertiary pediment that was once continuous with sediments that form the high plains (Chronic and

Chronic 1972). The mountain core of gneiss, schist, and granite, which contains many economic mineral deposits, is exposed throughout much of the Front Range. At the base of the eastern side of the Front Range, the original High Plains surface has been eroded during the Quaternary by the Cache La Poudre and South Platte Rivers to form the Colorado Piedmont, making the mountain front even more of a dramatic rise from the Great Plains. The western margin of the Front Range consists of a precipitous decline into the broad troughs of North and Middle Parks.

The eastern slope of the Front Range consists of long, steep slopes leading up to interfluves that range from sharp ridges to broad, nearly level upland surfaces. Much of the higher part of the Front Range extending from the crestline down to c. 2600 m (8528 ft) was glaciated during the Pleistocene (Richmond 1960). The work of the glaciers is seen in the cirques, tarns, deep U-shaped troughs of the upper courses of river valleys, and broad moraine-enclosed basins.

Much of the AR is critical watershed for growing urban areas on the Front Range and the Western US. Headwaters of the Colorado River are in Middle Park on the western slope of the Forest and several diversions bring west slope water underneath the Continental Divide to eastside urban areas. Many eastside streams are also diverted or dammed for irrigation and urban water use.

Soils of the northern Front Range are quite heterogeneous but generally are immature, typically coarse-textured and often rocky (Peet 1981). Many of the more stable soils above c. 2600 m (8528 ft) elevation are derived from Pleistocene glacial till. The lower montane zone (1750 - 2450 m; 5740 - 8036 ft) is dominated by ustolls at the lowest and driest sites and by cryoboralfs on more mesic sites. Ustolls are typical of meadows and open park-like stands of conifers. Cryoboralfs are dominant in the upper montane zone (2450 - 2850 m; 8036 - 9348 ft) and commonly support more mesic forests. In the subalpine zone (2850 - 3500 m; 9348 - 11,480 ft), mesic forests are typically underlain by cryorthods. More open woodlands are associated with cryoboralfs and poorly drained sites with cryaquods. Boggy soils are also common in the subalpine zone and are classified as borohemists and borofibrists (Peet 1981).

3.1.2. Pike-San Isabel N.F.

The PSI includes an area of more than 850,000 hectares (2.1 million acres), administratively divided into six districts (Fig. 3.2). In elevation the PSI ranges from c. 1790 m to c. 4390 m. A dominant feature of Pike N.F. is South Park, one of three major Colorado valleys sandwiched between the Front Range and a series of mountain ranges lying farther west. The PSI includes parts of the physiographic regions of the Front Range (including Pikes Peak), the Mosquito Range, South Park, the Sawatch Range, the Arkansas Hills, the Sangre de Cristo Range, and the Wet Mountains and Wet Mountain Valley (Larkin et al. 1980) (Fig. 3.3). Geologically, the area covered by the PSI is a complex of anticlines and synclines of Precambrian crystalline rock sandwiching a cover of faulted and folded Paleozoic and Mesozoic sedimentary strata, topped with thin gravels and interrupted by more recent Tertiary and Quaternary deposits and intrusions.

The Front Range, which abuts the high plains to the east, is a long uplift running south from Wyoming to approximately Cañon City, Colorado. At higher elevations in the Front Range, much of the overlying sedimentary strata have been eroded leaving an exposed core of gneiss, granite and schist (Chronic and Chronic 1972). Toward the southern end of the Front Range, mountains in the eastern part of Pike N.F. are commonly composed of granitic and igneous rocks of the Pikes Peak batholith (Moore 1992).

In the broad, high-elevation plain of the South Park area, the oldest formations are found on the west side in the Mosquito Range, with the youngest to the east along the Front Range (Chronic 1998). Hornblende gneiss and amphibolite are found in the northern reaches of the Pike. Andesitic lavas and ash are common over much of southern South Park as a result of the Tertiary eruption of the Thirty-nine Mile Volcanic Field (USDA Forest Service 1995). Additionally, much of the floor of South Park was once a saline lake bed. Consequently, salt deposits remain in many areas and have contributed to artesian mineral springs that were valued for their medicinal purposes by the Ute Indians and by early Euro-American settlers (Ingwall 1923, Chronic 1998).

In the southern reaches of San Isabel N.F., volcanic dikes and intrusions appear, radiating from the major igneous intrusion of the Spanish Peaks (Chronic 1998). The adjacent Sangre de Cristo Range is a narrow, long sweeping range extending from the Sawatch Range south into New Mexico. The dominant strata of the northern part of the range are composed of Precambrian igneous and metamorphic rock, while other areas in the range contain limestones, shales, and other sedimentary strata (Chronic and Chronic 1972).

To the northeast of the Sangre de Cristo Range lies the Wet Mountain Valley and then the Wet Mountains. The Wet Mountains are composed of a core of Precambrian granite, similar to the Front Range, but with more faulting apparent in the overlying sedimentary strata (Chronic and Chronic 1972).

The soils of the PSI vary widely with topography. Many open forest stands on gentle slopes have fairly well developed soils with moderate amounts of organic matter from the decomposition of dense grasses. Areas in southern South Park, where a significant ash layer is present, also have well developed soils. The majority of the soils detailed in the eastern Pike N.F. soil survey (Moore 1992) are of the Sphinx family and Sphinx family complex (> 57%), defined as sandy-skeletal, mixed, frigid, shallow typic ustorthents.

3.2. Climate of the Colorado Front Range

The climate in the region of the FR is predominantly controlled by its high elevation, continental interior location, and position relative to air mass movements (Greenland et al. 1985). Its interior location results in relatively dry conditions and wide differences between summer and winter temperatures. Synoptic-scale climate is dominated in the winter by westerly flow from the Pacific. During winter when northern Pacific maritime air masses meet the Rocky Mountains, orographic precipitation occurs on the western slope. Cold, high-pressure air masses originating over northern Canada and the Arctic Ocean occasionally bring dry and extremely cold air masses to the Front Range during winter. Tropical maritime air masses that form over the Gulf of Mexico bring warm humid air to the Front Range primarily during summer. On the eastern slope, occasional upslope conditions develop, especially in spring and autumn, pulling moist air up from the Gulf of Mexico, creating heavy precipitation along the eastern slope of the Rockies. In the summer, with the development of the North American monsoon, convection cells create locally heavy thunderstorms often with brief but intense precipitation (Barry et al. 1981, Barry 1992). The regional climate in Colorado may also be influenced by mid-tropospheric pressure anomalies associated with the Pacific-North American pattern (PNA). Cayan (1996) has demonstrated an association between anomalously low snowpack and winter PNA for most of the western United States.

In the northern Front Range, there are strong seasonal differences in temperature and precipitation in the AR. Mean monthly temperature records from climate stations in or near the AR (Table 3.1) show similar seasonal trends (Fig. 3.4). Precipitation generally increases from its minimum during the winter months of December and January to peaks in spring and late summer (Fig. 3.5). In the northern Front Range, precipitation from October through May is primarily associated with cyclonic storms, and at higher elevations occurs primarily in the form of snow. Precipitation typically peaks in April or May in association with upslope movements of moist air masses. The July-August peak in precipitation is associated with convective storms under the influence of the North American monsoon, and this peak is more pronounced further south in Colorado.

In the southern Front Range, mean monthly temperature records from climate stations in or near the PSI (Table 3.2) show similar seasonal trends (Fig. 3.6), but precipitation seasonality varies along a north-south gradient (Fig. 3.7). For most of the PSI region, precipitation generally increases from its minimum during the winter months of December and January to higher levels in the early spring. The highest amounts of precipitation occur in July and August, due to the influence of the North American monsoon. Northwards along the Colorado Front Range, the precipitation peaks for July and August become less pronounced with increasing distance from the source of the monsoon. Compared to lower elevations, the highest-elevation station at Leadville (3103 m) consistently recorded higher amounts of precipitation during the winter months (Fig. 3.7).

One of the more interesting climate patterns of the FR is the conspicuous drop in precipitation for the month of June along the eastern slope of the Rockies in south-central Colorado (Fig. 3.7; Donnegan 1999). June is a relatively dry month relative to the remainder of the growing season, April to September. This may represent a time when synoptic patterns are shifting from the dominance of westerly, Pacific flow in the winter and spring, to the southwesterly flow of the North American monsoon in the mid- to late-summer. While this change is occurring, the temporary position of the summertime mid-tropospheric subtropical ridge may prevent precipitation in the region (Carelton et al. 1990).

Elevation has a significant impact on seasonal moisture availability. Because high-elevation sites have cooler late-spring and summer temperatures and higher amounts of winter precipitation, these sites would also be expected to have higher moisture availabilities during the early part of the growing season. Additionally, the lag in snowmelt at higher elevations would be expected to supply soil moisture in mid-spring (i.e. June), when precipitation is greatly reduced. Low elevation areas experience a water deficit in June if snowpack has disappeared and precipitation is absent. Thus, tree growth is more sensitive to June moisture deficits at low than at higher elevations in the Front Range (Donnegan 1999). Variation in spring and especially June precipitation also has been shown to be a strong influence on establishment of ponderosa pine in the lower montane habitat of the northern Front Range (League 2004). Analogously, variation in spring (April-June) moisture availability has a greater impact on fire occurrence at low elevations (below c. 2500 m) whereas mid-summer (July-August) has a relatively greater impact on fire occurrence at high elevations (above c. 3000 m; Veblen et al.

2000, Sherriff et al. 2001). These differences in sensitivities of fire regimes to spring and summer moisture availability are inferred to be the result of earlier fuel desiccation at low elevations.

3.3. Vegetation Patterns

3.3.1 Arapaho-Roosevelt N.F.

The large area and great elevational range in the AR encompass a diverse array of vegetation types ranging from grassland prairies to alpine tundra, all included within the Southern Rocky Mountain Steppe-Open Woodland-Coniferous Forest-Alpine Meadow Province (Bailey 1995,1997). In the AR, vegetation pattern is strongly controlled by elevation and moisture gradients (Fig. 3.8; Peet 1981, 2000). Each of these factors plays a major role in the amount, distribution, and persistence of moisture available to vegetation. Elevation also affects temperature that controls growing season length and the amount of freezing that species must be able to tolerate.

In the northern Front Range, two general forest zones can be recognized according to elevation (Marr 1961): 1) a montane zone that includes ponderosa pine (*Pinus ponderosa* var. *scopulorum*) and Douglas-fir (*Pseudotsuga menziesii*); and 2) a subalpine zone that includes Engelmann spruce (*Picea englemannii*) - subalpine fir (*Abies lasiocarpa*) forests, lodgepole pine (*Pinus contorta*), aspen (*Populus tremuloides*), and limber pine (*Pinus flexilis*). Bristlecone pine (*Pinus aristata*) is a minor component of the subalpine zone of the AR, occurring only in the southern part of the AR. Blue spruce (*Picea pungens*) occurs primarily as a riparian species in the upper montane and lower subalpine zone occupying only a small surface area. Aspen, lodgepole pine, and limber pine cover types occur most extensively in the subalpine zone, but also occur to a limited extent in the montane zone. The montane zone borders the Plains grasslands to the east, and in the foothills of the eastern slope includes shrublands and meadows. At higher elevations, treeless areas are scattered through the forested lands, and are dominated by grasses, sedges, forbs, and sage (*Artemisia* spp.; Peet 2000). Although outside the scope of this report, alpine vegetation is described in Bowman et al. (2002). In contrast to the southern Front Range, woodlands of pygmy conifers are generally unimportant in the AR region. A disjunct population of piñon (*Pinus edulis*) occurs northwest of Fort Collins, but is believed to have been dispersed there by Native Americans only a few hundred years ago (Betancourt et al. 1991). Gambel oak (*Quercus gambelii*) occurs mainly south of the AR region from Evergreen southwards; however, it has been planted and is feral in the vicinity of Boulder

(Weber 1990). The most extensive forest cover types are ponderosa pine and Douglas-fir (sometimes classified as mixed conifer in Forest Service documents) in the montane zone, and spruce-fir, aspen, and lodgepole pine in the subalpine zone (Table 3.3).

Within the montane zone of the AR region, the ponderosa pine cover type is the dominant forest type along the eastern slope of the Front Range but is scarce on the western side of the continental divide (Fig. 3.9a). This is an important contrast between the eastern and western sides of the northern Front Range because on the east the lower subalpine zone is bordered by extensive ponderosa pine and Douglas-fir forests but on the west the lower subalpine zone abuts mainly with sagebrush-grasslands. On the eastern slope, open ponderosa pine woodlands dominate the lower montane zone (c. 1830 to 2350 m) of the northern Front Range (Marr 1961, Peet 1981). These are often sparse stands with larger surface areas covered by grasses and forbs than by trees. Along the lower limits of its distribution, ponderosa pine tends to be associated with rocky and coarsely-textured soils whereas grasslands are associated with more fine-textured soils (Peet 1981). Narrow bands of riparian forests of plains and narrow-leaf cottonwood (*Populus sargentii* and *P. angustifolia*, respectively) penetrate from the Plains upwards into the foothills.

In the montane zone there is typically a striking contrast in stand density and species composition on south- as opposed to north-facing slopes. On xeric, south-facing slopes ponderosa pine forms relatively open stands, sometimes with scattered Rocky Mountain juniper (*Juniperus scopulorum*). Stands on mesic, north-facing slopes are typically much denser and the relative proportion of Douglas-fir is greater. With increasing elevation, stand densities and the abundance of Douglas-fir increases. In the AR, as in the case of the ponderosa pine cover type, the Douglas-fir cover type is limited primarily to the eastern slope of the Front Range (Fig. 3.9b). The upper montane zone (c. 2440 to 2740 m) is distinguished by a gradual increase in the importance of Douglas-fir and by the presence of aspen and lodepole pine, especially towards higher elevations (Marr 1961). Nevertheless, within the upper montane zone stands dominated by ponderosa pine also occur so that this cover type extends over a broad range of abiotic and biotic conditions. The Douglas-fir/mixed conifer cover type generally occurs at higher elevations and at more mesic sites than the ponderosa pine cover type (Fig. 3.9b). Nevertheless, as discussed below, over much of their elevational range these two species are successionally related. Douglas-fir occurs both in relatively pure stands, and mixed with aspen lodgepole pine and ponderosa pine (hence the name “mixed conifer”). The aspen cover type is common in both the upper montane and lower subalpine zone (Fig. 3.9c). It can grow on xeric ridge tops where it occurs as relatively small trees, and it also occurs on mesic sites where it forms dense stands of tall trees. Compared to northwestern Colorado, aspen in the northern Front Range occurs in stands of relatively small extent and accounts for only a small percentage of the vegetation cover (Table 3).

Lodgepole pine forests become increasingly important with elevation in the upper montane and dominate large areas of the subalpine zone (Fig. 3.9d), and are the most extensive forest type of the AR (Table 3.3). The lodgepole pine cover type occurs extensively on both the western and eastern sides of the Front Range. Generally, the most mesic forested sites of the subalpine zone are characterized by the Engelmann spruce-subalpine fir type which is the second most extensive cover type of the AR (Fig. 3.9e). On many sites, the lodgepole pine type is successional to the spruce-fir type. The lodgepole pine type is more prevalent on drier topographic positions and often has sparse understories (Peet 2000). At sites that are too dry and rocky to support dense stands of spruce and fir, these two species form more open, patchy stands often mixed with aspen, limber pine, and lodgepole pine. Bristlecone and limber pine tend to dominate the driest and most wind-exposed sites in the upper montane and subalpine zone, often at sites of conspicuously rocky substrates; sometimes they form sparse woodlands at the highest elevations that will permit tree growth. They account for small percentages of the forest cover of the AR (Table 3.3). Bristlecone pine is limited to the southern part of the AR whereas limber pine has a broader latitudinal range within the AR (Fig. 3.9 f-g). Alpine treeline can be formed either by relatively erect, well-formed stands of conifers or by shrubby krummholz individuals. Krummholz tends to be more common on the eastern slope of the Front Range whereas treeline is more commonly formed by tall subalpine forest on the western side. Above treeline, alpine communities with grass and herbaceous cover are interspersed among rocky areas. Alpine tundra is not recognized as a separate cover type in the RIS data, and is probably encompassed in areas mapped as grassland or as barren/rock. Tundra is beyond the scope of this report on forest ecosystems, and there is substantial information available elsewhere on human impacts in the Front Range (Bowman et al. 2002).

Low-elevation, stream-side forests in the FR are characterized mainly by broadleaf deciduous trees. Common tree species are willows (*Salix* spp.) and plains cottonwood (Populus sargentii) below c. 1950 m and narrow leaf cotton wood (*Populus angustifolia*) at higher elevations (Peet 1981). Towards higher elevations, the riparian forests also include a large component of conifers such as Douglas-fir and spruces (both blue spruce and Engelmann spruce) and quaking aspen (Peet 1981).

On the eastern slope of the Front Range, shrublands are common at low elevations in the foothills and especially in the deep canyon of the Cache la Poudre River (Fig. 3.9h). The common shrub species are Rocky Mountain juniper, mountain mahogany (*Cercocarpus montanus*), skunkbrush (*Rhus triloba*), buckbrush (*Ceanothus fendleri*), ninebark (*Physocarpus monogynus*), and bitterbrush (*Purshia tridentata*). To the west of the continental divide, shrublands are common in North Park and Middle Park along the western edge of AR (Fig. 3.9h). These parks are characterized by expanses of open, rolling grassland and sage (*Artemisia* spp.) brushlands. Grasslands are common in these two large Parks and are scattered throughout the AR, especially at lower elevations (Fig. 3.9i).

3.3.2. Pike-San Isabel N.F.

The PSI also encompasses a range of vegetation types from grassland prairies to alpine tundra within the Southern Rocky Mountain Steppe-Open Woodland-Coniferous Forest-Alpine Meadow Province (Bailey 1995, Bailey 1997). In the PSI, vegetation zonation is controlled primarily by elevation, latitude, topographic position, and exposure (Peet 2000, Allen and Peet 1990, Bailey 1995) and in its broad features is similar to that described for the northern Front Range (Fig. 3.8). Each of these factors plays a major role in the amount, distribution, and persistence of moisture available to vegetation (Peet 2000, Allen and Peet 1990). Elevation also affects temperature that controls growing season length and the amount of freezing that species must be able to tolerate. Temperature and precipitation along the elevation gradient also strongly influence soil properties (e.g., pH and percent base saturation; Allen and Peet 1990). Generally, lower pH and lower percent base saturation correspond to lower temperatures and higher precipitation at higher elevations (Allen and Peet 1990).

Vegetation patterns of the PSI are broadly similar to those described for the AR. Noteworthy differences between the vegetation of the AR and PSI include the presence of gambel oak shrublands and pinyon-juniper woodlands in the PSI, and the absence of white fir (*Abies concolor*) from the AR. In the PSI, three general vegetation zones can be recognized according to elevation: 1) a dry shrubland component with Gambel oak (*Quercus gambelii)* along the lower slopes of the southern Front Range foothills (Powell 1987, Weber 1990); 2) a montane zone that includes ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*) and pinyon pine (*Pinus edulis*) - one-seeded juniper (*Juniperus [Sabina] monosperma*) woodlands; and 3) a subalpine zone that includes Engelmann spruce (*Picea englemannii*) - subalpine fir (*Abies lasiocarpa*) forests, lodgepole pine (*P. contorta*), aspen

(*Populus tremuloides*), limber pine (*Pinus flexilis*), and bristlecone pine (*Pinus aristata*). The most extensive forest cover types are ponderosa pine and Douglas-fir in the montane zone, and spruce-fir, aspen, and lodgepole pine in the subalpine zone (Table 3.4).

Open ponderosa pine woodlands dominate the lower montane zone. The ponderosa pine cover type, once again is more extensive towards the east where it often abuts the Plains grassland (Fig. 3.10a). In the middle montane zone, mixed-conifer assemblages of ponderosa pine, Douglas-fir, limber pine, bristlecone pine, and Rocky Mountain juniper (*Sabina scopulorum*) often occur at relatively xeric sites such as south-facing aspects and in areas transitional to grassland and shrubland. At more mesic sites in the middle to upper montane zone, stands of Douglas-fir are widespread (Fig. 3.10b). These occur both as relatively pure stands and also mixed with ponderosa pine, other conifers, and aspen. White fir (*Abies concolor*) becomes important in the upper montane zone in the southern part of the PSI (Weber 1990, Whitney 1992). In mid-elevation forests in the Sangre de Cristo Range, lodgepole pine and Douglas-fir are associated with acidic soils on north-facing slopes, whereas aspen and white fir are associated with base-rich soils on south-facing slopes (Allen and Peet 1990).

In the subalpine zone, somewhat higher precipitation and cooler growing season temperatures create more mesic habitats. Aspen occurs in both the upper montane and the subalpine zone (Fig. 3.10c). Aspen often forms large monospecific stands in more mesic habitats (Peet 2000), and the aspen cover type is substantially more extensive in the PSI than it is in the AR (Tables 3.3 and 3.4). Lodgepole pine (Fig. 10d) occurs extensively in the subalpine zone overlapping in its distribution with aspen and spruce-fir (Fig. 10e). Generally, the most mesic forested sites are characterized by the Engelmann spruce-subalpine fir type, which is the most widespread cover type of the subalpine zone (Table 3.4; Fig. 3.10e). The lodgepole pine type occurs on sites that are similar to the spruce-fir type and usually is successional to spruce-fir forests. The lodgepole pine type is more prevalent on drier topographic positions (Peet 2000). Mixtures of bristlecone pine, limber pine, lodgepole pine, Engelmann spruce, and subalpine fir interspersed with aspen occur at sites that are generally too dry for development of dense spruce-fir stands. Such mixtures often occur at wind-exposed sites on the edges of large parks dominated by sagebrush and grasses. Limber pine and bristlecone pine tend to dominate the driest, most rocky, and most wind-exposed sites in the upper montane and subalpine zone, sometimes occurring at alpine treeline (Fig. 10f-g). Spruce and fir, often in mixtures with limber and/or bristlecone pine, typically form alpine treeline.

In the southwestern part of Pike N.F. and in San Isabel N.F., piñon appears as a significant vegetation component mixed with juniper in dry, open woodlands (Fig. 3.10h); Powell 1987). Grasslands account for roughly 10% of the vegetation cover of the PSI (Table 3.4). South Park in Pike N.F., in particular, is characterized by expanses of open, rolling sagebrush-grasslands. San Isabel N.F. also contains smaller mountain parks, including the Wet Mountain Valley with grassland and low shrubland vegetation.

3.4. Human Settlement and Land Use

3.4.1. The Native American Period

Colorado's first human residents arrived more than 12,000 years ago, and were hunters who subsisted on megafauna of the eastern Plains of Colorado (Casells 1997, Wyckoff 1999). The Clovis Complex is the oldest of the Paleo-Indian divisons in North America which began over 11,000 years ago and affected large areas of North America, including much of Colorado (Cassells 1997). Archeological evidence of the Clovis Complex has been found at the Dent site near Denver, just east of the FR, and it is likely that this culture also extended into the foothills of the Rockies. This early hunting culture subsisted on large game such as mammoths, bisons, and camels. Overlapping and following the Clovis Complex is the Folsom Complex which also subsisted by hunting megafauna (Casells 1997). It is hypothesized that overhunting by paleoindians coincident with post-glacial climate change is responsible for the extinction of large numbers of taxa of megafauna that were present in North America prior to the Holocene (Martin and Wright 1967). However, the relative roles of hunting and climatic variation in the megafaunal extinction are uncertain and controversial (Steadman and Mead 1995). Whatever the cause of the extinction of the Pleistocene megafauna, it caused early peoples to concentrate hunting activities on the surviving bison herds and smaller game and to increase their plant collecting. Bison hunting continued to be the dominant subsistence strategy in the Plains east of the Rockies right into the 18^{th} and 19^{th} centuries (Wyckoff 1999).

West of the FR, northwestern Colorado was occupied in the 11^{th} century by the Fremont Desert Culture, which fused elements of the older Desert Archaic traditions of the Great Basin with Basketmaker-Pueblo influences from the south (Hughes 1987). By the 11^{th} century, the PSI area was surrounded by the Anasazi agriculturists to the southwest, the Fremont desert culture to the west, and the Plains hunting culture to the east (Wyckoff 1999). All of these groups probably affected the FR, especially the present-day area of the PSI. Their subsistence included cultivation of maize, beans, and squash in combination with collecting of wild plants and hunting of game. By the 13^{th} century, however, the Fremont Desert Culture as well as the

Anazasi Culture of southwestern Colorado mysteriously exited Colorado, and the residents of northwestern Colorado returned to older forms of hunting and gathering without agriculture (Wyckoff 1999).

During the 18th and 19th century, just prior to intensive Euro-American settlement around 1859, the Mouache band of the Ute Indians defended the South Park area as their territory for hunting, gathering roots and berries, and summer living (Hughes 1987, Crum 1996). Their occupancy of this area began perhaps as early as 500 to 1000 years ago. At the time of Euro-American settlement of the area of present-day FR, the Utes occupied most of western and northern Colorado. They are linguistically related to the Paiutes of Utah and the Shosonean peoples of Wyoming. They probably occupied western Colorado since as early as the early 1600s, and they may have evolved from the Desert Culture in c. A.D. 1100 (Casells 1997). The Utes became increasingly mobile with the introduction of horses after 1680 and dominated Colorado's mountainous areas (Wyckoff 1999). Their housing, known as wickiups, consisted of a cone of branches supporting each other or supported by a living tree. Wickiups, occurring singly or in groups of ten or more, can still be found in the forests of Colorado (Casells 1997). Archeological remains at the Roatcap Game Trail site near Paonia in west-central Colorado reveal that they hunted elk, deer, rabbit and bison.

Hunting, plant collecting, and agricultural activities of Native Americans undoubtedly had local impacts on the pre-19th century landscape of the Rocky Mountains, but the extent of these impacts is uncertain. Hunting and gathering activities provided the majority of the Ute's diet, and they only occasionally cropped maize or beans (Hughes 1987, Crum 1996). In the areas of the present-day AR and PSI, Ute modification of the landscape through agricultural practices was probably slight and spatially limited to riparian habitats at low elevation. They collected plants such as yucca, yampa roots, grass seeds, piñon nuts, and berries, but the magnitude of their impacts on these plant populations is unknown. Nevertheless, one disjunct stand of piñon to the north of Fort Collins is believed to be the result of dispersal by Native Americans (Betancourt et al. 1991). The Utes were generally nomadic, and their travels were shaped by the annual cycle of seasons and game movement. They hunted small game throughout their nomadic territory, and following late spring snowmelt, hunting parties ventured into the high Rockies in search of deer, elk, and mountain sheep (Stewart 1942, Wyckoff 1999). Archeological remains, even at high elevations in the subalpine and alpine zones, document at least seasonal use of these habitats by Ute hunters in the northern Front Range (Benedict 1975). The overall impact on the landscape from the mountain-dwelling Utes is not well documented, but early Euro-American observers of the PSI area noted that “trails worn down into solid granite rock on Sawtooth Mountain by ages of travel bear mute testimony of the extensive use of that country by the Indians long before the advent of the white men” (Agee and Cuenin 1924, p. 2).

Fire was potentially the most effective tool for landscape alteration by Native Americans, but the extent to which intentionally set fires increased the area that would have been burned by lightning-ignited fires is impossible to determine. Even the size of the Native American population in the Rocky Mountain region at the time of the earliest contacts with Spaniards in the 16th century is unknown (Thornton 1987). Estimates of the size of the Native American population north of Mexico are highly speculative and cover a wide range (Thornton 1987, Denevan 1992). Simmons (2000) gives a range of 5000 to 10,000 for the maximum Ute population over their entire range in present day Utah, western Colorado, and northern New Mexico at the time of first white contact. Allowing for substantial population decline during the 16th to 19th centuries due to the ravages of introduced European diseases, the Ute population in the FR still was small. Given the low population densities of the Native American inhabitants of the Rocky Mountain region (Thornton 1987, Simmons 2000), it is only through the use of fire that they could have had a spatially extensive impact on the landscape. Hunting was a motive for Native Americans to burn the vegetation of some habitats in the Colorado Rockies. Fire was widely employed by Native Americans in North America as a tool to drive game and to attract game through improved forage (Stewart 1956, 2002, Pyne 1982). Fires also may have been set in the Rocky Mountain region for warfare, communication, clearing of travel routes, clearing of tall vegetation that could conceal enemies around campsites, and to improve grazing for horses (Barrett 1980a, 1980b, Gruell 1985). These uses of fire are derived largely from early historical accounts by Euro-Americans and Native Americans, but how frequently or extensively human-set fires affected particular sites in the FR is unknown.

The most important source of information on the use of fire by 19th century Utes in Colorado is the ethnographic study of Omer Stewart (1942). In the 1930s, Stewart systematically interviewed elders from 14 bands of Utes from Utah, New Mexico, Arizona and Colorado. Eight of the bands formerly hunted in Colorado. These Ute elders reported that during the 19th century their hunting practices included the use of fire to drive antelope, rabbits, deer, and elk. The most commonly reported use of fire (i.e. by 7 of the 8 bands represented by the informants) was for driving rabbits out of thick brush and encircling them (Stewart 1942). Given the distribution of rabbits from the Plains to alpine treeline, such use of fire potentially was widespread in Colorado.

The extent and frequency of fires ignited by the Utes are likely to have varied according to habitat. For example, in grassland and shrubland habitats fuels dry out more easily than in dense forests, and such habitats were favorable to certain game animals (e.g. rabbits, antelope). It is likely that in the montane zone, human-set fires could have easily spread during most years because of consistently dry summers. Human-set fires may have been more frequent near the ecotone with the Plains grassland because the climate was more conducive to fire spread and this area was attractive to Native Americans due to game resources. Locally, in the lower montane habitat Native Americans may have increased fire frequency over the natural frequency permitted by lightning, but the anthropogenic contribution to total area burned was probably small. In the subalpine zone, fire spread is more dependent on exceptional drought so that human-set fires may have burned large areas of subalpine forest only under relatively rare climatic conditions, usually occurring at intervals of several to many decades (Veblen 2000, Sibold et al. submitted). In both the montane and subalpine zone, natural fire ignition by lightning is so common that it seems likely that in years when fuel conditions would support widespread and severe fires additional ignitions by humans probably had only a minor influence.

Intentional burning by Native Americans as a means of harassing white intruders during the 19th century has been widely reported for Colorado. Early settlers in the Front Range and in northwestern Colorado attributed many fires to purposeful or accidental acts of Native Americans, and many of these attributions later were reported as hearsay in government reports on early forest conditions (Jack 1900, Sudworth 1900). Extensive burning in the late 1840s in the Pikes Peak region was attributed by settlers in the 1890s to Native Americans who were attempting to drive the game out prior to the expulsion of the Native Americans from the area (Jack 1900). Both Sudworth (1900) and Jack (1900) noted, however, that early settlers sometimes attributed fires to Native Americans when they themselves had set the fires, and recently Baker (2002) also has questioned the reliability of early settler's reports of fires set by Native Americans. However, we do not have any objective basis for assuming that all early reports of fires set by Native Americans were false.

In northwestern Colorado, an increase in fires set by Native Americans may have resulted from the intensifying struggle between white settlers and Utes for control of the land in northern Colorado in the 1870s (Simmons 2000). During the late 1870s settlers argued that the Utes were burning forests in the Middle Park, North Park and Yampa Valley areas as a hostile gesture against the whites (Wier 1987). In July of 1879, the Governor of Colorado complained to Indian Commissioner E.A. Hayt that off-reservation Utes were burning forests and homes and destroying game in Middle Park and North Park (Simmons 2000). In the Park Range (north of the Gore Range on the west side of North Park in present day Routt N.F.), an extensive fire also was attributed to intentional burning by the Utes in a letter from the Superintendent of a local mining company (U.S.D.A. Forest Service 1972). Yet, the Utes complained to the Governor of Colorado that the charges against them were false, and at least one white observer noted that some of the fires had been started by whites (Wier 1987). George Grinnell, a traveler, passed through North Park and into Middle Park in late summer of 1879. He claimed that some of the fires in North Park were set by the ranchers to clear land of the sagebrush (Wier 1987). Thus, there are claims that both Utes and white settlers set fires in 1879, and, of course, some of the fires may have been ignited by lightning.

Although there is uncertainty about the sources of ignition, there is no doubt that vast areas of forest burned in central and northern Colorado in 1879-1880 as indicated by tree-ring dates of fires (Veblen et al. 2000, Sibold 2001, Bartlett-Howe 2001; Kulakowski and Veblen 2002). Numerous widespread fires occurred in northern and central Colorado, including the FR, in the 1870s and 1880s (Jack 1900, Sudworth 1900, Ingwall 1923; Appendix 1). According to tree-ring records of climatic variation this period included several exceptionally dry years (Cook et al. 1998). The legacy of post-fire stands of aspen, lodgepole pine, and spruce-fir from the fires of 1879 and the 1880s is still a dominant feature of the landscape of northern Colorado (Sibold 2001, Sibold and Veblen unpublished m.s., Kulakowski and Veblen 2002, Bebi et al. 2003).

The most detailed contemporary testimony attributing a large wildfire being set by Native Americans in the Front Range in the 19^{th} century is that of George Frederick Ruxton, an early explorer of the Colorado Rockies. He provides a firsthand account of a large fire he believed had been deliberately set by a group of Native Americans he encountered in south-central Colorado in the area west of present-day Manitou Springs in 1842. The fire was ignited in the early morning on a spring day when lightning would be an unlikely cause. The fire spread rapidly over a distance of at least five miles through “dry pines and cedars” as well as grassland and shrub. Ruxton (Porter and Porter 1950, p. 246) wrote:

> I had from the first no doubt but that the fire was caused by the Indians, who had probably discovered my animals, but, thinking that a large party of hunters might be out,

> had taken advantage of a favourable wind to set fire to the bottom, hoping to secure the horses and mules in the confusion, without the risk of attacking the camp. ...Besides the long sweeping line of the advancing flame, the plateaus on the mountainside, and within the line were burning in every direction, as the squalls and eddies down the gullies drove the fire to all points.

Even in the case of this human-set fire in the montane zone, exceptional drought probably favored its extensive spread. According to tree-ring records, 1842 was one of the driest years in the Front Range over the past several centuries and was also a year of widespread burning (Cook et al. 1998, Donnegan 1999, Sherriff et al. 2001).

Settlers blamed the extensive fires of 1879 and the 1880s on the Utes, but the Utes insisted that careless miners and railroad tie cutters caused them. Federal legislation in the 1870s that halted the logging of living trees may have motivated white settlers to intentionally burn forests to allow salvage logging (Simmons 2000). Thus, persuasive arguments can be made that fires were intentionally set by white settlers or by Utes angry over the loss of their lands. As noted by the early resource professionals (Jack 1900, Sudworth 1900), fires were probably set by both whites and the Utes, and undoubtedly many were ignited naturally by lightning. It is likely that some of the 19th century episodes of extensive burning of subalpine forests resulted from the coincidence of drought with increased anthropogenic ignitions by both white settlers and Native Americans. As discussed in section 5.2.3.2, however, widespread fires in the subalpine zone occur only under conditions of exceptional drought, and the second half of the 19th century was a period of particularly favorable fire weather.

3.4.2. The Euro-American Period: Arapaho-Roosevelt National Forest

Although permanent Euro-American settlement in the Colorado Rockies is usually dated from the mineral discoveries of the late 1850s, the AR region experienced some Euro-American impacts on the landscape prior to this time. Early presence of the Spanish in the northern Front Range is suggested by an 1859 report by a prospector named Samuel Stone (Buchholtz 1983). He reported finding the remains of an old mining camp, including shafts and cabins, near the foot of Longs Peak. French and American fur traders were active in Colorado in the eighteenth century. Groups of Missouri beaver trappers worked the Front Range between 1811 and 1817

(Buchholtz 1983). In 1820 Major Stephen Harriman Long's expedition traveled southwards along the foothills of the Front Range from the South Fork of the Platte with the assistance of French guides. For three decades following the Long expedition, fur trappers continued to visit the Front Range, and transported their furs to trading posts along the South Platte.

Although numerous white hunters, fur trappers, prospectors, and adventurers passed through the northern Front Range and Middle Park in the 1840s and early 1850s (Porter and Porter 1950, Hafen and Hafen 1956, Black 1969), few left written accounts of their observations. The earliest published description of Middle Park (to the west of the AR) was by Thomas Jefferson Farnham, a native Vermonter, who approached the Park from the south, down the valley of the Blue River, with a group of five adventurers. In July 1839 he described Middle Park as (Black 1969, p. 21):

> The valleys that lie upon this stream [the Grand and subsequently the Colorado River] and some of its tributaries are called by the hunters "The Old Park." ... Extensive meadows running up the valleys of the streams, woodlands skirting the mountain bases and dividing the plains, over which the antelope, black and white-tailed [mule] deer, the English hare [jackrabbit], the big horn or mountain sheep, the grisley [sic], grey, red and black bears and the buffalo and elk range–a splendid park indeed; not old, but as new as the first fresh morning of creation.

Farnham's group continued westward in Middle Park where they encountered numerous mountain men with Anglo-American surnames. Thus, it is clear that although numerous whites, mostly mountain men hunters and fur trappers, passed through the region later to become Arapaho and Roosevelt N.F. prior to the late 1850s, it was after the gold rush of 1858-59 that Euro-American impacts in the area accelerated dramatically. During 1859 and 1860, a dozen boom towns such as Denver, Boulder, and Golden grew up overnight. It is estimated that as many as 100,000 people came to Colorado as "Fifty-niners" (Buchholtz 1983). Among the early settlers was Joel Estes who in 1860 claimed Estes Park as an excellent cattle range and set up permanent residence there. He hunted game and sold meat to prospectors too busy to take time to hunt (Buchholtz 1983). Another early settler was Philip Crawshaw, who built a cabin about 1857 near Grand Lake. He trapped beaver along the North Fork of the Colorado River and traded fur for gold dust in Denver (Buchholtz 1983). Elk hunting became the dominant economic activity in the 1860s and 1870s in the Grand Lake area but was supplanted by the short-lived silver boom at Shipler Mountain that spawned the development of Lulu City.

Mining and related activities had major impacts on the landscape of the AR during the second half of the 19th century. Mining greatly altered landforms and drainage courses in the high country (Wyckoff 1999). Hydraulic mining re-configured streams and riparian habitats throughout the mineralized belt of the northern Front Range. Water was essential to mining, and by the 1870s ditches and flumes criss-crossed the high country of the AR. Downstream, discarded gravels were re-deposited in what must have been a dramatic change in riparian and aquatic ecosystems that affected much of the eastern slope of the Front Range. Floods of sediments and mining pollutants must have destroyed vast areas of fisheries during this period. Demand for livestock products by the mining settlements also resulted in heavy grazing in the high country of Colorado after 1859. Large herds of cattle ranged unchecked and uncounted across the Colorado high country throughout the latter half of the 19th century. Over grazing was widely perceived to be degrading the high country of Colorado, and was one of the motivations for creating federal timber reserves in the 1890s to 1905 (Wyckoff 1999).

Widespread timber harvesting began in the AR with the earliest settlements in 1859. In the mineralized belt in Boulder County and adjacent areas mining flourished in the 1860s and 1870s with a sequence of discoveries of gold, silver, and tellurium (Smith 1981). With the mining boom came the development of wagon roads and later railroads. Early in the 1860s an important wagon road was constructed from the mouth of Left Hand Canyon to Ward, then south to Brown’s Crossing (now Nederland), to Rollinsville, and on to Black Hawk (Kemp 1960). Roads traversing the Front Range were constructed through Berthoud Pass and Rollins Pass in the 1860s and 1870s. The Rollins Pass route later was paralleled by the Moffat Railway that ran from 1904 to 1929 over the continental divide until the Moffat Tunnel was completed in 1927 (Smith 1981). By 1883 a narrow-gauge railroad linked Boulder to mining communities in Fourmile Canyon and later to Ward and Eldora (Crossen 1978). Every aspect of the mining operation consumed vast quantities of wood: placer mining stripped away vegetation; hydraulic mining denuded extensive hillsopes; lode operations required large quantities of timber supports; milling and smelting consumed huge quantities of wood fuel, as did the mining towns themselves (Fritz 1933, Kemp 1960, Wyckoff 1999). The effects on the AR forests of the widespread logging that occurred in the second half of the 19th century in relation to the mining booms will be discussed in Chapter 6.

The mining era was also a time of widespread human-set fires. As discussed in section 3.4.1, fires were intentionally set by the Utes during the latter half of the 19th century in response to intrusions by white settlers and miners. Furthermore, many fires also were ignited both accidentally and intentionally by early prospectors and ranchers, and many of these fires burned for several weeks or more spreading over large areas (Ingwall 1923; Appendix 1). Some fires are attributed to carelessness of campers or sparks from locomotives, but others were deliberately set. Some fires were set "merely to clear away the fallen leaves so as to expose the naked rocks to the observation of the prospector" (Tice 1872:123). In 1871 in Boulder there were 51 indictments for illegal forest fires (Tice 1872), suggesting that local authorities were concerned about a high rate of human-set fires. In 1860, a forest fire raged through Gold Hill, destroying the recently founded town (Wolle 1949). The town was nearly destroyed again in 1894 by a large forest fire that started near Ward (Smith 1981) and burned most of the northern slopes of Fourmile Canyon (Goldblum and Veblen 1992). A single fire near the turn of the century is claimed to have burned an estimated 29,000 hectares of forest in the area of Eldora (Kemp 1960) and may be the burn depicted in photographs taken in 1897 and 1898 of the Eldora area (Veblen and Lorenz 1991). As discussed in section 5.3.2, the second half of the 19th century also was characterized by climatic conditions favorable to widespread fire.

Recognizing the need to preserve timbered lands in the west, a reserve system was established by Congress in 1891 (Wyckoff 1999). Roosevelt National Forest originally was part of the Medicine Bow Forest Reserve established in 1897, and in 1910 became the Colorado National Forest. In 1932 it was re-named Roosevelt National Forest. Arapaho National Forest was established in 1908.   The chief goals of the reserve and national forest system were to protect natural resources from over-exploitation and to protect forests from wildfire (Ingwall 1923). Active fire suppression began with the creation of the reserves, but it was not until the early 1910s to 1920s that sufficient personnel and equipment were available to monitor and extinguish fires. With increases in personnel, improvements in transportation infrastructure and water resources, fire suppression probably had greater effect after c. 1920.

3.4.3. The Euro-American Period: Pike-San Isabel National Forest

Prior to the massive permanent Euro-American settlement associated with the mineral discoveries of the late 1850s, such as the Pikes Peak gold rush, the PSI region was not devoid of settlement and probable Euro-American impacts on the landscape prior to this time (Wyckoff 1999). The earliest recorded European incursion into the study area was led by the Spaniard, Juan de Archuleta, who ventured into the Arkansas Valley in the 1660s (Wyckoff 1999). Several Spanish expeditions originating in Santa Fe during the 1700s passed near Pagosa

Springs and reached as far north as the Gunnison Valley, Grand Mesa, and northwestern Colorado. In 1779, Juan Bautista de Anza, then Governor of New Mexico, led an expedition of 600 men against the Comanche chieftain Cuerno Verde (Greenhorn; the source name of today's Greenhorn Mountain)(Calhoun 1956). De Anza's force traveled up the San Luis Valley and crossed the Front Range before heading back to New Mexico.

What is now eastern Colorado to the Continental Divide became part of the US with the 1803 Louisiana Purchase, although there was dispute with Spain and later Mexico over the exact boundaries of the Purchase in the southern regions of the PSI. Zebulon Pike's expedition in1806 was the first official US record of the resources of the Colorado Front Range (Calhoun 1956). In the early 19th century, the "Trapper's Trail" ran from Sante Fe and Taos through the San Luis Valley, up the Front Range, and then north to Ft. Laramie on the North Platte River (McTighe 1984). Due to both the fur trade and the development of the Santa Fe Trail in the 1820s, the Arkansas Valley became an important entryway from the East to both northern New Mexico and the southern Colorado Rockies. Several small trading and agricultural settlements were established in the Arkansas Valley in the 1830s and 1840s (Wyckoff 1999). American buffalo hunters established short-lived posts in the Arkansas Valley in 1829 and the 1830s. Hispanos from Taos hunted deer in the Spanish Peaks and Sangre de Cristo Ranges by the 1830s. During the 1830s and 1840s much of the PSI area was utilized by beaver trappers who were supported by fur trading posts in the Gunnison Valley and Green River on the western slope, and by posts along the Arkansas and South Platte on the eastern slope (Wyckoff 1999).

By the 1850s Hispanic settlers had established themselves where the upper Rio Grande Valley joins the San Luis Valley. Sheep and cattle herds were driven seasonally into the region to take advantage of pastures, and early 19th century travelers reported that more than fifty thousand head of livestock grazed in the valley (Wyckoff 1999). The Sangre de Cristo land grant was awarded in 1843, but threat of Indian attack made occupation sporadic. More permanent settlement of the San Luis Valley began in 1851 (Wyckoff 1999). Thus, Euro-American impacts in the form of grazing and logging for construction and for fuel began along the southern fringes of the PSI and followed two decades later in the northern area around Pike's Peak.

Early European visitors and settlers described the present-day area of South Park as being an abundant hunting and trapping ground with many beaver ponds and vast herds of elk, deer and buffalo (McConnell 1966, Harbour 1982, Wyckoff 1999). Euro-American trappers arrived in the South Park area as early as 1724, intensifying trapping after 1821 (McConnell

1966, Harbour 1982). Beaver pelts were in great demand until about 1845, at which time trapping activity dropped off markedly in South Park.

After the gold rush of 1859, Euro-American impacts in the area accelerated dramatically. With the eastern financial market crisis of 1857, settlers moved west into Colorado, trying their luck at mining gold which had been reported in South Park as early as 1806 (McConnell 1966, Harbour 1982). Discovery of decent pannings at Tarryall near Como in 1859 was followed by the Pike's Peak gold rush. Mining was at first done by simple panning, but was later replaced with sluicing, dredging and other hydraulic means. Watercourses were diverted and river channels were impacted profoundly as evidenced by slag heaps still remaining along drainages. The town of Fairplay quickly became a trading center for miners (Ingwall 1923). In 1861, ranching in the South Park area followed mining, providing cattle and some sheep for the growing markets. Within a period of two decades, large tracts of land were transformed from prairie to pasture and to a minor extent to crop land (Harbour 1982). Unregulated overgrazing was permitted in open forest and plains until about 1908 when fencing was constructed to enforce a permitting system (Ingwall 1923).

Mining and related activities heavily impacted forests in the region during the second half of the 19th century. Timber harvesting began in the PSI with the earliest settlements, but intensified with the growth of mining activities after c. 1859 (Jack 1900, Ingwall 1923). Even after the creation of the Timber Reserves in the 1890s, much illegal logging continued. In the late 1890s, in Plum Creek Reserve, Jack (1900:84) noted that “great quantities of railroad ties have in the past been cut in the reserve and sold to the various railroads having stations within hauling distance. The cutting of ties is still carried on, although only locally and in comparatively small numbers.” In the Manitou Park area, a large-scale timber operation of the park’s “virgin stand of fine western yellow pine and Douglas-fir” occurred in the 1880s. This involved the installation of a sawmill with a daily capacity of 70,000 board feet (running two shifts in 24 hours) and construction of a narrow gauge railroad to haul timber to the mill (Mason 1963). It is not known how much timber was cut, but a settlement for 17.5 million board feet taken from the public domain was made with the government, and the entire cut was estimated at 70 million board feet (Mason 1963). Logging was widespread throughout the PSI region before 1900 (Jack 1900, Ingwall 1923, Agee and Cuenin 1924), and its impacts will be discussed in Chapter 6.

The mining era was also a time of widespread human-set fires in the PSI as described for the AR (Appendix 1). In writing about the Black Forest area of Pikes Peak Reserve, Jack

(1900:66) noted that: “In June, 1896, fires destroyed a considerable portion of the living timber, and there was strong suspicion that the flames were started by lumbermen in order that their operations might come under the provision allowing dead timber to be removed from the Government reserves.”

Writing in 1924, the Forest Supervisor of Cochetopa N.F. (including part of present-day PSI) and the Forest Ranger from Salida commented on the “great many large burns” in the area:

> The various fires which occurred on Sawtooth mountain some thirty or forty years ago have been attributed largely to sheep men setting fire to the timber to improve forage conditions. A number of other fires have been laid to the cow men, who in the early days, would set fire in the spring to the old grass on the larkspur poison areas to keep their stock away from them. ...even in the old Territorial days prior to 1876, there was a law against setting fires on the Public Domain and leaving fires unattended. If a fire once started, it would usually burn for weeks at a time or until there was a rain to put it out,... (Agee and Cuenin 1924:18).

It is likely that the combination of burning, logging and livestock grazing would have had a much more severe and persistent impact on the landscape, than fire alone. These repeated disturbances in the same place may have transformed some forested areas to open vegetation.

Transportation routes quickly developed with the mining activity. Following discovery of gold near Leadville in 1858, a wagon road between Denver and Leadville, via South Park, was built on the trail the Utes used between the plains and the high mountain parks (Ingwall 1923). Several routes branched from this main road, which is currently the principal highway (U.S. 285). The Kansas-Pacific Railway reached Denver in 1870 and was linked to the mining regions across Colorado with a series of short lines (Ingwall 1923). The Denver, South Park and Pacific Railway finally reached the South Park area in 1879 (Ferrell 1981). These routes furnished the infrastructure to extract mineral and timber resources while increasing human population in the formerly remote mountain region. Along with the improved access to forested lands came increased ignition sources from locomotives, settlers, and loggers (Ingwall 1923, Ferrell 1981).

The area of present-day Pike N.F. was incorporated into the timber reserve system in 1892 to address the depredations described by Jack (1900) in his report to the Secretary of the

Interior. In 1905, the Pike's Peak Forest Reserve was created from the combination of the Pikes Peak, Plum Creek, and South Platte forest reserves, and renamed Pike N.F. in 1907 (USDA Forest Service 1989). The chief goals of the reserve and national forest system were to protect natural resources from over-exploitation and to protect forests from wildfire (Ingwall 1923, USDA Forest Service 1989). Active fire suppression began with the creation of the reserves, but it was not until the early 1910s to 1920s that sufficient personnel and equipment were available to monitor and extinguish fires. For example, until 1921, when the Western Aircraft Company contracted with the Forest Service to report forest fires over the Pike region, only a few lookout towers were available for monitoring fire activity (Ingwall 1923). With increases in personnel, improvements in transportation infrastructure and water resources, fire suppression probably had greater effect after c.1920.

4. CLIMATE AND VEGETATION HISTORY

Climatic variation can be a critically important influence on tree recruitment, growth, and mortality patterns (Brubaker 1986, Auclair 1993, Woodward et al. 1995, Savage et al. 1996, Villalba and Veblen 1998, Pedersen 1998) as well as on disturbances such as fire and insect outbreaks (Swetnam and Lynch 1989, 1993, Swetnam and Betancourt 1998). Consequently, we briefly review long-term trends in climate and vegetation history, and then recent (post-1500 A.D.) climatic variability.   Species response to abrupt or gradual climate events must be taken into account when assessing changes that may be the result of human land-use changes. Abrupt climate change, such as a major drought or an anomalous cold period that causes widespread mortality of a species at its environmental limit, can be a cause of major shifts in ecotones (Allen and Breshears 1998). Conversely, climate conditions favorable for plant regeneration may occur more slowly or episodically and lead to lags in re-establishment of an ecotone to some former position. Relatively subtle changes in climatic conditions can have long lasting impacts on vegetation patterns. Consequently, we briefly review climatic variation, especially over the past c. 500 years, as a background to discussing the possible effects of climatic variation on disturbances in Chapter 5.

4.1. Long-term Natural Variability in Vegetation Patterns

Recent human impacts on vegetation patterns need to be viewed in the context of longer-term ecological changes such as those associated with Quaternary climatic fluctuations. Based on fossil insect evidence, mean July temperatures in the southern Rockies during the late glacial maximum (LGM; c. 24,000 to 18,000 years before present) are estimated to have been 9 to 10^{o} C cooler than modern temperatures (Elias 1996). Modern summer temperatures were reached as early as 12,000 years B.P., and warmer than modern summer temperatures prevailed from 10,000 to 3,500 years B.P. (Elias 1996). Fossil pollen evidence from the Southern Rockies suggests that alpine treelines were about 300 m higher than present during 9,000 to 4,500 years B.P. (Fall 1988). Pollen evidence also suggests regional cooling from 4,500 to 3,100 B.P., slight warming between 3,000 to 2,000 yr B.P., and then cooling during 1550 to 1850 A.D. (Fall 1988). These broad shifts in temperatures, and associated fluctuations in precipitation (generally wetter in warmer periods and drier in cold periods), were accompanied by significant changes in vegetation patterns in the Colorado Rockies. In some cases, slow dispersal of species into areas that are climatically suitable for them may have continued to the present (Betancourt 1990).

During the LGM, limber pine was a dominant component of the montane forests over much of the western US, in contrast to its often restricted range today (Betancourt 1990). At the same time during the full glacial, ponderosa pine was restricted in its occurrence, apparently to a few mountain ranges in southern Arizona and Mexico. At the end of the glacial period, dominance of these two species switched, and the range of ponderosa pine expanded to fill the whole of the western US, although limber pine still dominates at low to middle elevations in Idaho, Montana, and Wyoming where ponderosa pine generally is absent. Two hypotheses have been put forth for this switch: 1) a reduction in summer rainfall over the West during the LGM that favored limber pine over ponderosa pine at lower elevations; and 2) changes in fire regimes to high frequency, low severity patterns during the Holocene that have favored ponderosa pine life histories (Betancourt 1990).

Although detailed reconstructions of Holocene vegetation and climate history are not available for the FR, the results of fossil pollen studies conducted in western Colorado in similar ecosystem types as in the FR are indicative of the regional patterns of climate and vegetation history. A long-term (c. 8000 yr) study of fire and vegetation dynamics from near Crested Butte, west of the PSI, documents the importance of historical and climatic factors in vegetation change (Fall 1997). Pollen and charcoal macrofossils were used to document changes in species dominance with changes in climate and in response to assumed changes in fire occurrence. From 8000 to 2600 years ago, the site was surrounded by a subalpine forest with greater dominance of Engelmann spruce than in nearby contemporary forests. Forest composition during this period implies a wet summer climate similar to that of the modern southern Rocky Mountains and Colorado Plateau. Aspen was successional to the spruce-fir forest during this period, although there also appears to have been a subalpine meadow or grassland present between 6400 and 4400 BP. Since 2600 BP, a lodgepole pine forest dominated, likely reflecting drier conditions and more frequent fires.

A paleoecological analysis of a high elevation (3165 m) meadow documents how changing environmental factors have affected vegetation communities for over 10,000 years in northwestern Colorado (Feiler and Anderson 1993). Spruce and fir have been present in this area since glacial retreat, at least 10,500 years before present (BP), and formed open stands mixed with significant non-forest areas of grasses, sagebrush, and ragweed. Over the next several hundred years the forest became more closed as temperatures became cool and wet. This spruce-dominated forest persisted until c. 8,100 BP, when temperature probably rose and precipitation decreased causing a fluctuation in the relative dominance of spruce and fir, an increase in importance of pine, and an opening of the forest community during the next 3,500 years. Temperatures cooled again c. 4,600 BP and remained relatively stable to the present. This period has been characterized by co-dominance of spruce and fir.

On shorter time scales, other climatically-driven changes in vegetation patterns may be the result of lags in the response of species to warmer conditions during the Holocene (Cole 1985, Betancourt 1990, Allen et al. 1998, Feiler 1994). Migrational lags may have important implications for understanding changes at the lowest forest borders in the AR area. The distribution of piñon appears to have been moving north during recent millennia, and apparently has established only recently in woodlands of the Front Range and on the central Great Plains (Betancourt et al. 1991, Allen et al. 1998, Feiler 1994, Swetnam et al. 1999).

Species response to abrupt or gradual climate events must be taken into account when assessing changes that may be the result of human land-use changes. Abrupt climate change, such as a major drought or an anomalous cold period that causes widespread mortality of a species at its environmental limit, can be a cause of major shifts in ecotones (Allen and Breshears 1998). Conversely, climate conditions favorable for plant regeneration may occur more slowly or episodically and lead to lags in re-establishment of an ecotone to some former position. Relatively subtle changes in climatic conditions can have long lasting impacts on vegetation patterns. In the northern Front Range, increased seedling establishment as well as accelerated leader growth of spruce and subalpine fir in the forest-tundra ecotone may have been triggered by a warm but wet period after ca. A.D. 1850 (Hessl and Baker 1997a, 1997b). Thus, there is a reasonably high probability that during the time period of intense human impacts (i.e. post-1850), vegetation conditions in the AR have also been affected by subtle but ecologically significant climatic variations.

4.2. Recent (post-1500 A.D.) Climatic Variability

Patterns of inter-annual climatic variability that may affect ecological processes in the FR are often related to variations in broad-scale ocean-atmosphere interactions such as the North American monsoon and El Niño-Southern Oscillation (ENSO) (Adams and Comrie 1997, Higgins et al. 1998, Carelton et al. 1990). From southern to northern Colorado there is a declining influence of the North American monsoon, as reflected in the south-to-north gradient of decreasing precipitation peaks in July and August associated with the summer monsoon (Donnegan 1999). The periodic occurrence of anomalous oceanic and atmospheric conditions in the tropical Pacific known as ENSO also influences temperature and precipitation conditions in Colorado. Although variable over multi-decadal time scales, the short-term periodicity of ENSO events is approximately 2-6 years (Michaelsen and Thompson 1992, Swetnam and Betancourt 1998). ENSO events exhibit a phase-locking, where positive (cold, Niña) and negative (warm, Niño) events tend to follow one another, with quick shifts in phases occurring typically within 3-4 months (Diaz and Kiladis 1992). During warm El Niño events in the U.S., the mid-latitude winter storm track is typically located further south than during non-El Niño years (Swetnam and Betancourt 1998), steering greater amounts of precipitation, and often warmer temperatures, toward the southern U.S. Broad-scale analyses of the Rocky Mountain region from Idaho to southern Colorado show weak but significant correlations of snowpack to ENSO activity (Changnon et al. 1990). In Colorado, spring precipitation tends to be above average in association with El Niño events and below average in association with La Niña events (Diaz and Kiladis 1992, Woodhouse 1993, Veblen et al. 2000).

The climate of northern Colorado is also teleconnected to warming and cooling of the northern Pacific ocean at decadal time scales (McCabe et al. 2004). This 20-30 year oscillation, known as the Pacific Decadal Oscillation (PDO), affects the climate of a broad area of western North America. In Colorado drought is associated with the negative (cool) phase of the PDO

(McCabe et al. 2004). The PDO has experienced only two full cycles during the 20th century, and future reversals of the PDO phase which could bring climate changes to the western U.S. outside recent memory (Gray et al. 2003). Multi-decadal (e.g. 20 to 30 years) cool and warm phases of sea surface temperatures from the northern Atlantic also have been shown to have a significant impact on the climate of North America (Gray et al. 2004). Following a warm phase of AMO from 1970 to 1990, it appears to have returned to a warm phase beginning in the mid-1990s (Gray et al. 2004). As described in section 5.2.3.2, the tree-ring record of fires in the Colorado Front Range shows significant influences of ENSO, PDO and AMO on the occurrence of years of widespread fires. Thus, changes in these teleconnected multi-decadal drivers of fire weather in the FR may be the source of substantial ecological changes in the next few decades.

At longer time scales, climate has fluctuated throughout the Holocene. In comparison to the past 10,000 years, climate has been relatively stable during the late Holocene (c. 0 AD to present) and has been comparable to that of today. However, the Little Ice Age (c. 1550 to 1850 A.D.) was a source of climatic variation that potentially affected ecosystems of the FR region. This period corresponded to a period of generally cooler and drier conditions terminating in the mid-19th century that affected large parts of North America and other continents. However, because this time period of anomalous climate is not synchronous globally and is characterized by a wide range of climatic anomalies, it is better to refer to it as the 1600 to 1850 period of anomalous climate rather than as a Little Ice Age (Hughes and Funkhouser 1998). In the southern Rocky Mountains, this anomaly is reflected in lower annual precipitation as reconstructed from tree-rings in the Southwest (Grissino-Mayer 1995), the advance of glaciers in the Colorado Front Range (Benedict 1973, Grove 1988), and in fossil pollen assemblages from southwestern Colorado (Fall 1988, Peterson 1994). However, the timing and nature of these climatic variations and how they may have differentially affected high versus low elevation habitats in Colorado are not well resolved. Nevertheless, changes in climatic conditions in the early to mid-19th century may have contributed to ecological changes still evident or in process in the modern landscape. For example, in the northern Front Range, increased seedling establishment as well as accelerated leader growth of spruce and subalpine fir in the forest-tundra ecotone may have been triggered by a warm period after ca. A.D. 1850 (Hessl and Baker 1997a, 1997b).

During the reference period for this assessment of historic range of variability, the Colorado Rockies were affected by severe drought in the mid-19th century as noted by early observers (US House of Representatives 1880, Sudworth 1900, Jack 1900). Several recent tree-ring based reconstructions of droughts and stream flows further document extensive and often severe droughts in the central Rockies east of the FR (Woodhouse 2001, Woodhouse and Brown 2001, Gray et al. 2003). As discussed in Section 5.3.2 the droughts of the late 19th century are strongly reflected in the fire history record of subalpine forests in northern Colorado, including the FR area. Several droughts that occurred over the past 2000 years in the central U.S. appear to have been longer in duration (i.e. multi-decadal) and affected larger areas than those of the 20th century (Woodhouse and Overpeck 1998). For example, the 16th century megadrought far exceeds in severity any droughts during recorded history and affected most of the interior West of the U.S., including the Rocky Mountains (Gray et al. 2003). The mid-19th century drought followed a period of sustained above-average rainfall in eastern Colorado and has been implicated in the decline of bison populations in the western Plains during this period (West 1997). Reports from early travelers on the western Plains in the 1820s to 1840s suggest abundant bison herds that had largely disappeared by the 1850s (e.g. Ruxton in Porter and Porter 1950), well before unrestricted hunting for meat and skins that began in earnest after the Civil War. It is likely that mid-19th century drought exacerbated effects caused by both Native Americans and early European emigrants in the western Plains that led to extirpation of bison from this area (West 1997, Woodhouse et al. 2002). Severe drought in the middle 19th century, beginning about 1842 and extending in some areas into the early to middle 1860s, is a prominent feature of precipitation or drought reconstructions from the southern Plains and Front Range (Woodhouse and Brown 2001), and streamflow on Boulder Creek (Woodhouse 2001; Cook et al. 1998; Fig. 4.1).

Recent (i.e. 20th century) climatic variation in Colorado needs to be considered in relation to global warming. At a global scale there is abundant evidence of warming during the 20th century (Intergovernmental Panel on Climate Change 2000; Crowley 2000). Although global climate models generally agree in predicting further global warming, there are great uncertainties about how climates of particular regions will vary in future decades (Kerr 2000, Couzin 1999). At the broad-scale of the Colorado Rockies (essentially all of western Colorado), analysis of the regional 100-year instrumental climate records indicates no trend in mean annual temperature or precipitation (Kittel et al. 1997). This pattern for the Colorado Rockies contrasts sharply with the patterns for the Rockies of Montana and Wyoming where there are statistically significant trends of warming. In different regions of the Colorado Rockies, temperature and precipitation trends are complex due to variables, which include mountain topography, and possible influences of land use/land cover changes from the Plains through upslope advective transport of water vapor to higher elevations (Chase et al. 1999; Williams et al. 1996). Consequently, potential differences in trends must be considered for different elevational zones and locations relative to the zone of dramatic land use change that has occurred over the past 100 years.

In the foothills-Plains area at the base of the Front Range there appears to have been some cooling since the early 1930s, probably due to irrigation and other land-use changes that have altered latent energy (Chase et al. 1999, Stohlgren et al. 1998). For the period 1952-1997, a gradient of climate stations from the lower montane zone to the alpine tundra in the northern Front Range recorded contrasting trends in mean annual temperatures for different elevational zones (Pepin 2000). Linear regressions of daily maximum and minimum temperatures from 5 weather stations extending from 1509 to 3749 m (4950 to 12,298 ft) yielded the following trends (Pepin 2000): 1) a weak trend towards warmer maxima temperatures at 1509 m; 2) a cooling trend of day-time temperatures at 1509 m; 3) a warming trend of both night and day temperatures at 2195 m (7200 ft); 3) a trend towards increasing maxima and decreasing minima (i.e. increased diurnal range) at 3048 m (9997 ft); and 4) an overall cooling trend at 3749 m. Thus, warming during the 1952-1997 period is concentrated at mid-elevations (2500 to 3000 m; 8200 to 9840 ft), but at 3749 m the trend is reversed. Annual precipitation at the station at 3749 m also has increased during this period (Williams et al. 1996). These results suggest that for annual data there are different trends in climate according to elevation along the eastern slope of the Front Range. Consequently, in summarizing trends in seasonal temperatures and precipitation in the AR, stations are grouped according to elevation and location on the west or east side of the Continental Divide (Fig. 4.2).

To characterize climatic trends in the AR, spring and summer temperatures and precipitation were used instead of annual averages because of the greater influence of growing season climate on ecological processes such as tree growth and fire (Fig. 4.2). Departures (standard deviations) from mean spring (March-May) and summer (June-July) temperature and precipitation were computed for four elevation regions of the northern Colorado Front Range over most of the 20th century: (a) High Elevation East of the Divide; (b) High Elevation West of the Divide; (c) Mid-Elevation on the Eastern Slope; and (d) Piedmont of the Eastern Slope (Fig. 4.2). The 21 precipitation or temperature records from these 13 stations (Table 3.1) were selected as representative of trends in different elevational zones after extensive screening of all weather records from the AR region available from the Colorado Climate Center of Colorado State University and the Mountain Research Station of University of Colorado. Many records were rejected due to high percentages (> 4%) of missing monthly data. To justify grouping together of station records for computation of regional means and departures for elevational zone, the following criteria were considered for each pairwise comparisons of seasonal values: 1) graphical similarity of seasonal variation; 2) significant correlations with all other stations in the elevational zone ($p < .05$; Pearson's correlation); and 3) homogeneity of trends according to a Mann-Kendall statistical test for randomness (Bradley 1976) as determined with the computer program subroutine HOM from the International Tree Ring Data Bank Library. Following homogeneity testing, a regional climate record was constructed by normalizing each station's record as departures from the mean, and then averaging the departures from the mean for each month across all stations through time. A 15-year centered-average smoothing curve shows long-term trends as compared to the mean (horizontal line) for the entire time period (Fig. 4.2).

At high elevations on both the eastern and western sides of the divide there is a conspicuous cooler period in the early 1980s (Fig. 4.2a-b) consistent with studies reporting trends in mean annual temperatures (Williams et al. 1996; Pepin 2000). However, since the early 1980s, spring and summer temperatures have been above the long-term mean for the two high elevation zones. Both zones also show slightly elevated growing-season temperatures during the late 1930s to early 1940s. The low elevation zone records a shift from cool to warmer summer temperatures in the 1930s coinciding with decreased summer precipitation (Fig. 4.2d). This shift from wetter, cooler conditions in the early 1900s to a more arid climate after 1932 has been previously noted and is evident in tree-growth patterns in the Front Range (Graybill 1989) and in streamflows in the upper basin of the Colorado River (Stockton 1976).

To summarize trends in annual precipitation and temperature, a similar analysis of climate stations was conducted for seven stations in or near the PSI (Donnegan 1999; Table 3.2). No significant differences among trends for different stations were noted, and consequently they are presented as an average for all seven stations (Fig. 4.3). Trends for the pre-1900 period primarily reflect the longer record of the Denver station. The dominant feature of the temperature record is the higher than average temperatures during the 1930s to 1960s. This period also was below average in precipitation. The severe drought of 1998 to 2002 is not included in these climate records.

From the perspective of understanding how climatic variation may affect ecological processes such as fire and insect outbreaks, these annual and seasonal patterns of temperature and precipitation indicate that: 1) a strong linear trend in temperature is not evident during the 20th century, when one might look for a local signal of global warming; 2) there has been a great deal of variability in seasonal and annual climate at annual and decadal time scales which would be expected to influence ecological processes; and 3) for high versus low elevations in the FR region, at the scale of individual years to a decade seasonal trends are not always in phase and sometimes show opposite trends. Thus, in interpreting trends in ecological processes such as tree growth, fire, and insect outbreaks which are sensitive to climatic variation, different temporal patterns may sometimes occur for high- versus low-elevation habitats.

Although multi-decadal trends are not prominent in the 20th century climate record of the FR area, consistent with the comprehensive regional analysis by Kittel et al. (1997), at a time-scale of one to four years the variations in temperature and precipitation are often great. Thus, although future trends at a decadal time scale are uncertain but probable, the high degree of short-term variation is a major feature of the climate of the FR area. Despite the difficulty of predicting future climate, climatic variation at both these time scales, as well as at centennial time scales, needs to be considered in resource planning.

During the past several centuries used as a reference period for assessing the historic range of variability of forest conditions in the FR the tree-ring record shows substantial and clear variability at annual, decadal and centennial scales (Gray et al. 2003).   This climatic variability had significant influences on the patterns of disturbance by insects and fire (see Sections 5.2.3.2 and 5.2.4.4) as well as direct influences on tree growth and demography.   Because the tree-ring evidence of the ecological effects of past climatic variation disappears over time, it is often only the more recent episodes of climatically-induced landscape changes that are fully detectable in the population structures and growth patterns of surviving trees. For example, there is abundant evidence from the tree-ring record of climate and fire history that climatic variability favored an increase in burning during the second half of the 19th century which in turn has affected the susceptibility of subalpine forests to recent disturbances by spruce beetle and windstorms (Section 5.4). Thus, during both the reference period and the time period of increased human impacts (i.e. post-1850), vegetation conditions in the FR have been affected by climatic variability. Some of these climatic effects are relatively clear but others may be so subtle that they have not yet been detected. The implication for the goal of this report is that in the following section we must be cautious in evaluating whether a particular vegetation condition can be ascribed primarily to human activities or to climatic variability or to a combination of both.

5. SUCCESSIONAL DYNAMICS AND DISTURBANCE PATTERNS

5.1. Stand Development and Successional Patterns of the Major Cover Types

The most important coarse-scale natural disturbances affecting succession in the Colorado Rockies are fire, insect outbreaks, windstorms, and, in particular habitats, flooding and snow avalanches. Our goal in this section is to provide some background on successional patterns for understanding the consequences of natural disturbances as well as of human impacts through logging, livestock raising, and altered fire regimes. The focus of this section is on characterizing stand development and succession by cover types, whereas in the subsequent section (5.2) the goal is to assess patterns in the major disturbance types.

In general, plant succession is currently recognized as a highly contingent process that varies depending on site and historical conditions (Pickett et al. 1987, Glenn-Lewin et al. 1992, McIntosh 1999). Successional pathways vary temporally depending on "initial conditions", such as site heterogeneity and landscape conditions related to the nature and severity of past disturbances, and long-term climatic variation. This is a shift away from an emphasis on stasis and deterministic successional patterns characteristic of traditional successional theory which continues to be reflected in concepts widely applied by resource managers in the western U.S. (Cook 1996). In modern successional theory, successional pathways are recognized to vary temporally depending on "initial conditions", such as site heterogeneity and landscape conditions related to the nature and severity of past disturbances, and long-term climatic variation. Succession also varies with often steep environmental gradients that occur over short distances in the Rocky Mountains. Slight variations in aspect or topographic position have been shown to result in major differences in stand development and successional pathways (Whipple and Dix 1979, Peet 1981, Veblen 1986a, Veblen and Lorenz 1986, Donnegan and Rebertus 1999). Despite these sources of variability, similarities of life history traits for members of the same species in the southern Rocky Mountains often result in relatively predictable successional patterns.

There are several reviews of stand development and species replacement in the southern Rocky Mountains (Peet 2000, Veblen and Lorenz 1991, Knight 1994), and in this section we will briefly summarize some of the major patterns. We also try to identify any differences in stand development and successional patterns that may exist within a cover type, either due to local (e.g. elevation) habitat variation or geographical variation (e.g. the northern vs. the southern Front Range). In this section we consider some of the widespread hypotheses pertinent to assessing the HRV of each cover type (e.g. ideas such as aspen decline). In subsequent sections on fire (5.2.3) and impacts of land use (6.4) we consider these ideas in more detail. In these subsequent sections, we also consider in greater detail possible explanations for some of the differences reported in studies of the effects of fire history on ponderosa pine forests in the southern and northern parts of the Front Range.

5.1.1 Ponderosa Pine Cover Type

In the context of disturbance ecology and stand dynamics, it is important to distinguish between lower elevations and higher elevations within the ponderosa pine cover type. The lower elevational range of the ponderosa pine cover type corresponds to habitats described in Marr's (1961) classification as the lower montane or foothills zone. Slope aspect and topographic position also differentiate the ponderosa pine cover type into different community types (Peet 1981) but for the sake of simplicity we will generalize the differences into lower vs. higher elevation zones of ponderosa pine-dominated woodlands and forests. The ponderosa pine cover type is especially widespread in the eastern part of the FR at low elevations bordering the Plains grassland (Fig. 3.5a). These lower elevation stands are typically dominated only by ponderosa pine, but may contain small components of Rocky Mountain juniper especially on rocky, more xeric sites whereas Douglas-fir can be found in adjacent mesic, ravines. Below about 2100 m (6888 ft) on xeric sites of the montane zone of the Colorado Front Range, relatively open woodlands of ponderosa pine occur in which age structures suggest that they are self-replacing. Young trees are often very abundant relative to older trees (Peet 1981, Mast et al. 1998). Today, disturbance regimes and stand structures in this habitat have been highly altered by humans, but historical photographs indicate that prior to the 20th century a significant portion of the lower montane landscape consisted of open park-like woodlands (Veblen and Lorenz 1986, 1991).

Increases in ponderosa pine stand densities during the 20th century in the lower montane zone reflect a combination of influences. Disturbances which expose bare mineral soil and reduce competition from herbaceous plants are favorable to the establishment and survival of ponderosa pine seedlings (Pearson 1934, Sackett 1984). Thus, in the FR disturbances such as livestock grazing, logging, road construction, and surface disturbances associated with mining, as well as both high-severity and low-severity fires have been interpreted as resulting in pulses of ponderosa pine establishment (Marr 1961, Veblen and Lorenz 1986, 1991, Mast et al. 1998, Kaufmann et al. 2000, Ehle and Baker 2003, and Sherriff 2004). However, seed production of ponderosa pine is highly variable from year to year so that seeds may not be available in sufficient quantity to allow abundant seedling establishment after all disturbance events (Schubert 1974). Furthermore, in the lower montane zone of the Front Range seedling establishment has been shown to be strongly coincidental with above-average spring moisture availability that in turn is associated with El Niño events (League 2004). Based on determining the germination dates of more than 500 tree seedlings sampled at 10 sites in the lower montane zone of northern Pike N.F. and AR, more than 93% established in only 7 years during the period from 1967 to 1992; over 85% established in just 4 years of above-average spring moisture availability (League 2004). None of these years of exceptional seedling establishment could be linked to fire occurrence at the sample sites. Thus, *for the lower montane zone* episodes of abundant ponderosa pine establishment may depend primarily on the coincidence of years of high seed production with favorable weather, and only secondarily on the availability of open sites created by disturbance. Once established, small juveniles of ponderosa pine will only survive if free from fire for at least 10 years and in most cases probably much longer depending on how site factors influence tree growth rates and fuel accumulation. We stress that all these disturbances, seed availability, and climatic variability, not just fire, influence the establishment success of ponderosa pine seedlings, and, therefore, may contribute to high stand densities in the modern landscape.

Mature ponderosa pine trees are well adapted to survive fire due to their protective, thick bark. In contrast, seedlings and saplings are highly susceptible to death from surface fire until well clear of the understory layer (Peet 1981, 2000). In open, park-like stands of ponderosa pine, overstory trees are rarely killed in surface fires except when local fuel loads are high or wind and/or dry conditions result in fire burning into localized canopy patches. In such stands, occurring mainly at the lowest elevations of the ponderosa pine cover type, reduction of fire frequency during the 20th century is associated with substantial increase in stand density and development of fuel conditions favorable for intensive crown fires (Peet 1981, Veblen and Lorenz 1986, Mast et al. 1998). Based on the reconstruction of historic fire types for an area of c. 61,000 ha on the eastern slope of the FR in the AR, this scenario appears to apply to < 20% of the ponderosa pine cover type over elevations from c. 1800 to 3000 m (4904 to 9840 ft; Sherriff 2004, Sherriff and Veblen unpublished m.s.). The percentage of the ponderosa pine cover type in the PSI for which this scenario is valid is unknown as discussed in later sections of this report.

Pure or relatively, pure stands of ponderosa pine, also occur at higher elevations in Marr's (1961) upper montane zone above 2439 m (8000 ft). Thus, in the upper montane zone the ponderosa pine cover type occurs both as relatively pure stands, and with significant components of Douglas-fir. The upper montane zone also includes the Douglas-fir or mixed conifer cover type discussed below. In the upper montane zone of the FR a large component of the ponderosa pine cover type consist of dense stands that originated after severe stand-replacing fires (see section 2.3 for definitions of fire parameter terms) as discussed in section 5.2.3.1. In the northern Front Range, age structure data from widespread montane stands that originated after the fires of the late-19th century clearly show the ability of ponderosa pine to form dense post-fire cohorts (Veblen and Lorenz 1986, Mast et al. 1998, Ehle and Baker 2003). Bell-shaped age frequency distributions are characteristic of relatively shade-intolerant tree species, such as ponderosa pine, when a major disturbance such as fire creates a sudden opportunity for new recruitment (Oliver and Larson 1990). Age frequency distributions of ponderosa pine in stands burned in the late 1800s or early 1900s in the AR indicate that following a fire, the recruitment period is typically 30 to 60 years, although this would be expected to vary with site conditions, seed sources, and perhaps with climate (Veblen and Lorenz 1986). Similar bell-shaped age frequency distributions of ponderosa pine are found for the pre-Euro-American settlement period in the northern Front Range, which have been interpreted as responses to earlier stand-replacing fires and supported by fire scars pre-dating the regeneration event (Mast et al. 1998, Ehle 2001). In stands that were relatively dense at the time of the fire, growth releases (increased radial growth) often occur on ponderosa pine or Douglas-fir that survive fires if the neighboring trees killed by the fire had been suppressing their growth through competitive interactions (Goldblum and Veblen 1992). Such growth releases may occur one to a few years after the fire or may lag 10 or more years depending on factors such as damage to the individual tree and the time required for dead-standing trees to decay and fall over (Goldblum and Veblen 1992).

Where Douglas-fir and ponderosa pine co-occur in the same stand, as they typically do in the upper montane zone, there is a tendency for the former to slowly, successionally replace the latter (Peet 1981). In relatively old post-fire stands, young Douglas-fir are typically present whereas juveniles of the shade-intolerant ponderosa pine are typically absent. This successional pattern is due primarily to the differences in shade tolerance of the two dominant species in stands that originated following a stand-replacing fire. For the AR, age structures and fire-scar records do not indicate that such stands were formerly characterized by frequent surface fires (Veblen and Lorenz 1986, Veblen et al. 2000, Sherriff 2004, Sherriff and Veblen unpublished m.s.). Thus, decreased fire occurrence during the 20th century should not be interpreted to mean that Douglas-fir are invading stands from which they were formerly absent. Rather, prior to 20th century fire suppression, both Douglas-fir and ponderosa pine would have been present in young post-fire stands at such sites following stand-replacing fires.

In the montane zone of Pike N.F. ponderosa pine stand structures and regeneration patterns similar to those described for the mid and upper montane zone of the northern Front have been found in the Cheesman Lake area (Kaufmann et al. 2000). In the Cheesman Lake area, four major vegetation structures are recognized: 1) forest patches with all trees post-dating certain years, probably indicating past stand-replacing fires; 2) patches in which tree ages do not reflect a past major stand-replacing fire; 3) riparian systems; and 4) openings and areas of very low forest density, some of which are interpreted to be due to lack of tree regeneration following pre-20th century severe fires. The proportion of the landscape with tree age structures suggesting past stand-replacing fires was estimated at 75% (Kaufmann et al. 2000). The area of the riparian landscape was described as "limited" and the area represented by old-growth stands was described as "elusive." Openings are interpreted as mostly transient rather than permanent because of the presence of remnant trees. As summarized from Kaufmann et al. (2000) these findings for the Cheesman Lake landscape are consistent with the literature on the montane zone of the northern Front Range (Veblen and Lorenz 1986, Mast et al. 1998, Veblen et al. 2000, Ehle and Baker 2003, Sherriff 2004, Sherriff and Veblen unpublished). Most significantly, there is general congruence among these studies that in the ponderosa pine cover type in the Front Range, stand-replacing fires played the dominant role in shaping the structures of these forests. For example, in the Cheesman Lake study area, Kaufmann et al. (2000) note that there is "no evidence of frequent surface fires in any one location (e.g. multiple fire scars a few years apart on single trees)..." This is similar to findings in the northern Front Range, except for the lowest elevations accounting for approximately 20% of the ponderosa pine cover type.

Despite the broad similarities between the findings of the Cheesman Lake studies (Kaufmann et al. 2000, 2001, Huckaby et al. 2001) and those of the mid and upper montane (i.e. above c. 2200 m) zone of the northern Front Range there are some differences among the interpretations of these studies (Veblen and Lorenz 1986, Mast et al. 1998, Veblen et al. 2000, Ehle and Baker 2003, Sherriff 2004, Sherriff and Veblen unpublished). The main discrepancies are the following conclusions for the Cheesman Lake study area by Kaufmann et al. (2000, 2001) and Huckaby et al. (2001): 1) They note that insect outbreaks did not reach epidemic proportions during the 20th century, and believe that outbreaks were also lacking in the pre-historic landscape as well. 2) By applying the Forest Vegetation Simulator model, they conclude that in 1900 more than 90% of the landscape had a crown closure of 30% or less in comparison with 55% in 1996. And, 3) they conclude that during the 1900s many pure ponderosa pine patches were converted to mixed patches through ingrowth of Douglas-fir. In contrast, outbreaks of mountain pine beetle, Douglas-fir bark beetle, and western spruce budworm reached epidemic proportions, killing many trees in the montane zone of the northern Front Range (and in areas of the PSI adjacent to the Cheesman Lake area) during the 20th century as well as preceding centuries and throughout the montane zone of the AR (see section 5.2.4). In the northern Front Range, in the absence of moderately frequent surface fires to maintain stands in a relatively open conditions, it seems unlikely that 90% of the landscape would have had crown closures less than 30%. Furthermore, such a high percentage of open stands is not consistent with historical photographs and stand structural data for the mid and upper montane zone of the northern Front Range (Veblen and Lorenz 1986, 1991, Mast et al. 1998, Sherriff 2004). Similarly, the hypothesis that pure ponderosa pine stands converted to mixtures with Douglas-fir has been examined and not supported on the basis of tree ages from the northern Front Range (Sherriff 2004). Thus, Kaufmann et al.'s (2001) conclusion that the current amount of Douglas-fir is "too high" (i.e. in comparison with the historic landscape) in the Cheesman Lake study area is not supported by the extensive tree age data from the northern Front Range (Sherriff 2004). Some of these discrepancies may be real differences in the abiotic (especially topography) and biotic environments between the southern and northern Front Range montane zones, and others may be due to differences in the interpretation of data on past forest conditions.

Although numerous studies of fire history and stand age structures indicate that in the mid and upper montane zone of the FR ponderosa pine often establishes abundantly after a severe fire, the same non-fire factors as discussed for the lower montane zone influence tree regeneration. Thus, in addition to fires, disturbance by livestock, mining, and road construction during the late 19th century and subsequently may have created opportunities for tree establishment. Seed availability clearly is a limitation on establishment of ponderosa pine, and this limitation may result either from annual variation in seed crops or destruction of seed trees by severe or repeated fires. It is likely that climatic variation following the creation of bare sites by disturbances also plays a role in the success of ponderosa pine establishment in the upper montane zone as it does in the lower montane zone (League 2004). However, under the slightly more mesic conditions of the upper montane zone, it is logical to expect moisture availability to be less limiting than it is in the lower montane zone.

We stress that in broad terms past fire regimes and forest conditions are similar for the

Cheesman Lake and northern Front Range studies of ponderosa pine forests. In particular, all recent studies of Front Range montane forests have found an important role for past stand-replacing fires, and have noted that the proportions of the landscape in stages of post-fire recovery would have varied over time in response to the episodic occurrence of widespread, severe fires. Some of the differences noted above may be due to assumptions used in the Forest Vegetation Simulator model to generate past forest conditions in 1900. However, given the recognition of the importance of widespread, severe fires in the late 1800s, reconstruction of conditions for 1900, a single point in time, would only capture part of the range of variability in forest conditions typical of the past several centuries.

5.1.2. Douglas-fir / Mixed Conifer Cover Type

At increasing elevation in the montane zone, the density of ponderosa pine-dominated stands increases, and on more mesic sites at higher elevations and/or on north-facing aspects it co-occurs with Douglas-fir and grades into the Douglas-fir or mixed conifer cover types (Fig. 3.5b). Where there are significant proportions of limber pine and/or lodgepole pine present, these stands are usually classified as the mixed conifer type. Large areas of relatively pure Douglas-fir are common in the PSI, such as in the Rampart Range, but are substantially less abundant in the AR (Tables 3.3 and 3.4). Lodgepole pine is an important component of this type only towards higher elevations, near the transition to the subalpine zone, and its ecology will be discussed as part of the subalpine cover type discussion. In the mixed conifer type, limber pine occurs at especially rocky, xeric sites. The following discussion is pertinent primarily to sites where Douglas-fir potentially can dominate the site or co-dominates the site mainly with ponderosa pine.

Douglas-fir is also thick-barked and highly resistant to fire as a mature tree although seedlings and saplings are easily killed by fire. Soil disturbances, such as those associated with fire, logging, and mining, create bare sites favorable to tree establishment. Following crown fires, both species regenerate to establish even-aged populations (Peet 1981, Veblen and Lorenz 1986). As stands form closed canopies, the less shade-tolerant ponderosa pine regenerates less successfully than the more shade-tolerant Douglas-fir. As time-since-last fire increases, so does relative dominance by Douglas-fir. However, once a crown fire triggers this pattern of successional development, subsequent disturbance by surface fires would be expected to retard the replacement of ponderosa pine by Douglas-fir by killing seedlings of both species. However, the AR age structures and fire-scar records do not indicate that surface fires played an important role in Douglas-fir dominated stands in the upper montane zone (Veblen et al. 2000, Sherriff 2004, Sherriff and Veblen unpublished m.s.). Instead, in the upper montane zone stand structures have been shaped mainly be stand-replacing or partially stand-replacing fires occurring at intervals usually much greater than 50 years in the same stands. In stands above c. 2200 m (7216 ft), trees with more than 3 fire scars are rare in search areas of c. 100 ha, even when there are trees present that show one or two fire scars older than c. 1800 A.D.. The long intervals between fires would have allowed fuels to accumulate and develop a laddered structure favoring crown fires. This is also consistent with growth releases indicating release of trees from competition (suppression) from canopy trees and the common pattern of post-fire cohorts of Douglas-fir (Veblen and Lorenz 1986, Goldblum and Veblen 1992, Sherriff 2004).

Mortality caused by outbreaks of insects can either retard or accelerate succession, depending on which tree species is the host (Hadley and Veblen 1993). Thus, mountain pine beetle outbreaks that kill ponderosa pine can accelerate succession from ponderosa pine towards Douglas-fir. In contrast, mortality of Douglas-fir caused either by western spruce budworm or Douglas-fir bark beetle will tend to retard successional replacement of ponderosa pine by Douglas-fir.

In the montane zone, there is much variability of successional pathways due to spatial heterogeneity related mainly to topography (Peet 2000). Depending on site conditions and seed availability, pure, dense stands of either ponderosa pine or Douglas-fir can develop. For example, in the northern Front Range, pure stands of Douglas-fir can form in steep ravines and north-facing slopes below 2000 m (6560 ft) although on open slopes it is restricted mainly to elevations above 2300 m (7544 ft). At drier, rocky sites limber pine may co-occur with ponderosa pine and Douglas-fir, and at such sites stand-replacing fires may result in conditions favorable for the regeneration of limber pine (Veblen and Lorenz 1986). Towards higher elevation, lodgepole pine co-occurs with Douglas-fir but in post-fire stand development it is gradually replaced by the latter species. Although there is a tendency for Douglas-fir to successionally replace ponderosa pine in most co-dominated stands, evidence of old trees that survived partially stand-replacing fires and the species composition of old stumps and snags indicate that the same species which dominated the site prior to a fire are the dominants of the site for at least a century following the fire (Veblen and Lorenz 1986). Return to the same dominant species following stand-replacing fires appears to be the typical pattern in the montane zone.

5.1.3 Shrubland Cover Types

In the RIS data, areas classified as shrubland on a physiognomic basis include substantial variation in species composition. In the northern FR, shrublands dominated by mountain mahogany (*Cercocarpus montanus*) and skunkbrush (*Rhus triloba*) form open, patchy communities at dry sites near the ecotone of ponderosa pine woodlands and the Plains grasslands (Peet 1981). Small Rocky Mountain junipers typically occur in small numbers in this shrubland type. Bitterbrush (Purshia tridentata), an important browse species for ungulates, is an important component of this community at more mesic sites, as is ninebark (*Physocarpus monogynus*) towards higher elevations (Peet 1981). Mountain mahogany communities in Wyoming have received a modest amount of research attention (Knight 1994), but in Colorado detailed studies of their ecology are apparently lacking.

Although not distinguished as a separate shrubland type in the RIS data, gambel oak (*Quercus gambelii*) is the dominant shrub species in a large proportion of the xeric shrublands at the base of the eastern foothills of the PSI area (Peet 1981).   Over its range in the western U.S., gambel oak occurs at elevations of c. 1,000 to 3,000 m (3,300 to 9,900 ft) often in an elevational zone between pinyon-juniper woodlands and aspen forests (Harper et al. 1985). Gambel oak normally occurs as a tall shrub or short tree in the PSI. This oak is sensitive to drought (Neilson and Wullstein 1983) and its distribution is probably limited at lower elevations by moisture stress and at higher elevations by competition. In Colorado, reproduction of Gambel oak is usually vegetative, resprouting from rhizomes, especially after fire (Harper et al. 1985). Following fire, the oak sprouts vigorously, grows quickly, and therefore leads to rapid recovery of the community (within c. < 20 years following the fire). In western Colorado, annual growth of Gambel oak has been shown to increase following fire and remain above pre-fire levels for at least 10 years following the disturbance Kufeld (1983). Fire is sometimes used to open tall dense stands of oak shrublands to make them more accessible and to improve forage for elk, deer and cattle (Kufeld 1983). Following intense fire in pinyon-juniper woodlands in western Colorado, Gambel oak may initially establish from existing rhizomes and eventually be succeeded by pinyon pine (Floyd et al. 2000).

Generally, little is known about the disturbance history and successional dynamics of Gambel oak communities. This lack of information makes an assessment of HRV difficult and any such assessment can only be tentative and approximate. This lack of information also points to a need for research to be conducted in these communities. For the sake of the present assessment, we will draw on the few studies that have been conducted on Gambel oak disturbance regimes, even though they were located outside of the PSI.

Fire has long been considered the most important natural disturbance process in Gambel oak communities (Brown 1958). Gambel oak occurs in several types of communities including oak-dominated shrubland and ponderosa pine-oak woodlands. Each of these communities is likely to have had a unique pre-settlement natural fire regime. In Mesa Verde National Park (MVNP) in southwestern Colorado, the pre-settlement fire regime of oak-dominated shrubland has been described as one of moderate-size (c. 800 - 2000 ha) stand-replacing fires (Floyd et al. 2000). A combination of fire suppression, grazing, and wet weather was probably responsible for a reduced occurrence of fire in the early 20th century. However, the fire regime in the latter half of the 20th century was again characterized by moderate-size stand-replacing fires. While several factors including fuels and topography affect fire regimes, the fire regime of MVNP was primarily controlled by weather (Floyd et al. 2000). These fires occurred during extremely dry summers, when it was difficult to impossible to control the fires even with modern fire-fighting technologies. Given this type of fire regime in which severe and extensive fires occur infrequently in association with unusual fire-promoting weather, it is unlikely that the 20th century suppression of fires during years of less extreme fire weather has greatly altered the natural fire regime.

5.1.4. Pinyon-Juniper Woodland Cover Type

Although absent from the AR, pinyon-juniper woodlands are a significant part of the landscape the San Isabel component of the PSI where they account for over 5% of the vegetated area (Table 3.4). Within the PSI, the pinyon-juniper cover type is especially widespread in the western part of the Forest (Figure 3.4). A major caveat regarding discussion of pinyon-juniper woodlands is that several different pinyon-juniper woodland types exist. For example, the pinyon - juniper in the Great Basin is composed of a different species of pinyon and juniper than what is found in the PSI (Evans 1988). The pinyon in the Great Basin is a single needle pine that has evolved to withstand much drier conditions due to a moisture regime that is quite different than that of the PSI. Thus pinyon-juniper woodlands in different climatic regions are likely to be ecologically different and therefore are likely to experience different disturbance regimes. However, due to the dearth of studies within the PSI, we must draw on data from other physiographic regions. We do so cautiously, with explicit urging that local research must be conducted on the PSI in order to describe the HRV of this National Forest.

Fire was historically the most important disturbance in pinyon-juniper woodlands prior to the introduction of heavy livestock grazing. Fires in pinyon-juniper woodlands can be either severe and stand-replacing in dense stands (Wright and Bailey 1982) or can be less severe or patchy on less productive and open sites (Johnsen 1962). A recent study of pinyon-juniper woodlands in western Colorado on the Uncompahgre Plateau and in the Gunnison Gorge and Black Canyon of Gunnison National Park, used age structures to show that many woodlands historically originated following infrequent and severe fires (Eisenhart 2004). Many other woodlands show no evidence of severe fire (Eisenhart 2004). These latter stands can be relatively old (100 - 1000 years) and can form self-replacing stands. Research from Mesa Verde NP shows that in old-growth pinyon-juniper woodlands juniper is longer lived while pinyon experiences higher rates of mortality (Floyd et al. 2003). Increases in stand densities are expected as an inherent feature of slow stand development in pinyon-juniper woodlands and do not necessarily reflect suppression of surface fires (Eisenhart 2004). Tree establishment within pinyon-juniper stands can be episodic and may be related to climatic variability. Climatic variability, at all temporal scales, has been important in influencing the distribution and density of pinyon-juniper woodlands across its range (Betancourt 1987; Betancourt et al. 1993; Tausch 1999). A regional-scale drought at the turn of the 21st century is likely to be contributing to a widespread and ongoing die-back of pinyon in parts of southwestern Colorado (Ogle 2001). Shade has also been suggested to have an important influence on seedling survival (Meagher 1943). Thus, disturbance that leaves coarse debris may hasten re-establishment of trees and may therefore have a significant influence on post-disturbance dynamics (Eisenhart 2004).

Overgrazing by livestock can reduce herbaceous ground cover, therefore altering the fire regime by reducing fuel continuity. The reduction in herbaceous cover can also reduce competition and therefore favor establishment and growth of pinyon and juniper (Gottfried et al. 1995). Thus, livestock grazing in pinyon-juniper woodlands is likely to have had a significant influence on their structure.

5.1.5 Quaking Aspen Cover Type

Southwards along the Front Range and into the Sangre de Cristo Mountains of New Mexico, quaking aspen stands tend to occupy a larger percentage of the landscape. Thus, in the PSI aspen accounts for more than twice the percentage of the vegetation that it does in the AR (Table 3.4). Northward in the FR, lodgepole pine occupies a larger portion of the range of habitat that potentially could be occupied by aspen (Peet 2000).

Quaking aspen can occupy a broad range of habitat types, varying from relatively xeric sites at lower elevation to more mesic ones at higher elevation (Jones 1985). Aspen is similar to lodgepole pine in its ability to dominate early post-fire stand development but it has substantially different life history traits. The underground portion of aspen clones is extremely long-lived, and it has been proposed to be among the oldest living organisms (Mitton and Grant 1996). Aspen have the capacity to reproduce either sexually or asexually, but the predominant mode of reproduction is asexual. Sexual reproduction in this species is thought to be exceedingly rare, with large-scale sexual establishment possibly occurring only once every 200 to 400 years (Romme et al. 1997). Aspen primarily regenerates through asexual vegetative shoots (suckers), which arise from long lateral roots most commonly in response to damage by fire or other disturbances (Schier et al. 1985). Following fire, aspen resprouts from underground rhizomes and produces abundant, rapidly growing root suckers that favor the initial dominance of aspen (Peet 2000). Reproduction of aspen in the FR appears to be overwhelmingly from suckering, but observations after the 1988 Yellowstone fires suggest that successful seed reproduction may depend on the coincidence of a wet spring, an absence of intense grazing, and good seed production during the year after a fire (Romme et al. 1997). Aspen ramets (stems from root suckers) are often short-lived, rarely exceeding 200 years of age, so that the longer-lived conifers may eventually take over dominance of post-fire stands in certain areas. In such cases, the conifers often establish at a later stage of stand development, but sometimes may initially co-dominate the site along with aspen.

Aspen stands typically have open canopies with relatively high light levels in the understory. Thus, spruce, subalpine fir, and lodgepole pine are able to establish and grow up through the aspen canopy (Peet 1981). The successional status of aspen is highly variable, both locally in relation to habitat and site history, and over broad geographical distances such as between the Front Range and Yellowstone N.P. In Colorado, aspen may either be seral to conifers or it may form self-replacing stands with little or no presence of conifers (Langenheim 1962, Peet 2000, McKenzie 2001, Kulakowski et al. 2004, Kurzel 2004). Although aspen is seral to conifers in some habitats, some aspen stands are self-maintaining (Langenheim 1962; Mueggler 1985, Peet 2000, Kurzel 2004, Kulakowski et al. 2004, Kulakowski, Veblen, and Kurzel, unpublished manuscript).

At a local scale, the successional status of aspen appears to be related to elevation and site history. Lower elevations that are transitional from steppe or shrubland to coniferous forest in the Southern Rockies are more likely to support self-replacing stands of aspen (Mueggler 1985, Peet 2000, Romme et al. 2001). Such stands are often exclusively dominated by aspen.

In western Colorado, aspen at higher elevations aspen tend to be seral to the conifers, particularly at sites where pre-disturbance vegetation was dominated by spruce and fir (Langenheim 1962, Kulakowski et al. 2004). Within its upper elevational range, aspen's seral vs. self-replacing status varies spatially, but aspen tends to maintain its seral status at a given site following disturbance (Kulakowski et al. 2004). At some sites in both Grand Mesa NF and the Flat Tops area of White River N.F., old (i.e. > 120 years) aspen stands appear to die back and then reproduce in a single recruitment phase forming a new relatively even-aged cohort even in the absence of coarse-scale disturbance (Kurzel 2004). Thus, although moribund populations of canopy-sized aspen in association with understories of conifers clearly indicate successional replacement of aspen, not all stands with dead or dying aspen dominants are successional to conifers. On favorable sites, aspen and the associated understory species may enhance the development of mollisols, which favor aspen regeneration and also have a higher pH, more organic matter, and a greater cation exchange capacity, all of which make a site less suitable for conifer growth (Cryer and Murray 1992; Jones and DeByle 1985). Early in the 20th century, Baker (1925) suggested that repeated fires could maintain self-replacing aspen stands by eliminating seed sources for the conifers while promoting the suckering of aspen.

At a broad geographical scale in Colorado there are some general trends in the successional status of aspen across the Rockies from the Front Range to northwestern Colorado. Based on studies conducted in Grand Mesa, Gunnison, White River and Routt National Forests, large parts of its range in northwestern Colorado aspen forms self-replacing stands (Kulakowski et al. 2004, Kurzel 2004). For example, in a 120,000 ha study area in Grand Mesa and western Gunnison National Forests, comparison of modern and historical (1898) vegetation maps as well as age structure studies have shown that the extent of self-replacing aspen is greater than the extent of seral aspen (Kulakowski et al. 2004). In contrast, in the Front Range, the percentage of aspen stands that are self-replacing appears to be relatively small compared to the large percentage of the landscape in which conifers appear to be capable of replacing aspen (Peet 1981, Kashian et al. 2004). Differences between the environments of northwestern Colorado and the Front Range which potentially may contribute to the larger proportion of self-replacing aspen stands in the northwest include differences in geology and soils, climate, and associated tree species. For example, soils in the Front Range tend to be more coarsely textured and infertile in comparison with Grand Mesa and the Flat Tops area of White River N.F. Snowpacks are less and spring melt occurs earlier on the eastern slope of the Front Range. And, the relative importance of lodgepole pine is much greater in the Front Range than in western parts of White River N.F. and Grand Mesa N.F. However, the roles of these and possible other factors in accounting for the greater percentage of seral aspen stands in the Front Range have not been systematically examined (Kashian et al. 2004).

Based on observations of aspen stands mainly in the area of Yellowstone National Park, Kay (1997) has provocatively argued that fire suppression and browsing by populations of native ungulates are “dooming” aspen. The ability of native herbivores to seriously affect the regeneration of aspen is well documented at the scale of individual stands (DeByle 1985), but it is often difficult to assess their impacts over larger areas. Studies in Rocky Mountain National Park in the northern Front Range have demonstrated that: 1) heavy use of aspen stands by elk can seriously impede aspen regeneration (Baker et al. 1997); 2) that the degree of elk impact on aspen regeneration is highly variable over larger areas (Suzuki et al. 1999); and 3) that for a large proportion of the Park, aspen is represented by young stems suggesting adequate regeneration (Kaye 2002). To predict the long-term sustainability of aspen on the eastern slope of the northern Front Range (mainly in the AR), an assessment sponsored by the Forest Service was initiated in 2003 (Kashian et al. 2004). Aspen successional trends were assessed in 92 stands located across the elevational range of aspen. Only 19% of the aspen stands had age distributions indicating a lack of recent recruitment and potential decline; these stands tended to be located at lower elevations where elk browse is heavier and fire suppression is more effective (Kashian et al. 2004). Multiple cohorts indicating self-replacement were found in 38% of the stands sampled. In 40% of the stands, aspen was interpreted to be seral to conifers, and most often these stands were at high elevation. Overall, this assessment of aspen concluded that “although some aspen stands are declining in the northern Front Range, the presence of multiple successional trends makes it unlikely that aspen is in danger of disappearing from the region” (Kashian et al. 2004).

Because aspen is seral to conifers over a significant portion of its distribution, the relative abundance of aspen at a landscape scale is likely to be sensitive to any major changes in fire regimes. The occurrence of extensive and severe fires are likely to increase the extent of aspen, while a cessation of burning may allow the successional replacement of aspen by conifers in particular habitats. In the subalpine zone of the FR and nearby areas, based largely on data collected in forests dominated by lodgepole pine, Engelmann spruce and subalpine fir, fire return intervals to the same study area or stand of a few thousand hectares have been estimated to be well over 100 years and in some cases over 500 years (Sibold 2001, Sibold et al. unpublished m.s., Romme and Knight 1981, Kipfmueller and Baker 2000). Although fire history studies have not been conducted in areas of exclusively aspen dominance, it is highly likely that fire return intervals are not greatly different in aspen stands from the surrounding coniferous forest. Given the long interval between occurrences of widespread fires in the historic fire regime, it is unlikely that fire suppression over the past c. 80 years has resulted in conditions that are outside of HRV. Nevertheless, if stand-replacing fires are eliminated from the FR, the amount of aspen in the landscape will likely be reduced in the long term.

For the AR, the number of aspen stands included in the RIS database is too small to draw definitive conclusions about the ages of aspen over a larger area. However, this small data set indicates a lack of aspen stands originating earlier than 1860 A.D. (Fig. 6.2). The preponderance of relatively young aspen stands in this AR data set is consistent with the idea that most stands have originated after the period of widespread burning in the late 19th century and following logging operations in the mid-20th century. We caution that the size of this data set is too small to draw firm conclusions. However, based on 92 stands sampled in the northern Front Range, Kashian et al. (2004) found that nearly 70% included ramets 100 to 140 years old which again implies a major episode of regeneration following the widespread burning of the second half of the 19th century. For the PSI, the number of aspen stand ages in the RIS data set is substantially greater but still not large enough for a robust interpretation. In the PSI, aspen stand ages are greater, with some stands originating in the early 1800s (Fig. 6.2). There are peaks in aspen stand ages which may reflect the widespread fires of the second half of the 19th century and regeneration after recent (post-1960) logging (Fig. 6.2). Thus, for both the AR and PSI most aspen stands are relatively young and many date from episodes of widespread burning in the late 19th century or logging in the 20th century. Although the sample size for aspen stand ages is small, it does not support the view that aspen is not regenerating due to fire suppression. The relatively large proportion of young aspen stands, including many established since 1950, does not support the view that aspen is in decline in the FR.

5.1.6 Lodgepole Pine Cover Type

In the subalpine zone and upper montane zone of the FR, stand-replacing fires often result in new stands dominated by lodgepole pine (Whipple and Dix 1979, Peet 1981, Veblen and Lorenz 1986). Lodgepole pine, often viewed as the archetypal post-fire species, establishes from large quantities of seed released by serotinous cones and initially grows relatively rapidly on sites of favorable habitat. Thus, the amount of lodgepole pine in FR may have increased following the extensive and severe burning during the late 19th century. There is a high degree of variability in the percentage of trees with non-serotinous cones that appears to be linked to disturbance history (Muir and Lotan 1985, Schoennagel et al. 2003). Older stands tend to have higher percentages of open cones. Following fire at less favorable sites or where seed is initially available in insufficient quantities, seedling establishment and growth may be much slower, so that tree recruitment occurs over periods of 30 to 50 years (Veblen and Lorenz 1986). "Dog hair" stands are extremely dense stands in which trees grow very slowly and do not vary much in size. Such exceptionally dense stands appear to reflect abundant availability of seed, favorable climatic conditions for initial seedling survival, and then lack of self-thinning of the stand. Towards its lower elevational range, lodgepole pine is successionally replaced by Douglas-fir, and at higher elevations by Engelmann spruce and subalpine fir (Marr 1961, Peet 1981, Veblen and Lorenz 1986). Typically, lodgepole pine is a seral species in the FR. However, over a restricted elevational and moisture range where the shade-tolerant conifers are absent, lodgepole pine can form self-perpetuating stands (Peet 1981, Parker and Parker 1994).

At higher elevations in the subalpine zone on moderately xeric sites, post-fire colonization is likely to be dominated by lodgepole pine, either alone or with varying numbers of other conifers and aspen. Establishment of lodgepole pine at these higher elevations is prolonged, often lasting well over 100 years (Whipple and Dix 1979, Veblen 1986a). After fire, Engelmann spruce may also establish in abundance coincidentally with lodgepole pine. Where seed sources are available and where the site is less xeric, spruce is more likely to play a significant role in the early phases of stand development. In approximately 200-year old stands, as the canopy cover closes, seedling establishment of lodgepole pine typically ceases and seedling establishment of spruce is drastically reduced. In contrast, fir often does not begin to establish abundantly for at least 50 years following stand initiation but continues to establish abundantly throughout the remainder of stand development (Whipple and Dix 1979, Veblen 1986a). This pattern, however, is highly variable and probably depends on seed availability and site conditions as well as climatic conditions during the early post-fire stages of stand development. As self-thinning occurs in the post-fire cohort of lodgepole pine, dominance steadily shifts towards spruce and fir, often until the pine are entirely eliminated (Peet 2000).

Potentially, infrequent surface fires may occur in the sparse understory of the relatively dry lodgepole pine forests (e.g., Kipfmueller and Baker 2000, Sibold 2001). Such fires would retard successional replacement of lodgepole pine by killing seedlings of spruce and fir. However, data from the Medicine Bow Range and from Rocky Mountain National Park indicate that most of the area dominated by lodgepole pine has not supported repeated or extensive surface fires (Kipfmueller and Baker 2000, Sibold 2001, Sibold et al. unpublished m.s.). We stress that current knowledge of fire history in lodgepole pine-dominated stands in the southern Rockies of Colorado and southern Wyoming does not support the idea that surface fires were an important part of the historic fire regime in this forest type.

Where blowdown occurs in lodgepole pine-dominated stands with understories of spruce and fir seedlings, it can dramatically accelerate stand development towards spruce and fir (Veblen et al. 1989). Due to their abundance as small understory individuals, shade-tolerant species (especially subalpine fir) survive the blowdown better and increase their growth rates in response to the reduced competition from the canopy trees. Spruce beetle outbreaks, that result in the death of large Engelmann spruce, and occasionally large lodgepole pine, in old post-fire stands have a similar effect of accelerating successional replacement of the shade-intolerant pine by the more shade-tolerant spruce and subalpine fir (Veblen et al. 1991c). Mountain pine beetle (*Dendcroctonus ponderosae*) outbreaks can also significantly influence lodgepole pine. Such outbreaks tend to shift dominance towards the non-host species (spruce and fir). In long-lasting post-fire stand development, the effects of such subsequent canopy disturbances by wind or insect outbreaks depend on the timing of the disturbance relative to the stage of stand composition and structure.

5.1.7. Limber Pine and Bristlecone Pine Cover Types

Limber pine is physiologically well adapted to grow under the driest conditions of the post-fire environment in the subalpine zone. At extreme sites of thin, rocky soils exposed to strong winds, limber pine typically dominates stand development following an intense fire (Peet 1981, Veblen 1986a, Donnegan and Rebertus 1999). Its seeds are dispersed to such sites mainly by birds (Tomback et al. 1990). At these harsh sites, limber pine often is the sole colonist for many decades, but eventually is replaced by Engelmann spruce and subalpine fir at most sites. Limber pine may facilitate establishment of spruce and fir by providing microsites protected from full sunlight and strong winds (Rebertus et al. 1991). At extreme sites of shallow soils and exposure to strong winds, spruce and fir are unable to establish even after a mature stand of limber pine has developed. At such sites, limber pine forms open woodlands of often very old trees (> 1000 years) in which there is sporadic regeneration at a low rate (Rebertus et al. 1991).

Bristlecone pine forests are structurally similar to limber pine forests, and the two species often co-occur in the AR (Baker 1992). Towards the mesic end of its distribution bristlecone pine may play a successional role similar to that of limber pine in post-fire development of spruce-fir forests. However, both bristlecone and limber pine also occur in habitats too dry for the development of dense stands of the more shade-tolerant conifers, such as along windy ridges at the perimeters of large parks (Donnegan 1999). Following crown fires at these harsh sites, there is relatively sparse (but still dominant) regeneration of bristlecone pine (Baker 1992).

5.1.8. Engelmann Spruce-Subalpine Fir Cover Type

In addition to the patterns described above for post-fire development from dominance by shade-intolerant aspen and lodgepole pine toward dominance by Engelmann spruce and subalpine fir, the latter two species also regenerate directly following fires and can co-dominate the site from the time of stand initiation, especially at sites that lack seed sources of pines or root suckers of aspen (Rebertus et al. 1992). Following fire, Engelmann spruce is likely to establish in greatest abundance, and there may be a lag in the establishment of fir. For example, age-structure studies have shown that fir establishment typically lags that of spruce by many decades (Whipple and Dix 1979, Veblen 1986a, Rebertus et al. 1991); however, where seeds are available, both species regenerate immediately following fire (Doyle et al. 1998). The reestablishment of spruce and fir following fire is significantly affected by the size of the burn, by the availability of seed, and by climate (Peet 1981; Alexander 1984; Tomback et al. 1990; Rebertus et al. 1992).

At some sites, bimodal age distributions of spruce indicate both an immediate post-fire cohort and a second cohort that establishes c. 150 to 250 years after the initial cohort begins to thin (Aplet et al. 1988). At other, apparently less favorable, sites where canopy closure is less complete, bimodal age distributions of Engelmann spruce are not found (Whipple and Dix 1979, Veblen 1986a, Veblen et al. 1991a, Rebertus et al. 1992). Such variation in age structures with site conditions is an illustration of the importance of site variation to stand development patterns in the subalpine zone (Peet 1981, Veblen 1986a). Consequently, it is not prudent to assume that stand development of post-fire spruce-fir stands fits a single model but rather to expect variation in the timing and relative abundances of establishment by the dominant tree species (Rebertus et al. 1992).

Given the long period of post-fire stand development before the initial post-fire colonists die (> 500 years), there is a high probability that the stand will be affected by a major canopy disturbance in the form of a windstorm or spruce beetle outbreak (Veblen et al. 1989,1991a, 1991c, Eisenhart 1999). These large-scale canopy disturbances typically reduce the relative dominance of the main canopy by spruce, and result in vigorous releases of subcanopy individuals of both species (Schmid and Hinds 1974, Veblen et al. 1989, 1991c; Kulakowski and Veblen 2003). In old stands, in gaps created by the windthrow of a single tree or small groups of canopy trees, both species regenerate (Veblen 1986b, Veblen et al. 1991a).

5.2. Disturbance Patterns

The following review of disturbance patterns and their ecological consequences is organized by type of disturbances (e.g., wind, fire, insects). For some types of disturbances, the discussion is organized according to two broad elevational zones: 1) montane forests including ponderosa pine, and Douglas-fir; and 2) subalpine forests including spruce-fir, lodgepole pine, limber pine, bristlecone pine, and aspen. For other types of disturbances, such as some insects, the discussion must focus on particular forest types. Wherever possible, the discussion will focus on studies and data from the FR. When no data or studies are available from the FR, the discussion will necessarily have to be based on information from nearby areas in the Rocky Mountain region.

5.2.1. Wind

Exceptionally strong wind storms occasionally cause extensive blowdowns in subalpine forests of the southern Rocky Mountains and are important determinants of stand development patterns (Alexander 1987, Veblen et al. 1989). For example, in 1987, a tornado blew down 6000 ha (14,820 ac) of forest in the Teton Wilderness (Fujita 1989, Knight 1994). In 1997, easterly winds of 200-250 km/hr blew down over 10,000 ha (24,700 ac) of forest on the western slope of the Park Range in Routt N.F. (Flaherty 2000; Baker et al. 2001). Blowdowns of moderate size (e.g. < 100 ha; 250 ac) have also been documented in Routt NF (Kulakowski and Veblen 2003). Small blowdowns of approximately a fifth to several hectares are also common in the subalpine forests of Colorado (Alexander 1964, Veblen et al. 1991a). The higher incidence of blowdowns in the subalpine zone compared to the montane zone in the Colorado Rockies reflects differences in tree species' susceptibility to wind damage and greater frequency of extreme wind speeds in the higher elevation zone. The principal conifers of the subalpine zone are shallow rooted and not windfirm, whereas in the montane zone both ponderosa pine and Douglas-fir have deep roots and are windfirm. In the subalpine zone high wind events are more common, rugged terrain may create greater turbulence, and there is higher probability of heavy snow loads which contribute to treefalls (Alexander 1987). In this zone, windthrow is greater where topographic or logging patterns constrict and therefore accelerate winds (Alexander 1964). Other features that increase the hazard of windthrow in relation to cutting operations include shallow or poorly drained soils, location on leeward cutting boundaries, dense stands, infestation by root and butt rots, and steeper slopes (Alexander 1964).

While susceptibility to damage from severe wind storms largely depends on storm meteorology and local topography, it also varies with forest cover type and stage of post-fire stand development. In comparison to the conifers of the subalpine zone, aspen is more resistant to wind damage (Veblen et al. 2001). Aspen is better able to withstand high-speed winds because of its more flexible bole and extensive root system that makes uprooting more difficult. Furthermore, if the wind event occurs when aspen is leafless there will be major differences in snow loads in comparison with the coniferous species. Analyses of damage patterns associated with the severe 1997 Routt blowdown (Baker et al. 2002, Kulakowski and Veblen 2002) show that: 1) stands more recently affected by stand-initiating fires were less affected than old-growth stands; and 2) aspen-dominated stands are less affected than spruce-fir stands. Thus, in 1997 large areas that had burned in the late 19th century and were dominated by relatively young lodgepole pine and/or spruce-fir experienced less blowdown damage than older coniferous forests. Stands with greater components of aspen in the canopy were also less affected by wind damage. In the case of wind events less severe than the 1997 Routt Divide blowdown there is evidence that even moderate differences in stand ages affect susceptibility to wind damage. For example, comparison of wind disturbances in spruce-fir stands > 400 years old and an adjacent c. 250-year old post-fire stand revealed the latter to be less susceptible to small blowdowns (< 0.3 ha) (Veblen et al. 1991a). Weather conditions preceding a storm event may also contribute to windthrow. For example, the high precipitation during the two months prior to the October 1997 Routt Divide blowdown may have increased waterlogging of soils making many trees less windfirm

Successional consequences of a blowdown in the subalpine zone are highly contingent on the time since last severe fire, which largely determines stage of seral development. Critical to the response to the blowdown is the presence or absence of juvenile spruce and fir in lodgepole pine-dominated post-fire stands. For example, in 1973 a blowdown of a 350-year old post-fire forest in Rocky Mountain National Park accelerated succession from lodgepole pine towards subalpine fir and spruce (Veblen et al. 1989). In the absence of lodgepole pine, however, in 1934 a blowdown of an old-growth spruce-fir forest in Routt NF shifted forest dominance from Engelmann spruce towards subalpine fir (Kulakowski and Veblen 2003). When blowdowns change species dominance from spruce to fir, stand susceptibility to beetle outbreak may be reduced until spruce regains dominance of that stand. However, blowdowns may also trigger outbreaks of spruce beetle (see Section 5.4). Where land use has altered forest structures through changes in fire regimes and logging, the potential response to natural wind storms may have been altered.

5.2. Snow and Flood Events

Snow may cause disturbance to vegetation in several ways. Deep snows that persist late into a growing season may limit plant regeneration (Holway and Ward 1963, Butler et al. 1992). Furthermore, long lasting snow cover can lead to intensive overgrazing by ungulates in areas that are snowfree (Gilbert et al. 1970). Late snowmelt may also limit the success of coniferous trees by favoring the blackfelt snowmold (Herpotrichia nigra) (Cooke 1955; Knight 1994). Heavy snowstorms occasionally strip limbs and break the canopies of large conifers, or cause the collapse of dense patches of young conifers or aspens.

In the Colorado Rockies, snow avalanches are locally important on steep slopes at high elevations in subalpine forests (Ives et al. 1976, Carrara 1979). Avalanche paths can lower treelines below climatic limits (Walsh et al. 1994). Sites that are most frequently disturbed by avalanches tend to be dominated by small flexible shrubs, while at somewhat lower elevations in runout zones, the time between successive avalanches is often enough to permit establishment of trees (Knight 1994). Relatively small Engelmann spruce, subalpine fir, and quaking aspen characterize avalanche paths of moderately frequent events (Carrara 1979, Veblen et al. 1994). The greatest damage is done by infrequent large avalanches that extend beyond the usual terminus zone of avalanches. After such events, aspen is likely to recover dominance of the site more quickly than spruce or fir due to its flexible stems and vegetative reproduction.

Flooding causes local watershed changes that can significantly alter riparian ecosystems (Baker 1989, Malanson and Butler 1990). Riparian areas often contain highly diverse ecosystems, and geomorphic processes along stream channels both create and remove habitats. Coarse woody debris is an especially important component for creating fish and aquatic habitats in stream channels (Harmon et al. 1986, Malanson and Butler 1990). Plant species diversity has been related to the area of sediment bars, stream sediment loads, and coarse woody debris washed into stream channels from upper reaches, much of which depends on watershed-scale vegetation patterns (Malanson and Butler 1990). However, vascular plant species richness at 115 riparian sites over a 300 km (190 mi) length of the western slope in Colorado did not spatially correlate with indicators of fluvial disturbance (Baker 1990). Watercourses and riparian area have been significantly affected by human modification of streams and rivers in the FR area (Wohl 2001). In addition to altering the natural flood regime, the construction of dams, diversions, and other structures that affect water flow has reduced the water yield of most major water courses in this area (USDA Forest Service 2002a).

The presence and abundance of beavers (*Castor canadensis*) also has an influence on the riparian ecology of the FR area. The increased availability of subsurface water associated with beaver dam construction may increase the growth of riparian vegetation (Wohl 2001). Furthermore, because beaver dams increase water storage along rivers, they contribute to more uniform stream flows. Conversely, removal of beavers may cause early failure of their dams, resulting in increased sediment transport and pronounced channel widening. Beaver were widespread in North America prior to Euro-American settlement. Intensive trapping of beaver in the 19th century led to a near extinction of this species in Colorado (USDA Forest Service 2002a). However, during the 20th century, due to reintroduction and protection, beaver populations have rebounded.

Flooding has long been an important disturbance process in certain habitats of central northern Colorado. Wohl (1992) documented multiple floods on the Poudre River extending back to 6000 years before present based on slackwater flood-laid deposits. These floods likely were the result of both local damming on the river in addition to upstream heavy rainfall events. Paleoflood events extending back to 7546 BC are also documented on Bear Creek that flows into the South Platte River near Denver (Grimm et al. 1995). The study in Bear Creek found a consistent decrease in flood discharge with increasing elevation, reflecting greater importance of heavy rainfalls in lower elevations as a driving force of flooding in the montane zone (see Pruess et al. 1998 for a similar study in the San Juan Mountains).

In 1869, the Hayden Survey traveled the length of the Front Range to New Mexico. The Hayden Survey was one of four major surveys of the West that were conducted by the US Geographical and Geological Surveys between 1867 and 1879 (Bartlett 1962). This survey reported highly variable flows on the Cache la Poudre River, Cherry Creek in Denver, and the Arkansas River. It also reported that six to seven years prior to their visit in 1869, severe flooding in the Front Range had washed away cabins on the floodplains of several streams. Thus, regulation of stream flow by dams is a departure from the naturally high variability of stream flow in the FR.

Changes in vegetation structure and disturbance regimes have been implicated in increased stream damage after modern fires, such as from the 1996 Buffalo Creek fire on the South Platte (Illg and Illg 1997). It is likely that previous intense fires, such as the 1851 fire dated for montane ponderosa pine forests at Cheesman Lake (Brown et al. 1999), resulted in flood events similar to the 1996 Buffalo Creek event. Concern with watershed protection has led to forest restoration plans designed to protect overstory vegetation from catastrophic fires (Foster Wheeler Environmental Corporation 1999). Thus, although crown fires sometimes followed by severe erosion events are part of the natural range of variability of the FR, vegetation managers may choose to mitigate the severity of such events through vegetation treatments in order to protect valuable water resources.

5.2.3. Fire

5.2.3.1. Varying Fire Regimes Along the Montane to Subalpine Gradient

The determinants of a fire regime, or the spatial and temporal occurrence of fire within a specified area, include fuel type and condition, ignition sources, topography, and weather at the time of ignition (Baker 2003). Fuels are fundamentally controlled by productivity and decomposition rates permitted by the regional climate, but humans also can greatly modify fuel types through logging, fire suppression, road construction, or any other activity that alters fuel, stand, or landscape structure. In addition, potential ignition sources include both humans and lightning. Primarily due to climatic gradients associated with elevation, it is possible to generalize about fire regimes in the subalpine *versus* the montane zone and for the major forest cover types (Table 5.1). However, there is much local variation associated with topography and vegetation structure that results in some variability of fire regimes even within the same cover type, most notably within the ponderosa pine cover type.

More frequent occurrence of fire in the montane zone compared to the subalpine zone is supported by analysis of fire ignitions in different cover types in the AR, PSI and Rocky Mountain National Park based on observational records during the 20th century (Fig. 5.1; Table 5.2). Cover types of the ignition points were determined for over 1000 fires in the AR (1970-1995), over 1900 fires in the PSI (1961-1998), and for 254 fires in RMNP (1915-1995). Expected and observed frequencies of fire were compared for each cover type on the basis of the extent of each type. All data sets show similar patterns of greater fire frequency in the ponderosa pine and Douglas-fir cover types and at lower (< 2750 m) elevation (Fig. 5.2). The greater fire occurrence in the montane zone is not due simply to higher numbers of human-set fires because lightning-ignited fires are also more frequent in the montane zone. Although greater frequency of human-set fires is expected for lower elevation sites that are easier to access by humans, the greater incidence of lightning-ignited fires at low elevation is not consistent with the greater frequency of lightning strikes at higher elevations (Reap 1986). This implies that fuel conditions, probably fuel desiccation, are more conducive to fire in the lower elevation forest types. In most years, not just years of unusual drought, summer fuel conditions are sufficiently dry to allow the ignition and spread of at least small fires in the montane zone.

In the FR, fires occur less frequently than expected in the lodgepole pine and spruce-fir cover types and at higher elevations (Table 5.2). This reflects the dependence of fire ignition and spread on unusual drought at high elevation. This pattern also holds for the other subalpine cover types (aspen, limber pine, and bristlecone pine) in the PSI; these cover types are much less extensive in the AR and RMNP. Fires also occur less frequently than expected in the pinyon-juniper cover type (Table 5.2). Given the warm, dry conditions typical of this habitat, it is likely that lack of fuel continuity explains the lower than expected fire occurrence in the pinyon-juniper cover type. Although there are no significant differences in fire frequency on north- versus south-facing slopes for the AR and RMNP, in the PSI fire frequencies differ according to slope aspect. Xeric, south-facing slopes have higher fire frequencies whereas mesic, south-facing slopes have lower than expected fire frequencies.

Overall, the modern pattern of less frequent than expected fires in the subalpine zone and more frequent than expected fires in the montane zone is consistent with the long-term tree-ring record of fire history in these zones. It is difficult to quantitatively compare fire frequencies between the subalpine and montane zones because of differences in abundance of fire-recording tree species and especially because of the higher probability of the destruction of fire-scarred trees by subsequent stand-replacing burns in the subalpine zone. However, there are consistent *qualitative* differences in the fire frequency and effects of fire in the subalpine and montane zones. The reasons for these differences include differences in the influence of climatic variability on fuel desiccation at high versus low elevations as well as differences in fuel quantities and configurations (Veblen 2000, Baker 2003, Schoennagel et al. 2004).

Historic Fire Regimes of the Montane Zone

Fires in the lower montane zone (i.e. below c. 2100 m; 6888 ft) of formerly open ponderosa pine woodlands are interpreted to have been predominantly surface fires carried mainly by herbaceous fuels (Peet 1981, Veblen et al. 2000, Sherriff 2004). Fires in the lower montane zone were more frequent than at higher elevations, because weather conditions that desiccate herbaceous fuels sufficiently for fire spread are more common at lower elevations years (Veblen et al. 2000). Because of the relatively high frequency of these fires (i.e. return intervals sometimes shorter than 10 yrs at scales of c. 100 ha (Fig. 5.3a) and the survival of many fire-scarred trees, these fires must have been relatively low-intensity, surface fires. These interpretations are based largely on fire-scar data from study sites below c. 2100 m (6888 ft) in the northern Front Range (Veblen et al. 2000, Sherriff 2004, Sherriff and Veblen unpublished m.s.). These are the lowest elevation sites for which fire-scar data are available for the FR.

Although surface fires may have been the modal fire type in the lower montane zone, in most of the montane zone above c. 2200 m (7216 ft) pre-1900 stand-replacing or partially stand-replacing fires in areas dominated by ponderosa pine are well documented by historical photographs and even-aged stand structures in conjunction with fire-scarred remnant trees in the northern Front Range (Veblen and Lorenz 1986, 1991, Hadley and Veblen 1993, Mast et al. 1998, Ehle and Baker 2003). In stands of c. 100 ha fires often did not recur in less than 50 years and sometimes 100 years; trees with more than two or three scars, even on trees initially scarred in the 1600s, are rare (Fig. 5.3b; Veblen et al. 2000, Sherriff 2004). Historical photographs show widespread stand-replacing fires, especially on north-facing slopes and at higher elevations within the montane zone of the northern Front Range; historical photographs also show scattered areas of open woodlands and grasslands in the montane zone where there was a greater likelihood of surface fire (Appendix 2; Veblen and Lorenz 1991). Likewise, in the montane zone of Pike N.F. tree age structures in combination with fire-scarred remnant trees indicate that stand-replacing or partially stand-replacing fires also occurred in the ponderosa pine and Douglas-fir forests (Brown et al. 1999, Kaufmann et al. 2000). But, in the same areas short fire recurrence intervals (sometimes less than 10 years) to the same small stand or even to the same tree indicate that non-lethal surface fires also affected some areas of the montane zone (Rowdabaugh 1978, Skinner and Laven 1983, Goldblum and Veblen 1992, Brown et al. 1999, Kaufmann et al. 2000, Donnegan 1999, Veblen et al. 2000). Thus, the pre-20th century fire regime of the montane zone can be described as “mixed and variable” (Brown 1995, Kaufmann et al. 2000) to reflect occurrence of both surface and stand-replacing fires. We use the term mixed-severity fire regime to describe this fire regime type, which was characteristic of most of the montane zone of the FR.

Studies of fire history in the Front Range indicate that the historic fire regime of the montane zone was substantially different from conditions found in ponderosa pine forests in the Southwest. For example, in the Southwest numerous fire history studies have shown mean fire return intervals of less than 10 years for fires that scarred at least 10% of the fire-scar recording trees in sample areas of approximately 10 to 100 ha (Swetnam and Baisan 1996), whereas in the Front Range at comparable spatial scales historic fire intervals were generally much longer (Laven et al. 1980, Goldblum and Veblen 1992, Brown et al. 1999, Brown and Shepperd 2001, Veblen et al. 2000, Kaufmann et al. 2000, Ehle and Baker 2003). Although some individual sites of approximately 4 to 100 ha in the Front Range have recorded moderately high fire frequencies of mean fire intervals of 12 to 20 years (Brown et al. 1999, Veblen et al. 2000, Brown and Shepperd 2001), the more common pattern is one of fire recurrence (based on > 10% trees scarred) in the range of 40 to over 100 years. Thus, there is a consensus among fire history researchers that widespread, surface fires in Front Range ponderosa pine stands recurred much less frequently than in southwestern ponderosa pine forests (Brown and Shepperd 2001, Kaufmann et al. 2000, Veblen et al. 2000, Veblen 2003b, Ehle and Baker 2003, Sherriff 2004).

As a broad generalization, the relative importance of stand-replacing fires increases with elevation from the lower montane zone of open ponderosa pine stands to the upper montane zone of denser ponderosa pine and Douglas-fir, or the “mixed conifer” zone. Recent research in the AR indicates that even within the ponderosa pine cover type a much smaller proportion of the landscape than previously believed fits the notion that surface fires were the primary fire type, maintaining formerly open woodlands (Sherriff 2004, Sherriff and Veblen unpublished m.s.). This study related spatial variation in historic fire regimes to variation in abiotic and biotic predictors of fire regimes in the ponderosa pine zone on the eastern slope of the northernFR in the AR. Fifty-four fire history sites were examined for different fire frequency types in relation to habitat variables in a c. 61,000 ha study area encompassing the full range of ponderosa pine from 1800 to 3000 m elevation in the AR. Logistic regression was used to relate three fire frequency classes to environmental predictors and to derive probability surfaces for mapping purposes. The results indicate only about 20% of the ponderosa pine zone historically had a relatively high frequency of fire return intervals in the range of 10 to 30 years. More than 80% of the ponderosa pine study area is reconstructed to have had moderate (mean fire interval > 30 years) to low (MFI > 40 years) fire frequency. Fire severity was interpreted to be mainly high (i.e. stand-replacing or partially stand-replacing) in the moderate and low fire frequency classes. The high fire frequency class, interpreted to be mainly low severity fires, is clearly delimited by low elevations; it occurs primarily below 2100 m (6880 ft). The moderate and low fire frequency types occur across a broad range of elevation and are related to variations in aspect, slope and distance to ravines. Only about 20% of the ponderosa pine zone of the northern Front Range fits the widespread notion that the historic fire regime was characterized mainly by frequent surface fires that maintained open, savanna-like stands (Sherriff 2004, Sherriff and Veblen unpublished m.s.).

In the southern Front Range, the studies of fire history and past forest conditions conducted at the Cheesman Lake have also concluded that the ponderosa pine cover type had an historic fire regime of both stand-replacing and non-stand-replacing fires (Brown et al. 1999, Kaufmann et al. 2000, Kaufmann et al. 2001, Huckaby et al. 2001). These studies are based on intensive study of a single 35 km2 area of ponderosa pine-dominated forest covering part (2100 to 2400 m) of the elevation range of the montane zone. Based on elevation alone, the Cheesman Lake study area would not include the habitat types characterized in the northern Front Range as having an historic fire regime of frequent, low-severity fires (Sherriff and Veblen unpublished m.s.). Thus, the "mixed and variable" fire regime found in the Cheesman Lake study area should be a comparable to the areas classified as having "moderate" or "low" fire frequency within the ponderosa pine zone of the northern Front Range (Sherriff 2004, Sherriff and Veblen unpublished m.s.). The Cheesman Lake and northern Front Range studies include overlapping elevations and forests of similar dominance (i.e. mainly by ponderosa pine, but with co-dominance by Douglas-fir at some sites). In the northern Front Range these habitats were classified as having fire return intervals of > 30 years (moderate) or > 40 years (low) at spatial scales of c. 100 ha, and fires were interpreted to have been mainly stand-replacing; evidence of widespread surface fires was relatively limited, and such fires were not interpreted to have had a significant impact on forest structures (Sherriff 2004, Sherriff and Veblen unpublished m.s.). In the Cheesman Lake study area, fires that burned $5km^2$ or larger areas occurred in 1534, 1587, 1631, 1696, 1723, 1775, 1820, 1851, and 1880 (Brown et al. 1999). Tree ages and fire-scar locations are interpreted as indicating that these widespread fires were stand-replacing in places but burned through the forest floor without causing significant tree mortality in other places (Kaufmann et al. 2000). Thus, the historic fire regime of the Cheesman Lake study area of ponderosa pine forests is believed to have had both a lethal and non-lethal component that left many surviving canopy trees (Kaufmann et al. 2001, Huckaby et al. 2001). The results from the Cheesman Lake area coincide with those from the northern Front Range (Sherriff 2004, Sherriff and Veblen unpublished m.s., Ehle and Baker 2003) in that the historic fire regime of the ponderosa pine cover type included an important component of lethal, stand-replacing or partially stand-replacing fires. However, the extent and ecological effects of low-severity fires apparently is interpreted by Kaufmann et al. (2000, 2001) to have been greater than in the interpretations for the northern Front Range by (Sherriff 2004, Sherriff and Veblen unpublished m.s., Ehle and Baker 2003). In the Cheesman Lake study area some severe widespread fires, such as the 1851 event, are believed to have created long-lasting openings that were not colonized by trees for well over 100 years Kaufmann et al. (2000, 2001). Such events probably also occurred in the northern Front Range but were not directly addressed in the relevant studies (Sherriff 2004, Sherriff and Veblen unpublished m.s., Ehle and Baker 2003).

In the upper montane zone, Douglas-fir/mixed conifer and lodgepole pine stands are typically even-aged, and the regeneration of these forests was interpreted by the earliest scientific observers to be largely controlled by stand-replacing fires (Clements 1910, USDA 1920). Age data from the northern Front Range indicate that most of these stands originated during the episode of extensive burning and logging dating from the mid-1800s to early 1900s (Moir 1969, Veblen and Lorenz 1986). These earlier studies are consistent with the trend towards greater dominance of the historic fire regime by stand-replacing fires as elevation is increased throughout the montane zone (Veblen et al. 2000, Sherriff 2004, Sherriff and Veblen unpublished m.s.).

Historic Fire Regime of the Subalpine Zone.

In the subalpine forests of the FR and nearby areas, continuous canopy fuels of the relatively dense coniferous forests permit widespread stand-replacing crown fires (Clagg 1975, Romme and Knight 1981, Veblen 2000, Veblen 2000, Kipfmueller and Baker 2000, Sibold 2001, Baker 2003). Site conditions permit the development of dense stands with continuous canopy fuels and often laddered fuels that are favorable to the occurrence of crown fires. Unusually dry weather conditions needed for extensive fire spread in the otherwise relatively mesic subalpine zone typically occur at long intervals, leading to great fuel buildup between fires.

In spruce-fir and lodgepole pine forests of the subalpine zone in the southern Rockies, including forests of the FR, numerous studies of forest structure and fire history document that these forests have been shaped primarily by infrequent severe (stand-replacing) fires as opposed to low-intensity surface fires. In areas of continuous forest in the subalpine zone of northern Colorado and southern Wyoming vast areas have burned in single stand-replacing events as indicated by extensive even-aged tree populations (Whipple and Dix 1979, Romme and Knight 1981, Veblen 1986a, Aplet et al. 1988, Parker and Parker 1994; Kipfmueller and Baker 2000, Sibold 2001, Kulakowski and Veblen 2002; Kulakowski et al. 2003, Howe and Baker 2003, Buechling and Baker 2004, Sibold et al. unpublished m.s.). These forest age structure studies are consistent with historical photographs taken in the late 19th and early 20th centuries in northern Colorado showing hundreds to thousands of hectares of dense, burned subalpine forest with few if any surviving trees (Sudworth 1900, Veblen and Lorenz 1991). These photographs were taken well before any fire suppression was enacted by land management agencies, and they clearly support the interpretation from forest age structure studies that severe, stand-replacing fires were characteristic of these forests prior to any putative effects of fire suppression.

Although stand-replacing fires are the predominant fire type in the subalpine zone, historically there has been a low incidence of surface fires in these forests. The important questions about surface fires in the historic fire regime are: (1) How extensive and frequent were surface fires? (2) What were the effects of surface fires on tree establishment and mortality? A large fire event is likely to have both a stand-replacing and a surface-fire component. In some cases the surface fire component may be limited to areas where crown fuels are discontinuous (e.g. in meadows). However, modern observations and fire scars indicate that some fires burn as surface fires in small areas of subalpine forest. Presence of fire scars dating to a single year on multiple trees within the same stand indicate that in limited areas surface fires have burned in lodgepole pine as well as in spruce-fir stands (Sibold 2001, Kulakowski et al. 2003). However, estimated reconstructions of areas burned by different fire types indicate that only small percentages of subalpine forests are burned by surface fires in comparison to stand-replacing fires (Kipfmueller and Baker 2000, Buechling and Baker 2004, Sibold 2001, Sibold et al. unpublished m.s.). For example, in a c. 30,000 hectare area of subalpine forest in RMNP less than 3% of the study area had been affected by surface fires (Sibold et al. unpublished m.s.). Fire-scar data indicates that surface fires occurred less frequently in spruce-fir forest than in lodgepole pine stands (Sibold 2001). However, the important observation, even for lodgepole pine stands, is that trees rarely record evidence of more than a single surface event. Multiple fire scars of different dates on the same tree are exceedingly rare in the interiors of subalpine forests. This strongly implies that surface fires occurred either as small components of fire events that were stand-replacing over larger areas or that they occurred very infrequently in the absence of a stand-replacing event. Where the effects of these surface fires have been examined in lodgepole pine as well as in spruce-fir forests, they have not been found to have a significant influence on tree mortality or tree regeneration patterns (Kulakowski et al. 2003, Sibold et al. in progress).

Evidence that large infrequent fires dominated the historic fire regime of the subalpine zone in Colorado also comes from the paleoecological record of charcoal deposition. In northern RMNP, fire history in a subalpine watershed has been analyzed by radiocarbon dating of thick charcoal bands in a sediment core covering the last 6250 years; the results imply that most or all of this 24 km^2 watershed burns on average once every 480 years (Madole 1997). Fossil pollen evidence also suggests that stand-replacing fires with long (centennial) return intervals have been important in the subalpine forests of western Colorado for much of the Holocene (Fall 1997).

As noted in the chapter on Methodology, using a single number such as a rotation period or mean fire return interval to characterize historic fire regimes that are known to have varied both temporally and spatially in large areas of subalpine forest is fraught with methodological limitations. Thus, results from different studies of fire history should not be regarded as precise estimates of fire rotations but rather should be interpreted to mean that fire rotations in subalpine zone are on the order centuries. Fire rotation periods (time period to disturb the entire area once by fire) have been estimated for stand-replacing fires to range from c. 145 to over 500 years for subalpine forests studied in southeastern Wyoming and northern Colorado (Veblen et al. 1994, Kipfmueller and Baker 2000, Sibold 2001, Buechling and Baker 2004, Sibold et al. unpublished m.s.). However, the interpretation of rotation period needs to consider that some areas within a large study area may have burned at shorter or much longer intervals than the mean fire return interval. For example, in the most extensive study of fire history yet conducted in subalpine forests in Colorado, in 5 study areas of equal size in a c. 30,0000 hectare forested area in Rocky Mountain National Park rotation periods averaged about 280 years; yet, over 25% of the forested area did not burn in the past 400 years (Sibold et al. unpublished m.s.). This and all other fire history studies conducted in subalpine forests in the southern Rockies document the following key features of historic fire regimes: 1) the predominant fire type was stand-replacing rather than surface fires; 2) large areas (often more than 50% of study areas > 4000 ha in size) burned in single events; and 3) intervals between significant fires within the same study area of c. 4000 ha were often greater than 100 years.

Fire regimes are spatially variable within the subalpine zone and due to methodological limitations summary information on fire history only rarely can clearly indicate the differences in fire history among forest types within the same watershed. Higher elevation zones of spruce-fir experience fire less frequently than lodgepole pine that occurs at lower elevation and on topographically more xeric sites. For example in Rocky Mountain National Park virtually all of the lodgepole pine zone burned in the last 400 years whereas over 30% of the spruce-fir forest had not burned over the same time period in each of 5 watersheds (Sibold et al. unpublished m.s.).

Limber pine stands in the subalpine zone typically occur on drier and more wind-exposed sites than the other subalpine forest types, and are characterized by a greater frequency of fires than in the adjacent mesic forests of spruce-fir (Sherriff et al. 2001, Sibold 2001). Since these fires leave many surviving trees and tree age data do not indicate the presence of post-fire cohorts, most of these fires were low-severity events (Sibold 2001). These fires may have burned in the crown in small groups of tree and on the surface where fine fuels were sufficiently abundant to carry a fire. The key characteristic of these fires is that they were small, many burning over surfaces of a few hectares or less. Discontinuous fuels in this habitat would have prevented continuous fires over large areas in this habitat, typically near treeline. However, other limber pine stands on more favorable sites are clearly seral to spruce-fir stands and are characterized by long interval stand-replacing fires typical of the spruce-fir cover type.

Although the data on fire history in subalpine forests in Colorado are derived primarily from the tree rings of conifers, the fire regime of aspen forests in the Southern Rockies is also generally described as being one of relatively infrequent and stand-replacing fires (e.g. Romme et al. 2001). However, aspen also occurs as either sparse stands with understories dominated by herbaceous vegetation or as small clusters of trees surrounded by meadow. In meadows with sparse populations of aspen, fires are more likely to burn as surface fires, and potentially such sites supported more frequent surface fires than the surrounding coniferous forest. However, this is only conjecture because there are no fire history studies conducted specifically in aspen-meadow habitats in the FR. On the hand, fire return intervals potentially may be lower for habitats with dense stands of aspen in comparison with adjacent stands of spruce-fir or lodgepole pine. This conjecture is based on the observation that fires often stop when they travel from a conifer stand into an aspen stand. For example, in 2002 in the Routt NF, fires that severely burned conifer stands, usually stopped at adjacent aspen stands and did not travel a significant distance as a surface fire in aspen.

5.2.3.2. Influences of Climatic Variability on Fire Regimes

The relationship of fire occurrence and behavior to fire weather at temporal scales of hours to seasons is well understood and used to predict short-term fire hazard (Rothermel 1983).

At an inter-annual scale, synchronous occurrence of fire-scar dates from areas too large for fire to have spread from a single ignition point is strong evidence that regional climate is influencing fire regimes. For the area from southern Wyoming to southern Colorado, widespread burning in 1880 was recorded in early documentary sources (Sudworth 1900, Plummer 1912, Ingwall 1923), as well as in tree-ring studies of fire history (i.e., Skinner and Laven 1983, Zimmerman and Laven 1984, Goldblum and Veblen 1992, Kipfmueller 1997, Veblen et al. 2000, Brown et al. 1999; Kulakowski and Veblen 2002; Kulakowski et al. 2004, Buechling and Baker 2004, Sibold et al. unpublished m.s.). Other individual years that recorded fire scars at disjunct locations over this large area include 1654, 1684, 1842, 1851,1859-1860, 1871-1872, 1879-1880, and 1893-94 (Kipfmueller 1997, Alington 1998, Brown et al. 1999, Veblen et al. 2000, Donnegan 1999, Donnegan et al. 2001; Sherriff 2000, Sherriff et al. 2001, Sibold 2001; Kulakowski and Veblen 2002). Such synchrony of fire years suggests that at a regional scale extreme weather increases fire hazard over extensive areas from southern Wyoming to southern Colorado.

The occurrence of many widespread fires in the second half of the 19th century that is widely documented for northern Colorado (Figs. 5.5) coincides with increased frequency and severity of drought in the tree-ring proxy records of climate for the Rocky Mountain region (Gray et al. 2003, Woodhouse et al. 2002). Indeed, tree rings sampled at numerous sites in northern Colorado (Cook et al. 1998, Veblen et al. 2000, Buechling and Baker 2003) indicate that all of the major fire years listed above correspond with significant drought during the year of the fire and/or the year immediately preceding the fire year. Over the period from 1700 to 1900, reconstruction of the Palmer Drought Severity Index indicates that the three driest years were 1842, 1851, and 1880 (Cook et al. 1998) and these were also years of widespread burning in the Front Range.

Variations in ENSO have been shown to coincide with variations in fire history across a range of forest types in Colorado (Veblen et al. 2000, Sherriff et al. 2001, Veblen and Kitzberger 2002, Grissino-Mayer et al. 2004). In the montane zone of the northern Colorado Front Range, based on samples of stands dominated by ponderosa pine but sometimes including other conifers, comparison of tree-ring records of fire and climatic variation from 1600 to the beginning of fire suppression in 1920 indicate that fire is strongly associated with inter-annual climatic variation (Veblen et al. 2000). Warmer and drier spring-summers, indicated in instrumental climatic records (1873 to 1995) and in tree-ring proxy records of climate (1600 to 1983), are strongly associated with years of widespread fire. Years of widespread fire also tend to be preceded by 2 to 4 years of wetter than average spring conditions. Thus, years of widespread fire tend to occur during dry years closely following years of above-average moisture that increase the production of fine fuels (Veblen et al. 2000). Alternation of wet and dry periods lasting one to a few years is conducive to fire spread, and is strongly linked to El Niño-Southern Oscillation (ENSO) events. The warm (El Niño) phase of ENSO is associated with greater moisture availability during spring in the FR that results in a peak of fire occurrence several years following El Niño events. Conversely, dry springs associated with La Niña events were followed by more widespread fire during the same year (Veblen et al. 2000). There is a highly similar pattern of ENSO influences on fire occurrence in Pike N.F. (Donnegan 1999). A similar pattern of ENSO and fire for Arizona and New Mexico (Swetnam and Betancourt 1992, 1998) indicates a regionally extensive association of fire and ENSO activity in the southern Rocky Mountains.

The favorable effects of above-average moisture two to several years prior the fire event is found primarily in lower elevation ponderosa pine forests (Veblen et al. 2000, Sherriff 2004). At increasing elevations within the montane zone and in the subalpine zone, fire occurrence is associated primarily with drought, which in turn is often associated with La Niña events (Sherriff et al. 2001, Sherriff 2004, Sibold and Veblen unpublished m.s.).

The period from c. 1780 to 1840 was a time of reduced ENSO activity in comparison with the second half of the 19th century (Michaelsen and Thompson 1992, Swetnam and Betancourt 1998). This is manifested as reduced year-to-year variability in tree-ring widths of ponderosa pine in the period from c. 1780 to1840 in the Southwest (Swetnam and Betancourt 1998) and in the Colorado Front Range (Donnegan 1999, Veblen and Kitzberger 2002). The period from c. 1780 to 1840 is also a time of reduced fire occurrence in the Front Range (Veblen et al. 2000, Donnegan 1999, Donnegan et al. 2001, Brown et al. 2000). Fewer or less extreme ENSO-related cycles of wet, fuel-producing El Niño events closely followed by dry La Niña events may explain this period of reduced fire occurrence in ponderosa pine forests. Fewer or less extreme La Niña events would explain the low fire occurrence in subalpine forests from c. 1780 to 1840. Based on tree-ring evidence from sites widely dispersed in the Front Range, after 1830 there is a gradual increase in the variability of tree-ring widths in the late 1800s (Donnegan 1999, Donnegan et al. 2001, Veblen and Kitzberger 2002). Increased variability in tree-ring widths may indicate greater ENSO variability at that time, and in conjunction with increased ignitions by humans (see below) probably accounts for the increase in fire occurrence during the latter half of the 19th century. As noted below, decadal scale changes in the north Pacific and Atlantic ocean temperatures may also have contributed to the relatively low fire occurrence from c. 1780 to 1840 followed by higher fire occurrence in the second half of the 19th century.

In subalpine forests in northern Colorado, drought and fire years are significantly associated with La Niña events (Sherriff et al. 2001, Sibold and Veblen unpublished m.s.). When La Niña events coincide with the negative (cool) phase of the Pacific Decadal Oscillation their influence on drought and fire occurrence in subalpine forests of the northern Front Range is enhanced (Sibold and Veblen unpublished m.s). Conditions in the Atlantic Ocean also influence the climate of the Rockies (Gray et al. 2004). For example, the positive phase (warm) of the Atlantic Multidecadal Oscillation coincides with increased fire occurrence in subalpine forests in the northern Front Range (Sibold and Veblen, unpublished m.s.). Overall, fire occurrence in the Front Range shows strong teleconnections to ocean-atmosphere patterns in the tropical Pacific, northern Pacific and Atlantic oceans. This is a key point because these ocean-atmosphere interactions are known to have fluctuated significantly at decadal to centennial time scales during the reference period of this assessment. Such climatic variation imparts a high degree of natural variation to the fire regimes and forest conditions of the FR.

5.2.4. Insects and Diseases

In the Front Range the important forest insect pests include mountain pine beetle (*Dendroctonus ponderosae* Hopkins), Douglas-fir beetle (*D. pseudotsugae* Hopkins), spruce beetle (*D. rufipennis* Kirby), western tent caterpillar (*Malacosoma californicum* Packard), pandora moth (*Coloradia pandora* Blake), Douglas-fir tussock moth (*Orgyia pseudotsugata* McDunnough), and western spruce budworm (*Choristoneura occidentalis* Freeman) (Schmid and Mata 1996). Important pathogens affecting many of the conifers include dwarf mistletoe (*Arceuthobium* spp.) and *Armillaria* root diseases. Each of these disturbance factors and how humans potentially could have modified their occurrence will be considered separately below. Other decay, root, and foliage diseases and pathogens affect Colorado forests but are not included, either because they are minor in the FR or because no potential linkage to human activities could be argued.

All of these pathogen and insect species are natives of western North America and have co-evolved with their hosts for millennia. It is also important to note that even for contemporary ecosystems, quantitative data on populations (e.g., numbers of insects per tree or area) are rare. An insect population may be qualitatively classified as at endemic or epidemic levels, or in terms of the amount of tree mortality they cause over an area (Schmid and Mata 1996). It is often difficult or arbitrary to define the temporal limits of insect outbreaks that are typically spatially heterogeneous within a particular landscape or region.

In addition to native pathogens, a significant threat exists from the white pine blister rust

(*Cronartium ribicola*) that was introduced in 1910 to the northern Rockies from Europe. White pine blister rust has reached limber pine populations in the Redfeather Lakes area of the northern Front Range in the last few years (Johnson 1999) and is a major threat to limber and bristlecone pine populations in the state (Johnson 1997). Its spread down the Front Range is believed to be only a matter of time (D. Johnson, personal communication). Whitebark pine and limber pine populations in the northern Rockies have been infected severely, causing widespread mortality and loss of an important food source (pine seeds) for many wildlife species. The presence of this and other exotic pathogens obviously is well outside the HRV for these ecosystems.

5.2.4.1. Bark Beetles

All of the bark beetles (*Dendroctonus* spp.) tend to attack larger trees (typically > 10 to 20 cm diameter; > 4 to 8 inches diameter), and their attacks are normally lethal (Schmid and Mata 1996). They bore through the bark, create egg galleries, mate, and deposit eggs in the phloem layer. They carry with them fungi, which in conjunction with the beetle's excavations, results in blockage of water- and nutrient-conducting tissues, thus killing the tree.

Mountain pine beetle

In the southern Rocky Mountains, mountain pine beetle (MPB) primarily attacks ponderosa pine and lodgepole pine. MPB has been the most destructive of the bark beetles in the western US; during epidemics nearly 100% of overstory trees have been killed over many square kilometers (Schmid and Mata 1996). In the montane zone, outbreaks may convert mixed-aged stands to young stands of ponderosa pine or accelerate succession towards Douglas-fir (Amman 1977). In the subalpine zone, elimination of overstory lodgepole pine by a beetle outbreak can accelerate succession towards more shade-tolerant Engelmann spruce and subalpine fir. Mountain pine beetle outbreaks may recur in the same general region within about 20 years and to the same stand in about 50 to 100 years, depending on how much of the original stand was killed by beetles (Schmid and Amman 1992). Durations of outbreaks are quite variable (2 to 14 years) and may decline rapidly if weather becomes unfavorable to beetle populations (Schmid and Mata 1996). Outbreaks also are believed to increase the likelihood of fire occurrence over a period of about 2 years while dead needles persist on trees (Schmid and Amman 1992), but longer-term influences on forest flammability may be quite complex. Fall of dead needles may temporarily decrease fuel continuity in the canopy, but subsequent in-growth of understory trees in combination with fall of dead trees may eventually increase fire hazard (Knight 1987). Fire-injured trees are generally more susceptible to attack by mountain pine beetle (Amman and Ryan 1991).

It is widely believed that increased stand densities associated with fire exclusion in the 20th century have increased the susceptibility of ponderosa pine stands to outbreaks of mountain pine beetle (e.g., Roe and Amman 1970, Schmid and Mata 1996). However, there are no long-term data (e.g., based on tree-ring records) of the frequency or duration of outbreaks to examine this hypothesis. Furthermore, the proportion of the ponderosa pine cover type in which stand densities have increased significantly during the fire suppression era is on the order of 20% in the AR (Sherriff 2004, Sherriff and Veblen unpublished m.s.). Occurrence of extensive MPB outbreaks in the late 1800s and early 1900s (Roe and Amman 1970) clearly indicates that not all outbreaks can be attributed to stand structural changes resulting from modern fire exclusion.

Management activities designed to suppress MPB epidemics in the early stages of development include cutting, bark-peeling, and/or spraying small groups of infested trees. These activities may be combined with the use of pheromones to attract beetles. Prevention activities are aimed at changing forest conditions over large areas to make forest stands less attractive to beetles. Nevertheless, numerous studies have shown that stand thinning of initially dense stands decreases rates of attack by mountain pine beetle (Larsson et al. 1983, Olsen et al. 1996, Feeney et al. 1998). Increased resin flow and overall insect resistance in ponderosa pine are improved when initially dense, young stands are thinned (Feeney et al. 1998). However, in mature low-density stands, the resistance of widely spaced trees to attack by mountain pine beetle is not likely to be improved by thinning. Although silvicultural techniques are effective at reducing stand-level susceptibility to MPB, some incidence of natural MPB outbreaks characterized ponderosa pine forests during the reference period.

Numerous mountain pine beetle outbreaks occurred during the 20th century throughout the southern Rocky Mountains, including the AR (Roe and Amman 1970). For the AR an outbreak was reported for 1908-1910, which pre-dates any significant effects of fire suppression on forest conditions (USDA Forest Service 1920). Observations in the Lake Creek area of the Sangre de Cristo Range during the 1970s indicated that the beetles preferentially attacked trees infected by dwarf mistletoe (Frye and Landis 1975). Existence of probable relationships between beetle attack, dwarf mistletoe infestation, and fire exclusion provides a further mechanism by which human impacts may have altered the occurrence of MPB outbreaks during the 20th

century. However, given the presence of MPB outbreaks prior to the implementation of fire suppression policy, some incidence of MPB outbreaks must be considered as part of the historical range of variability.

Overall, MPB outbreaks have long been a part of the dynamics of the ponderosa pine and lodgepole pine forests in the FR, and certainly occurred in these forests prior to any effects of fire suppression or logging. There are no long-term (i.e. multi-century) reconstructions of the history of MPB outbreaks in Colorado, and consequently it is impossible to compare the extent of 20th century outbreaks with those of earlier centuries. It could be argued that if past management has resulted in larger areas of dense lodgepole pine stands in the FR, then susceptibility to MPB infestation may have increased. However, as previously discussed (Section 5.3) there is no evidence that surface fires formerly thinned lodgepole pine forests which would have been a key premise underlying the argument that fire suppression has increased tree density at a stand level. On the other hand, at a broader spatial scale it is possible that fire suppression has resulted in a somewhat more homogeneous age of stands over a broad area, but this is just conjecture. It is much more likely that current stand ages in the lodgepole pine zone of the FR are more homogeneous due to the widespread fires of the late 19th century, and, in fact a large percentage of the extant lodgepole pine stands originated in the late 19th century (Figs. 6.2 and 6.3). However, previous episodes of drought-related extreme burning are likely to have occurred many times in this landscape, and the large extent of relatively even-aged lodgepole pine stands is believed to be within the historic range of variability for this landscape. That implies that to the extent current MPB outbreaks may be related to the existence of widespread, dense stands of lodgepole pine, they should be considered to be within the historic range of variability.

Douglas-fir beetle

The Douglas-fir bark beetle (DFB) can cause widespread mortality of Douglas-fir in the southern Rocky Mountains. Defoliation by western spruce budworm may increase Douglas-fir susceptibility to beetle attack, and its epidemics appear to have arisen during and expanded following outbreaks of western spruce budworm (Hadley and Veblen 1993, Schmid and Mata 1996). Stands attacked in the Front Range tend to have higher tree densities and show poor tree growth during the five years prior to an outbreak (Negron 1998). DFB tends to kill larger, dominant trees more frequently than smaller trees. Many of the same potential interactions with fire previously mentioned for mountain pine beetles apply to DFB (Cates and Alexander 1982).

The earliest recorded outbreak of DFB for the AR is from 1911 when approximately 20 hectares of Douglas-fir were reported dead or dying in the Millers Fork area northwest of Drake (USDA Forest Service 1920). DFB outbreaks have been observed to last 5 to over 10 years, and intervals between outbreaks in the same areas may be on the order of 15 to 35 years during the 20th century (Hadley and Veblen 1993, Schmid and Mata 1996). Major epidemics in the northern Front Range occurred in 1934-1938, 1950-1951, and 1984-1990 (Hadley and Veblen 1993). Schmid and Mata (1996) suggest that if DFB attacks always occur after spruce budworm epidemics, their pre-20th century recurrence interval on the Front Range may have been on the order of 25 years as that is the recurrence interval for western spruce budworm derived from tree-ring studies (see below). However, there is no direct evidence of DFB recurrence intervals prior to the early 20th century, and it is not possible to determine quantitatively if the frequency of DFB outbreaks during the latter half of the 20th century is outside the historical range of variability. Nevertheless, reports of logging of large diameter Douglas-fir in the Front Range in the late 19th century (Jack 1900) indicate that the pre-20th century incidence of DFB outbreaks was not too great to prevent the survival of abundant individuals of the host species.

Spruce beetle

The spruce beetle (SB) in the southern Rocky Mountains mainly infests Engelmann spruce, but under outbreak conditions lodgepole pine is also attacked (Alexander 1987, Schmid and Mata 1996). Endemic SB populations infest fallen trees and scattered live trees but during outbreaks can kill most canopy spruce over extensive areas. Spruce < 10 cm (4 inches) in diameter usually are not attacked, nor is subalpine fir, and their accelerated growth following the death of surrounding canopy trees can be used to date outbreaks (Veblen et al. 1991b). Stands containing large (i.e., > 55 cm diameter; > 21 inches diameter) spruce and especially those in valley bottom sites are most susceptible to outbreaks.

SB outbreaks result in a massive shift in dominance in basal area from spruce to fir. This shift is due both to mortality of large spruce and in-growth of formerly suppressed seedlings and saplings of subalpine fir that are typically the most abundant tree species in the understory (Veblen et al. 1991c). Some new seedling establishment of spruce and fir may occur but not of the seral lodgepole pine, probably due to inhibition of seedling establishment by an already established understory.

Blowdowns or accumulation of logging debris are usually the immediate triggers of outbreaks (Schmid and Frye 1977), which is an important distinction from outbreaks of mountain pine or Douglas-fir beetle. Fallen trees provide abundant food and winter protection for promoting the growth of SB populations which then attack living trees. A strong windstorm in 1939 in northwestern Colorado is believed responsible for the largest recorded epidemic of the 20th century in the Rocky Mountain region (Massey and Wygant 1954). This 1940s SB outbreak killed 4.3 billion board feet of timber in White River, Grand Mesa, and Routt National Forests (Massey and Wygant 1954). Tree-ring methods and historical photographs document the occurrence of an earlier SB outbreak in the mid-1800s in northwestern Colorado that likely was at least as extensive as the 1940s outbreak (Baker and Veblen 1990, Veblen et al. 1991b, 1994, Eisenhart and Veblen 2000). This widespread SB outbreak in the mid-1800s, as well as outbreaks recorded in fossil records (Feiler and Anderson 1993), occurred prior to any significant impact of Euro-Americans on the subalpine forests of northwestern Colorado in the form of either logging or fire suppression. Severe and widespread SB outbreaks are clearly a natural component of disturbance regimes in the subalpine zone.

The frequency of severe outbreaks in the same stand is limited by lack of trees large enough to be susceptible to beetle attack (Schmid and Frye 1977). At a stand scale, lack of large-diameter spruce for 70 to 100 years after a severe outbreak or a stand-replacing fire prevents that stand from being attacked even when surrounding older forest is infested (Veblen et al. 1994, Schmid and Mata 1996). However, at a regional scale, northwestern Colorado was affected by two major outbreaks in a span of only c.100 years (Veblen et al. 1991b). At a smaller scale in a 594 ha (1467 acres) area of subalpine forest in northwestern Colorado, tree-ring methods documented three extensive SB outbreaks since the early 1700s (Veblen et al. 1994). Mean return interval and rotation period for SB outbreaks were 117 and 259 years, respectively, which made disturbance by SB more important, at least spatio-temporally, than disturbance by fire in this valley.

Although tree-ring studies of pre-20th century SB outbreaks have not been conducted in the FR, early observers documented a widespread outbreak in Pike N.F. In 1905, on the southern slopes of Pike's Peak at about 3000 m, the entomologist A.D. Hopkins attributed the "vast destruction of spruce" to an epidemic of SB "some fifty years ago" (Hopkins 1909:127). Given the long persistence of beetle-killed spruce, the epidemic could have occurred as early as the 1830s or 1840s. Hopkins implied that beetle-caused tree mortality predisposed forested areas to burning. He wrote: "...there has been a most intimate interrelation of destructive bark beetles and forest fires in the denudation of the vast areas of once heavily forested lands in the Rocky Mountain region, and in very many cases the insects have first killed the timber, and the fire has then followed..."   (Hopkins 1909:127).

A causal relationship between insect outbreaks and subsequent fire hazard is difficult to establish (Knight 1987, Schmid and Mata 1996).   Increased hazard of fire initiation could result from exposure of snags and dry logs to lightning strikes. Greater hazard of fire spread due to the abundance of fine fuels on standing dead trees may be short-lived as needles fall and decay during a relatively short period of 2 to 5 years. In contrast, greater fuel loads from the dead and down coarse material as well as standing dead trees may increase the potential fire intensity for many decades. Although it is widely believed that the mortality caused by an SB outbreak would increase subsequent risk of fire ignition and spread, this was not necessarily the case following the 1940s SB outbreak in northwestern Colorado. While the slow decay and fall rate of the dead-standing trees (Hinds et al. 1965) could imply that there may be an increased risk of fire initiation and spread over many decades, empirical data from White River National Forest does not support this notion (Bebi et al. 2003; Kulakowski et al. 2003). Following the severe 1940s beetle outbreak, there was no increase in fire frequency in beetle affected stands during the 50 years following the outbreak, nor were there extensive and severe fires (Bebi et al. 2003) until the extreme drought of 2002. This suggests that ignition and spread of fire in subalpine spruce-fir forests is much more influenced by extremely dry weather rather than by abundance of coarse, dead fuels, such as those following beetle outbreak. However, on-going research on the severity of the 2002 fires in the Flat Tops area of western Colorado indicates greater severity fires in old stands of beetle-killed trees in comparison with adjacent young stands that originated after late 19th century fires (Christof Bigler, Dominik Kulakowksi, and Thomas Veblen unpublished data). Due to the importance of fine fuels to fire initiation and spread, it is likely that SB outbreaks increase those components of fire risk during a relatively short period of 2 to 5 years when fine fuels from dead needles and twigs are more abundant. If extreme fire weather does not occur during that relatively short period when dead fine fuels are still on the trees, the effect of the SB outbreak on fire risk is weakened. However, when fires do occur in association with extreme drought in the subalpine zone, they burn more severely in areas of old forest with abundant beetle-killed trees. Thus, the severity component of fire hazard is increased by previous SB outbreaks.

Because logging operations clearly can trigger SB outbreaks (Schmid and Frye 1977), there is the potential that management can increase the incidence of SB outbreaks over its historical range of variability, certainly at stand and landscape scales. However, at a regional scale there is abundant evidence that extensive SB outbreaks occurred in Colorado prior to the

20th century. If a regional scale SB outbreak were to affect the FR during the 21st century, that would be considered within the range of historical variability for this type of widespread but infrequent natural event.

5.2.4.2. Defoliating Insects

Western Tent Caterpillar

The western tent caterpillar (WTC) is a native caterpillar that defoliates primarily aspen above 2400 m, but WTC is relatively rare in the FR (Schmid and Mata 1996). Endemic populations cause partial defoliation that rarely results in the death of the tree. However, during epidemics complete defoliation for multiple years results in substantial growth reduction and mortality in aspen stands. If defoliation causes the death of many overstory trees, the stand may be converted to a stand of young aspen stems or succession towards dominance by shade-tolerant conifers may be accelerated.

Near La Veta Pass in the southern Front Range, stands suffered heavy mortality after 6 years of nearly complete defoliation by the WTC in 1980 (Schmid and Mata 1996). For the aspen stands of northern New Mexico and southern Colorado, the frequency of epidemics is 10 to 20 years, but single epidemics can last more than 10 years. Presently, there is no evidence that changes in stand structure or fire exclusion over the 20th century have significantly influenced the occurrence of tent caterpillar epidemics in the FR (Schmid and Mata 1996).

Pandora Moth

Pandora moth (PM) is a defoliating moth that attacks ponderosa pine and lodgepole pine in Colorado (Schmid and Mata 1996). PM generally causes growth reductions and increased stress that can lead to mortality from other insect or disease agents. Historically, outbreaks of PM have not been common or extensive in Rocky Mountain forests (Schmid and Mata 1996).

A tree-ring study in Oregon reconstructed historical PM outbreaks over the past c. 600 years (Speer et al. 2001). Outbreaks typically lasted 6 to 8 years and outbreak timing exhibited quasi-periodic fluctuations centered around 20 and 40 year intervals. In the Oregon study, the 20th-century outbreaks were not more synchronous (extensive), severe, or longer in duration than outbreaks in previous centuries, but there was an unusual reduction in activity in 1920-1980 (Speer et al. 2001). The authors speculate that the 20th-century reduction in PM may have resulted from logging and removal of large ponderosa pine in mid-century, but they note that the evidence of a causal relationship between logging and reduced outbreaks of PM is only suggestive, not conclusive. Schmid and Mata (1996), however, conclude that stand density and structure do not influence the start of PM outbreaks and, therefore, changes in forest structure in recent decades probably has not contributed to increased moth activity in Colorado. For Colorado, there is no information available on pre-20th century outbreaks of PM.

Douglas-fir Tussock Moth

The Douglas-fir tussock moth (DFTM) is a defoliator that attacks primarily Douglas-fir and true firs, but can attack other conifers. During the 20th century, it has not been a significant pest in the southern Rocky Mountains (Lessard and Raimo 1986). However, two of the most significant known outbreaks of DFTM occurred in the southern Front Range. The first was in 1937 on Cheyenne Mountain near Colorado Springs and affected about 60 to 80 ha (148 to 198 acres; Lessard and Raimo 1986). The next major outbreak started in the lower South Platte valley in 1983 and continued through the rest of the decade, eventually causing almost total mortality of Douglas-fir in an area of approximately 324 ha (800 acres) above Deckers. It is not known whether such outbreaks are within the HRV of the species or may be related to effects of humans on forest structure.

Western Spruce Budworm

Western spruce budworm (WSB) is a defoliating moth with larvae that feed on needles and cones. The WSB primarily defoliates Douglas-fir and white fir in the southern Rocky Mountains (Schmid and Mata 1996). Extensive defoliation by WSB over several years can produce high levels of tree mortality, but large percentages of attacked trees usually recover after a defoliation event. Competitively suppressed trees and trees stressed because of poor site conditions suffer higher rates of mortality (Cates and Alexander 1982). Young, vigorous post-fire stands may show minimal defoliation by WSB whereas multi-tiered stands with high stem densities and a range of tree sizes are more severely affected (Hadley and Veblen 1993). WSB outbreaks in mixed stands of Douglas-fir and ponderosa pine tend to shift dominance towards pine (Hadley and Veblen 1993). During outbreaks, WSB larvae also consume cones and seeds which further impedes the ability of stands to recover from attacks (Schmid and Mata 1996). Aspect and relief appear to influence the spatial and temporal patterns of WSB outbreaks through their influences on rate of post-fire stand development and therefore on the abundance and size of the host species in mixed Douglas-fir and ponderosa pine stands (Hadley

1994). Dry, south-facing aspects result in slower rates of post-fire stand development that reduce susceptibility to WSB by reducing the density of the host species. More complex relief contributes to less uniform stand structures, and therefore lower susceptibility to WSB (Hadley 1994).

In the northern Front Range, montane forest structures were significantly altered by outbreaks of WSB during the early 1940s, late 1950s, and late 1970s-1980s, and by outbreaks of Douglas-fir bark beetle during the mid-1930s, early 1950s, and mid-1980s (Hadley and Veblen 1993). Tree-ring studies indicate that, since the early 18th century, epidemics in the southern Rocky Mountains have occurred at a frequency of about 20 to 33 years in the same stands (Swetnam and Lynch 1989, Shimek 1996, Veblen et al. unpublished data). In one stand in the Rampart Range of Pike N.F., tree-ring data document that WSB has coexisted with Douglas-fir in the stand for over 500 years (Veblen et al. unpublished data).

Given the apparently greater susceptibility of stands with suppressed understories of Douglas-fir saplings, it is logical to consider whether land-use practices have created stand structures that are more susceptible to WSB outbreaks. Studies from Montana to New Mexico show that during the 20th century, WSB outbreaks have been severe and synchronous over large areas (McCune 1983, Anderson et al. 1987, Swetnam and Lynch 1989, Hadley and Veblen 1993, Shimek 1996). Based on tree-ring patterns in 10 sites extending from Carson N.F. in New Mexico to Roosevelt N.F. in Colorado, Swetnam and Lynch (1989) identified a relatively long period of reduced WSB activity in the first few decades of the twentieth century, followed by an increased synchrony of outbreaks since the 1940s. Widespread timber harvesting in the montane zone during the late 1800s and early 1900s opened up stands that may not have become susceptible to outbreaks again until the 1940s when they became relatively closed stands with multiple-level canopies (Swetnam and Lynch 1989). Increased 19th century burning in the upper montane zone in parts of Colorado also would have created extensive areas of post-fire even-aged stands that more or less synchronously become susceptible to WSB outbreaks after some 50 to 80 years of stand development (Hadley and Veblen 1993). During the initial decades of stand development, these post-fire stands are not highly susceptible to outbreaks, but as stands continue to age they take on a multi-tiered structure with subcanopy populations of suppressed Douglas-fir that increase stand susceptibility.

Fifteen tree-ring records of budworm activity for the area extending from the Sangre de Cristo Range northwards along the Front Range now permit a better assessment of long-term changes in budworm outbreaks (Fig. 5.6). These records do *not* show a clear pattern of altered frequency or increased synchrony of outbreaks for 20th century in comparison with earlier centuries. Although there is a high degree of regional synchrony of outbreaks during the second half of the 20th century, early outbreaks (e.g. late 1700s) also were highly synchronous over large areas. If trends in the extent and severity of WSB outbreaks since the 1940s continue to intensify over the early decades of the 21st century, then this disturbance agent may depart from its historical range of variability. However, the current record of 20th century outbreaks of WSB is not outside the historic range of variability as indicated by tree-ring records from the 1600s to the present.

5.2.4.3. Pathogens

Dwarf Mistletoes

Dwarf mistletoes (*Arceuthobium* spp.) are hemi-parasites that infect most pine species and Douglas-fir in Colorado. In Colorado, it has been estimated that 50% of lodgepole pine trees are infected by mistletoe (*A. americanum*) (Johnson 1997). In the Southwest, southwestern pine dwarf mistletoe (*A. vaginatum*) causes more damage to ponderosa pine than any other pathogen (Hawksworth 1961). Mistletoe infections weaken trees and make them more susceptible to attack by other pathogens, such as mountain pine beetle (Frye and Landis 1975). Although timber resources have been impacted severely in many areas, mistletoes are important for structuring forest stands and for creating habitat and resources for bird communities (Bennetts et al. 1996). Thus, mistletoe infections are an important consideration where the primary management goal is timber production, but deformed trees due to mistletoe infections are clearly part of the historic range of variability.

Macrofossils of dwarf mistletoes have been found in packrat middens as far back as 21,500 (± 500) years ago in the Southwest (Van Devender and Hawksworth 1986), evidence that these parasites and their hosts have co-evolved over long periods in this region. For the southern Front Range, Jack (1900:45) noted that mistletoe “was found to check and distort the growth of a great many trees in some localities, and in some cases it eventually caused their death.” Specifically for Roosevelt N.F., mistletoe was reported to be attacking all the conifer timber species at least as early as 1911 (USDA Forest Service 1920).

Severe crown fires in lodgepole pine forests may sanitize a stand, whereas surface fires leave infected trees and allow continued germination of mistletoe seeds (Zimmerman and Laven 1987). Stands in which greater numbers of trees survive the most recent stand-replacing fire have higher rates of mistletoe infection than in post-fire stands (Kipfmueller and Baker 1998b).

Generally, percent of trees infected increases with the age of post-fire lodgepole pine stands (Zimmerman and Laven 1984). However, in the Medicine Bow Range some old stands are not infected, suggesting that factors in addition to time-since-fire are important to mistletoe infection (Kipfmueller and Baker 1998b). Spatial spread patterns, not just stand age, are an important determinant of infection levels (Kipfmueller and Baker 1998b).

At landscape and regional scales, trends in mistletoe infection are uncertain. In the montane zone characterized by a mixed severity fire regime and heterogeneity of stand ages, it is likely that infection rates have probably been continuously high over the past several centuries. In the lower subalpine zone of lodgepole pine forests, it is more likely that fires of extreme extent may have resulted in past periods of reduced infection. Increased occurrence of stand-replacing fires during the late 19th century may have temporarily reduced infection rates in the young post-fire stands. Spread of mistletoe can be slow, and long-distance dispersal is rare. However, if large numbers of infected trees survived the stand-replacing fires, they would have permitted more rapid re-infection. These, however, are just conjectures on possible changes in the incidence of mistletoe infections in the FR.

To the extent that fire suppression is resulting in decreased abundance of young post-fire lodgepole pine stands and greater abundance of stands > c. 100 years old, this increased landscape homogeneity may be leading to increased infection over larger areas. However, historical data on pre-20th century stand structures would be needed to test this hypothesis. Furthermore, it is likely that any decrease in post-fire stands < 100 years old would have been at least partially offset by an increase in young lodgepole pine stands following logging. Past silvicultural practices that led to the incomplete removal of infected trees also may have promoted more rapid re-infection of stands. Brooms on infected trees could have created ladder fuels that led to local crown fires in heavily infected stands (Dahms and Geils 1997), but brooms also occurred as part of the natural range of variability. In the montane zone, if fire suppression has resulted in increased crown continuity in some ponderosa pine stands, this could also lead to increased infection rates simply because of greater tree density (Dahms and Geils 1997). However, in the montane zone high stand densities are more likely to have resulted from the extensive fires of the late 19th and early 20th centuries and from logging (see Section 6). Fire suppression contributes to this pattern of younger trees in particular habitats (e.g. low elevation ponderosa pine) but the major explanation for young dense stands is regeneration following burning or logging.

Armillaria and Other Root Diseases

Root diseases caused by various species of decay fungi are common throughout conifer forests of the western US (Wood 1983). Decay fungi injure trees by killing roots or causing heart rots. Trees die because of increased susceptibility to bark beetles or windthrow. Fungi can persist for decades in the roots of stumps or snags and spread through root contact between healthy and infected trees (Dahms and Geils 1997). Mortality is usually of canopy trees, and disease centers may persist for hundreds of years (Dahms and Geils 1997). In Southwestern coniferous forests, it has been found that root diseases and their associated pathogens are responsible for mortality of 34% of trees in stands surveyed (Wood 1983). In southern Colorado (including San Isabel N.F.) James and Goheen (1980) found that 80% of the trees infected with root disease also had bark beetles, suggesting that the root pathogens initially invade trees and predispose them to attack by insects. Dahms and Geils (1997) suggest there may be a positive feedback between fungi and beetles, with greater beetle-caused mortality resulting in greater sources for fungi inoculum.

There does not appear to be any historical information on root diseases, although like other pathogens in conifer forests of the AR, these species are natives and have co-evolved with their hosts. For the Southwest, Dahms and Geils (1997) speculate that root disease fungi have responded to increased host material, especially the presence of stumps in many logged stands. Increased tree density also increases chances of root contact with infected trees in disease centers, causing further mortality and infection in stands. However, for the FR as a whole there is no evidence that current levels of root diseases are outside the historic range of variability.

Aspen Diseases

Aspen stands are often severely affected by a variety of diseases that cause tree mortality and damage the commercial value of the wood (Hinds 1985). These include fungi and viruses that cause leaf diseases, many decay fungi that attack stems and roots, and canker-causing fungi that attack the bark. Such diseases are native to the southern Rockies, but at least at a stand scale human activities may increase rates of infection. For example, canker infection associated with wounding of trees can increase dramatically in managed stands (Walters et al. 1982). At a stand scale, many aspen diseases appear to have been increased by wounding associated with logging, but at a regional scale conversion of older stands to younger stands may have reduced the occurrence of some diseases (Hinds 1985). Wounding by elk, deer, and other wildlife also facilitates the spread of many aspen diseases.

In a survey of trees wounded by campers in Colorado, Hinds (1976) found an incidence of canker infection double that of trees unaffected by campers. Blazing of trees by early Euro-Americans as well as Native Americans probably also affected the health of individual trees. Thus, there are numerous mechanisms through which humans can alter the spread and severity of aspen diseases. However, there is no evidence that current levels of aspen diseases are outside the historical range of variability.

5.2.4.4. Climatic Variability and Insect Outbreaks

Weather profoundly affects the life cycles of insects as well as the capability of trees to respond to insect attacks, yet the effects of climatic variation on the occurrence of insect outbreaks may be quite complex (Swetnam and Lynch 1993, Logan et al. 1995). Mortality of bark beetles is increased by cold winters, and low temperatures are likely to be the major restriction on bark beetle outbreaks at high elevations (Massey and Wygant 1954; Frye et al. 1974). Generally, warmer temperatures promote bark beetle outbreaks both through their favorable influence on the life cycle of the insect and drought-related declines in the tree’s ability to withstand attack (Frye et al. 1974, Amman 1977). However, non-climatic factors related to stand structure also play such important roles that the association of outbreaks with particular types of weather is difficult to verify quantitatively.

The influences of temperature on the life-stages of the mountain pine beetles have been extensively examined through a combination of temperature experiments in the laboratory with field measurements of phloem temperatures in lodgepole pine and phenological modeling (Bentz et al. 1991, Logan and Bentz 1999, Bentz et al. 2001). These studies show that there is significant geographic variation in heritable differences in life-history parameters such as development time that are related to temperature. There are inherent temperature thresholds in each life-stage that help to synchronize population dynamics with seasonal climatic thresholds. These results explain why MPB beetle populations have historically either irrupted or remained at low levels in certain thermal environments. They also imply that with future temperature increases habitats which previously had been thermally hostile to the MPB will become thermally favorable habitats (Logan and Bentz 1999).

Similar to the temperature influences on MPB, both laboratory and field measurements have shown that higher temperatures accelerate the life cycle of SB (Hansen et al. 2001). Based on an analysis of SB outbreaks in southeastern Utah and western Colorado, Hebertson (2004) identified mean December temperature as the most important variable for classifying outbreak and non-outbreak years. Warmer Decembers favor outbreaks by enhancing the winter survival of SB. Warm late summer and early fall temperatures also are predictive of outbreaks, presumably because they allow more rapid completion of the life cycle of SB. Drought during the previous five years was also found to be predictive of SB outbreaks (Hebertson 2004), which is consistent with research showing that bark beetles are attracted to drought-stressed trees (Paine et al. 1997).

Although it has long been believed that drought pre-disposes Douglas-fir stands to outbreaks of WSB (Cates and Alexander 1982), recent research from Colorado and New Mexico suggests that it is instead wet periods that favor outbreaks in this region. In northern New Mexico, 24 tree-ring records of outbreaks from 1690 to 1989 indicate a tendency for outbreaks to coincide with years of increased spring precipitation (Swetnam and Lynch 1993). Tree-ring records from the Colorado Front Range also show that WSB outbreaks are associated with moister years (Veblen et al. unpublished data). Although non-climatic changes in stand structures may also play a role in predisposing stands to outbreaks (Swetnam and Lynch 1993), these data indicate a significant role is also played by climatic variation. Due to the significant role of climatic variation in creating conditions conducive to insect outbreaks, much caution should be exercised in relating outbreaks to stand structural changes caused by humans.

Overall, there is abundant evidence that climatic variability is a major determinant of outbreaks of insect pests in Rocky Mountain subalpine forests. The emphasis placed on the importance of climatic influences on these outbreaks does not contradict the importance of non-climatic pre-disposing factors acting at the level of individual stands. These include attributes such as high tree densities, high basal area, slow tree growth rates, and reduced vigor of individual trees. However, due to the significant role of climatic variation in creating conditions conducive to insect outbreaks, and the lack of evidence that fire suppression has dramatically altered forest structures at least at higher elevations, the primary causes for regional-scale forest insect outbreaks in the FR appear to be natural-climatic variability.

6. IMPACTS OF MODERN LAND USES ON THE LANDSCAPE

Local and regional distributions of tree species and forest types depend on complex interactions between abiotic factors (e.g., climate, soils, topography) and biotic processes (e.g., competition) which are often significantly altered by disturbances. Disturbances occur from both natural and human sources. In this section we review how land uses may have altered disturbance patterns and vegetation conditions.

6.1. Livestock Raising

Heavy grazing by livestock in many parts of the world has been associated with reduction or extirpation of some native plant species, proliferation of exotic plant species, and major changes in vegetation structure (Jarvis 1979, Huenneke 1988). Livestock change the vegetation through selective feeding habits and the differential ability of plants to withstand grazing, browsing, and trampling. The altered vegetation composition in turn alters litter characteristics, sometimes resulting in changes in decomposition rates, soil biotic activity and nutrient cycling. Effects of livestock on forest and woodland ecosystems are highly variable according to site-specific conditions of plant species palatability, soil conditions, type of livestock, the livestock grazing regime, and even climatic conditions which affect availability of alternative forage species.

Studies conducted elsewhere in the western U.S., but in ecosystems similar to those of the FR, suggest that livestock populations probably had significant impacts in the FR in terms of species composition of some plant community types, the dynamics of ecotones between arboreal and non-arboreal communities, and fire regimes. For example, in montane forests in Zion National Park, Utah, livestock appear to have contributed to an increase in tree density by improving conditions for tree seedling establishment by reducing competition from grasses and forbs (Madany and West 1983). In the savannas of the Southwest, increased herbivory by livestock also resulted in changes in the competitive relationships between grasses and woody plants, and has been implicated as a major cause of increased tree encroachment at the margins of grasslands (Archer 1994). In contrast, in many subalpine habitats in the western U.S., tree invasions of meadows have been linked to decreased pressure from livestock following periods of overgrazing (Dunwiddie 1977). Fuel reduction by grazing in the Southwest is believed to have contributed to reduced fire frequency after the late 1800s, which would have further promoted tree seedling survival in formerly more frequently burned grasslands or open woodlands (Cooper 1960, Archer 1994, Touchan et al. 1996). In many ecosystems in the West (including subalpine ecosystems), heavy grazing during the 19th century is believed to have facilitated a major shift from native plants to exotic invaders (Baker 1978, Mack 1989, Dull 1999).

6.1.1 Livestock Raising: Arapaho-Roosevelt N.F.

Following the mineral discoveries of the 1850s, cattle ranching became a major economic activity throughout the AR and the PSI (Jack 1900, Agee and Cuenin 1924, Wyckoff 1999). Large native herbivores, such as deer, elk, antelope, mountain sheep, and bison occurred in large numbers in the AR in the 19th century (Wyckoff 1999). This raises the question of whether livestock were simply occupying niches vacated by large native herbivores whose populations declined or disappeared quickly in the late 19th century. For example, mule deer and elk can damage or kill small trees either through excessive browsing or through rubbing their antlers against the bark. However, there are important differences in the patterns of disturbance created by native herbivores and livestock (Lauenroth and Milchunas 1989). For example, bison traveled in large herds that likely moved on when resources were depleted. Lauenroth and Milchunas (1989) suggest that bison grazing was likely of heavy intensity but low frequency for a given area, while later cattle grazing is high frequency but lower intensity. Bison were very abundant in the Plains adjacent to the AR and also occurred in the Front Range and in Middle and North Parks (Ingwall 1923, Porter and Porter 1950 and Black 1969). However, it is likely that bison had less of an impact on grassland conditions than did the subsequent livestock grazing of the late 19th century. Analogously, under a presettlement bison-grazing regime, it is probable that grass fuels would have had time to recover between periods of herbivory, whereas heavy grazing from livestock probably significantly reduced grass fuels. Although it is important to recognize that presettlement fluctuations in the populations of native elk, deer, and bison must have had impacts on vegetation conditions of the AR, the nature of these impacts can only be conjectured.

The fact that livestock impacts began early and were spatially extensive in the AR is important in evaluating the results of studies of animal impacts conducted during the 20th century. Local extirpations of and depletion of seed sources for more palatable species may have occurred prior to scientific description of the vegetation. Thus, exclosure studies initiated in the 20th century long after these changes occurred may be of limited value in assessing the impacts of livestock on presettlement vegetation conditions.

Overgrazing by livestock in the late 1800s to early 1900s has been suggested as an explanation for increases in stand densities in ponderosa pine woodlands in the northern Front Range (Marr 1961). Overgrazing may have promoted tree encroachment into grasslands, both by reducing competition from herbaceous plants and by reducing herbaceous fuels for fire. In open ponderosa pine stands, a positive feedback exists between herbaceous fuels and fire.

Fires promote graminoids and forbs by killing woody plants before they can establish and exclude herbaceous plants through shading or other competitive mechanisms. Heavy grazing can reduce competition from grasses and expose bare mineral soil for tree seedling establishment. Studies of grazed and ungrazed ponderosa pine forests subject to fire suppression in Utah and central Washington have documented higher densities of small trees in the grazed stands (Rummel 1951, Madany and West 1983).

6.1.2. Livestock Raising: Pike-San Isabel N.F.

Early reports of patterns of livestock use and their impacts on the vegetation suggest that to at least some degree livestock had altered vegetation conditions in the PSI by the late 1800s (Jack 1900, Agee and Cuenin1924). Although quantitative data are not available on livestock numbers for specific sites in the PSI during the 19th century, there is ample documentation that livestock were abundant and widespread by the late 1800s. Seasonal use of the San Luis Valley for grazing sheep and cattle herded northwards from present-day New Mexico began as early as the 1830s and 1840s (Wyckoff 1999). Following the mineral discoveries of the 1850s, cattle ranching became a major economic activity throughout the PSI. By the 1880s, the number of ranchers may have peaked but stock numbers were believed to have continued growing substantially until at least 1920 (Agee and Cuenin 1924). Agee and Cuenin (1924) concluded that forage conditions were better in the 1850s and 1860s than in the 1920s in the Salida District of PSI. For example, they wrote (1924:21):

> ...the notes made by General Beale's party when they went through the country in June, 1853, show that the creek bottoms and moist slopes in the vicinity of Cochetopa pass were covered with dense stands of clover. It seems reasonably certain that this part of the country has undergone a change in type from overgrazing. There is no clover now along the creek bottoms on either side of the pass. It also appears reasonably certain that many of the other ranges on the eastern slope have undergone changes in type.

Agee and Cuenin's conclusions were informed both by interviews with early settlers and field observations during the early 1900s and are consistent with Jack's observations. In describing range conditions in the Pike's Peak Reserve in the late1890s, Jack (1900:71) wrote:

> The pasturage is undoubtedly greatly inferior to that which formerly existed, and in past years much of the ground has been made to support a larger number of

> cattle than was warranted by the conditions. Excessive pasturage near the streams has greatly reduced or destroyed the grasses and other herbage and shrubs which should hold the soil and modify the flow of surface waters.

In describing several cattle ranches in the Plum Creek Reserve (including the Manitou Park area and the Rampart Range), Jack (1900:80-81) noted:

> In summer the cattle on these ranches are usually allowed to roam at large over any part of the reserve and are brought into sheltered places at the approach of severe winter weather. There may be 50 or 60 persons having ranches upon the reserves, who, in the aggregate, probably have between 1500 and 2000 cattle and horses. This, however, does not represent the total number of cattle grazed on the reserve in summer, because a great many are annually sent into the reserve from ranches outside of the boundaries, sometimes at a considerable distance from them. ... As the forested lands are rarely densely covered, some grasses, furnishing scattered and limited grazing, are found almost everywhere; but it is naturally along the creeks that the best and only important pasturage is found...That the grazing is often excessive and too localized is apparent to anyone following many of the streams, particularly the tributaries of Trout Creek and West Creek, in the southwestern part. The consequence is that the pasturage has deteriorated greatly... Moreover, the excessive number of cattle in some localities is more or less damaging to young forest growth, as even young conifers like Douglas spruce [*Pseudotsuga menziesii*] are occasionally browsed upon, and many seedlings are destroyed by trampling.

For the South Platte Reserve (the northern central and western part of Pike N.F., west of the South Platte River), Jack (1900) also noted that stock owners grazed their cattle, and in this Reserve also significant numbers of sheep, on government land. He estimated that the maximum number of cattle kept on the Reserve in the 1890s was probably about 5000, but because cattle were sent seasonally to the Reserve from distant ranches it was difficult to estimate the numbers of livestock. Jack (1900:101) noted that even in Lost Park (Kenosha Mountains), an area showing “less molestation by human agencies than any other in the reserves,” from several hundred to two or three thousand cattle were seasonally present. When he visited this area in September 1898, there were not more than 400 to 500 cattle. Jack (1900:102) reported that sheep grazing was injuring the sources of small streams and especially small trees near treeline:

> The vegetation of the high mountain slopes becomes badly trampled and cut up by hoofs, as well as reduced by excessive grazing; and in the hollows or ravines, where the streams originate or take definite form, the protective covering of low shrubs, which are chiefly willows, become very much injured or totally destroyed by trampling and browsing, leaving the ground bare and exposed, and liable to be washed away by any heavy rain.

In assessing the overall range condition of the Reserve, Jack wrote (1900:102):

> ...it seems to be the unanimous opinion of the earlier settlers that there has been a very decided reduction of the grazing value of the land as compared with its condition when first used for this purpose. The chief reason is obvious to these ranchmen, who admit that there has been overpasturage, too many cattle on the same ground year after year trampling it, especially near water, so as to expose the roots of the grasses, keeping the latter as closely cropped as though devoured by grasshoppers, and preventing any possibility of production of seed for regeneration.

Jack's description of the early impacts of livestock in the Pike N.F. region is corroborated by Ingwall's (1923) interpretations of grazing impacts derived largely from interviews with old timers who had observed the country in the late 1800s. In describing the history of grazing in the South Park region, Ingwall noted that "the cattle grazed freely over all the region with little or no regulation on the part of their owners. The result was that the range was overstocked and depleted." He also corroborated the seasonal use of the region by cattle driven in from distant areas. The old timers who first brought cattle into the area in the late 1860s and 1870s claimed that the range was at least four times as good as it is today [i.e. 1920s] on the lower Platte and Buffalo ranges. They mentioned that gramma grass was the predominant grass of the early days. Both Ingwall and Jack related that the old timers reported severe droughts in the early 1880s that forced them to remove some of their stock.

Although the magnitude and spatial extent are not quantified, these early historical descriptions of the PSI document the following pattern of livestock utilization and impacts during the period from c. 1860 to 1900: 1) livestock roamed freely over the early timber reserves, even in areas that were remote from human settlements; 2) the most severe damage due to over grazing was in riparian habitats; 3) in at least some non-riparian habitats livestock damaged tree regeneration through browsing and trampling; and 4) during the first few decades of livestock impact, local observers reported a decline in productivity and shifts in species composition of some plant communities which they attributed to livestock. Although less well documented for the AR, these same patterns are believed to apply to the northern Front Range too.

6.2. Logging and Other Vegetation Treatments

Public concern over the effects of unregulated logging in western forests was a major incentive for establishment of the National Forests, and subsequently, timber harvesting at times has been the central focus of the management of these same areas. Unrestricted harvesting, burning, and livestock grazing during the early settlement period were major reasons for establishment of the first Forest Reserves in Colorado, including those in the area of the AR and PSI (e.g., Jack 1900, Ingwall 1923). Later timber harvests on National Forest lands have continued to influence landscape patterns of forest structure and composition.

6.2.1. Logging: Arapaho-Roosevelt N.F.

Impacts of early harvesting on especially the montane forests of the AR were profound. Timber harvesting in the Front Range often was accompanied by burning of slash or other accidentally-ignited fires, some of which apparently burned over very large areas (Ingwall 1923, Veblen and Lorenz 1991). Lumbering began in the AR with the very earliest settlements in the mid1800s and intensified with mining activities in subsequent decades (Wyckoff 1999). Historical photographs attest to the heavy impacts on forests surrounding mining camps in the AR (Veblen and Lorenz 1991; Appendix 1). The rapidity and extent of deforestation around early mining settlements were extreme. For example, the slopes around Central City had been described as still “densely wooded” by Horace Greeley in 1859, but by 1866 the same area was described as: “The timber has been wholly cut away, except upon some of the more distant steeps, where its dark green is streaked with ghastly marks of fire.” (quoted in Wyckoff 1999).

Later cutting was for ties for the growing railroad system in the Front Range, to the mining districts, and to the growing urban areas. The forests of Boulder County, particularly those of the montane zone, were heavily logged and burned during the nineteenth-century mining booms (Fritz 1933, Kemp 1960) as shown by historical photographs (Veblen and Lorenz 1991). Age structures of these stands spread throughout the montane zone of Boulder County today confirm that they date mostly from the late 1800s with only scarce ponderosa pine and Douglas-fir surviving the episode of logging and burning (Moir 1969, Veblen and Lorenz 1986). Although logging was more intensive in the mineralized belt such as in Boulder County, even in the area of present-day Rocky Mountain National Park extensive areas were logged to supply ranches and early tourist resorts in the late 1800s and early 1900s (Buchholtz 1983).

Much of the early timber harvesting was selective removal of the largest trees first. Although most of the mills were small, the combined effect was to remove many of the largest and oldest trees from all stands that were readily accessible and close to growing cities and towns along the Front Range. The greatest effect of this harvesting was to remove older trees and older stands from much of the montane zone. The removal of the largest and oldest trees is generally reflected in the late 20th century age structures of ponderosa pine stands which in the northern Front Range often have relatively few trees older than 100 to 200 years (Veblen and Lorenz 1986, Mast et al. 1998).

RIS data on cover types in the AR affected by logging pertains mainly to harvesting during the second half of the 20th century (Fig. 6.1; Table 6.1). These data indicate that lodgepole pine is the cover type with the largest percentage of surface area under some type of silvicultural management. Although 86% of the surface area of the AR is classified as "natural stands" (not managed in Table 6.1) this reflects only the recent status of these sites and largely ignores early logging activity. For example, most of the ponderosa pine and Douglas-fir cover types were subject to heavy logging during the second half of the 19th century (Veblen and Lorenz 1991) but this is not reflected in the RIS data on post-1950 management status (Table 6.1). However, the combined effects of early logging and stand-replacing fires are reflected in the RIS data on stand ages; both these disturbance types would have triggered new tree establishment which is reflected in the abundance of stand-origin dates in the late 19th to early 20th centuries in these cover types (Fig. 6.2).

In the AR, age data from the RIS database stand inventories indicate generally younger stands in the ponderosa pine and Douglas-fir forest types than in the lodgepole pine and spruce-fir cover types of the subalpine zone (Figs. 6.2 and 6.3). The RIS database for the AR includes a single stand each of ponderosa pine and of Douglas-fir that originated prior to A.D. 1800 despite the ability of these species to reach ages of at least 500 years in the Front Range. The absence of old ponderosa pine and Douglas-fir stands reflects selective logging of the larger trees as well as mortality caused by fires, mountain pine beetle, western spruce budworm, and Douglas-fir bark beetle outbreaks. The scarcity of old ponderosa pine and Douglas-fir in the AR is confirmed by studies based on more abundant tree data collected at the stand level (Veblen and Lorenz

1986, Mast et al. 1998, City of Boulder 1999). Overall, the generally younger ages of ponderosa pine and Douglas-fir compared to Engelmann spruce point to greater impacts of early settlement harvesting and unrestricted fires in the montane zone. For all cover types, other than spruce-fir, there is an abrupt increase in the number of stands originating during the late 1800s. This is particularly evident for lodgepole pine which established in great abundance after c. 1860 as indicated in the RIS data (Fig. 6.2) and in other studies (Moir 1969, Veblen and Lorenz 1986). This is consistent with the increase in stand-replacing fires that occurred during this period (see section 5.3) and with regeneration following logging in the early 20th century.

Impacts of 19th-century timber harvesting in the subalpine zone may not have been as great as in the montane zone because of generally less accessibility to the higher elevation forests. However, throughout the Front Range, heavy harvest occurred in the vicinity of mining communities, and large areas of the lower subalpine zone were burned and then re-established during the early settlement period (Ingwall 1923, Moir 1969, Veblen and Lorenz 1986, 1991). During the 20th century, however, most timber harvesting in Colorado has been concentrated in the subalpine zone in the spruce-fir, lodgepole pine, and aspen cover types. Clearcutting was common from the 1950s to 1970s in the all these forest types. Although the removal of the tree cover by intensive logging is superficially similar to some natural disturbances, there are important differences between the effects of logging and of disturbances such as intense fire, insect outbreaks, and blowdown. The degree of removal of organic matter from the site is much greater than in the case of natural disturbances, which leave abundant coarse woody debris on the site. Forest regeneration patches in previously logged areas typically lack the abundant dead-standing and fallen trees that are an important habitat for some wildlife species (Hutto 1995). Unlike fire, clearcut logging does not create a blackened seedbed or remove all the fine litter. Forest thinning which leaves small fuels on the site, in the absence of prescribed burning, may not reduce fire hazard, and, in fact could increase it. Furthermore, timber harvesting can have a variety of impacts on soil properties (e.g. compaction, altered carbon/nitrogen ratios) that are dissimilar to the impacts of fire on soils (Rumbaitis del Rio 2003). As discussed below, logging and associated road construction fragment the landscape and create patterns that differ substantially from those created by natural disturbances.

6.2.2. Logging: Pike-San Isabel N.F.

Lumbering began in the PSI the beginning of mining activities in the middle 1800s and intensified with mining activities in subsequent decades. “Wood was one of the greatest necessities in the early mining camps and there was always a steady demand for lumber, building logs, lagging, stulls and fuel wood, from 1859 to 1873" (Ingwall 1923:30). Later cutting was for the railroad system in the Front Range and for the growing urban areas. Jack's 1898 map shows numerous lumber mills scattered throughout the South Platte and Pike's Peak regions. Extensive tie hacking, fuel wood cutting, and saw-timber harvest impacted much of the lower elevation forests along the mountain margins, and had the result of shifts in forest and woodland boundaries with adjoining grasslands (Ingwall 1923). In describing forest conditions on the Pike's Peak, Plum Creek and South Platte Reserves (the precursors of most of the PSI), Jack (1900:43) wrote: "Of all the reserves established by the Federal Government, the three under consideration have probably been the most damaged by fire and been subject to greatest depredations by timber cutters. ...There are a very few thousand acres of merchantable timber where the ax has not been used with evident effect."

Much of the early timber harvesting was high-grading, or removal of the largest trees first. Although most of the mills were small (Jack 1900, Ingwall 1923), the combined effect was to remove many of the largest and oldest trees from all stands that were readily accessible and close to growing cities and towns along the Front Range. In 1898, Jack noted that for the area of Pike N.F. nearly all easily accessible Douglas-fir over 8 inches in diameter had been logged. In the PSI, age data from the RIS database stand inventories indicate generally younger stands in the ponderosa pine forest type than those of the subalpine zone (RIS database; Fig. 6.3). The RIS database includes no ponderosa pine stands and just one Douglas-fir stand over 200 years old despite the ability of these species to reach ages of at least 500 years in the Front Range. In general, the absence of old trees from the montane zone implies more extensive harvest and early settlement fires in the lower elevations of the PSI area, consistent with findings for the northern Front Range (Veblen and Lorenz 1986, 1991). Generally younger ages of ponderosa pine and Douglas-fir stands compared to spruce-fir reflect the greater impacts of early settlement harvesting and fires in the lower montane zone (Fig. 6.3).

Data on forest age and structural characteristics in the montane zone are available for an unlogged reference landscape on the South Platte River at Cheesman Lake (Kaufmann et al. 2000) which are consistent with the interpretations given above of the impacts of logging from the RIS data base. This study compared forest structure in the unlogged montane ponderosa pine landscape at Cheesman Lake to an adjacent landscape with a history of harvest. In all diameter classes, tree density in logged plots matched or exceeded those in the unlogged reference area. One-fifth of plots in the unlogged landscape had trees older than 400 years, but no trees older than 300 years were found in logged plots. However, equal numbers of plots in both landscapes had trees older than 200 years and distribution with aspects and topographic positions was similar between the two areas. Results from this study suggest that tree age structure of the logged landscape will regain an age distribution similar to that of the pre-settlement structure if further harvest of old trees is curtailed. Tree size structure in the unlogged landscape could be matched in the logged landscape by removing excess trees in all diameter classes in the logged landscape. However, the unlogged landscape also has a greater range of variability in open areas and stand heterogeneity that will need to be considered at a landscape scale (Kaufmann et al. 2000). An important implication of the study at Cheesman Lake, is that ecological restoration requires relatively detailed studies before appropriate restoration prescriptions can be developed. Due to the high degree of heterogeneity of stand structures during the reference period, a single prescription, such as logging all diameters smaller than a certain size, is not an appropriate prescription for the entire cover type of ponderosa pine.

6.3. Roads and Other Forms of Fragmentation

Landscape heterogeneity exists in natural ecosystems because of both underlying environmental variability and spatiotemporal variability in disturbance regimes (Knight and Reiners 2000, Veblen 2000). Human-caused alteration of natural patterns has occurred from roads, fence lines, and power lines as well as tree harvest (Knight et al. 2000). Roads, fence lines, and power lines as well as clearcut logging are widespread causes of changes in landscape characteristics, typically fragmenting formerly continuous habitats. The following discussion emphasizes the impact of logging on the historic range of variability of wildlife and other ecosystem components. Discussion of the benefits of logging to society is beyond the scope of this report, but such benefits obviously must be considered in making land use decisions.

Fragmentation of formerly continuous habitats, whether they are forest or other vegetation types, is recognized as a major threat worldwide to wildlife species (Knight et al. 2000). Fence and power lines affect only a small portion of the landscape, whereas roads and harvest units can affect large areas of forest land. Roads and fence lines may result in relatively permanent habitat fragmentation for some wildlife species but not others (e.g., birds), may provide avenues for the introduction of exotic species (Weaver et al. 2001), and may facilitate other human impacts such as increased fire ignitions along roadsides. Logging, most obviously clearcut logging but also selective logging, clearly changes landscape structure in a way that is outside of the historic range of variability (Reed et al. 1996, Tinker et al. 1998). Forest fragmentation creates habitats that are less suitable for wildlife species that require forest interior habitats while also increasing habitat for species adapted to edges. Thus, forest fragmentation can have either positive or negative influences on the size of the population of a particular wildlife species. However, forest fragmentation has the net effect of making rare species even rarer and common species more common (Beauvais 2000).

Several papers in a recent book on forest fragmentation in the southern Rocky Mountains (Knight et al. 2000) focused on landscape alteration caused by road construction and logging. Baker and Knight (2000) measured over 2200 miles (3680 km) of roads on the AR in the R2TF database for an overall road density of .75 mile/mile^2 (1.21 km per 2.62 km^2). This figure was in the middle of road density for national forests in Region 2 and, of course, this density varies considerably for different areas of the forest. Forest fragmentation caused by road construction has been examined in the southern Rocky Mountain floristic province in Roosevelt N.F. in north-central Colorado (Miller et al. 1996), Bighorn N.F. in north-central Wyoming (Tinker et al. 1998), and in the Medicine Bow N.F. in southeastern Wyoming (Reed et al. 1996). To assess the effects of clearcut logging and road construction on landscape patterns, these studies have compared landscape patterns in roadless and roaded areas (Miller et al. 1996), utilized time series of RIS data from the 1950s to 1990s (Reed et al. 1996), and interpreted landscape configurations from natural disturbances *versus* logging and roads from satellite imagery (Tinker et al. 1998). These studies document major changes in landscape patterns associated with the impacts of roads and clearcut logging. All three studies were conducted primarily in subalpine forests and their most important findings about the effects of clearcutting and roads include: 1) decreases in patch sizes; 2) increases in patch densities, total edge perimeter, and edge densities; and 3) simplification of patch shapes at the landscape scale.

Such studies describe a high degree of subalpine forest fragmentation especially since the 1950s. For example, the study in Medicine Bow N.F. showed that the landscape in 1993 was more fragmented than the landscape of the Oregon Cascades, which has been the subject of national attention and restoration action (Reed et al. 1996). These studies and others demonstrate that timber harvesting and road building have turned many western coniferous forests into landscapes of isolated patches of interior forests embedded in a matrix of human-created edges (Tinker et al. 1998). These studies (Reed et al. 1996, Miller et al. 1996, and Tinker et al. 1998) also identify some limitations on quantifying the departure from naturalness caused by roads and clearcut logging. However, there is no uncertainty that the types of habitats created by logging are outside the historic range of variability of the landscapes of the southern Rocky Mountains.

The ecological consequences of forest fragmentation by roads and logging are complex. As reviewed in Reed et al. (1996), Miller et al. (1996), Tinker et al. (1998), and Baker and Dillon (2000), studies conducted in many types of forested ecosystems around the world have shown that the effects may include: changes in nutrient cycling and microclimates along edges; shifts in species composition and abundance; increased outbreaks of forest insect pests; and decreased viability and changes in genetic structure of certain populations. Although natural processes (e.g., fire, blowdown) also can cause analogous changes, road construction and logging are generally different from natural patterns in terms of the microsites created as well as the landscape-scale patterns that are created (Mladenoff et al. 1993). Current knowledge of habitat use in the Rocky Mountain region suggests that clear-cutting and road construction are shifting mammalian populations toward more generalist and human-tolerant species at the expense of forest-adapted species (Beauvais 2000).

Based on studies in the northern Rockies and in southeastern Wyoming, logging has been shown to facilitate the entrance of exotic plant species into subalpine forests. Although closed-canopy forests, such as intact spruce-fir forests, are relatively resistant to invasion by exotic plant species, logging increases the presence of invasive exotics (Weaver et al. 2001, Selmants and Knight 2003). Even 50 or more years after logging, exotic species have been shown to persist in logged subalpine forests (Selmants and Knight 2003). Natural disturbance by wildfire or blowdown is also likely to facilitate invasion by exotic plant species due to the creation of open sites and reduced competition from native species (Weaver et al. 2001). However, due to the spread of exotic species along logging roads, logging is likely to be more effective at accelerating the rate of invasion by exotics (Parendes and Jones 2000, Weaver et al. 2001).

Roads and logging may also affect regeneration patterns of tree populations through both direct microsite alteration and through indirect alteration of fire regimes. For example, construction and subsequent abandonment of roads, railroads, and power-line rights of ways are generally observed to be more favorable to establishment of pines than to later successional species such as Douglas-fir or subalpine fir. However, sometimes there is no or little tree regeneration after clearing for railroads and subsequent abandonment. For example, the rights-of-ways along many abandoned railroad lines in the central Rockies of Colorado are characterized by abundant cut stumps indicating formerly dense forest covers and relatively sparse tree regeneration. Even up to a hundred years after abandonment, formerly forested sites still have a grass cover type. Why tree regeneration has failed at such sites is not clear and requires research.

Roads and railroads have potentially altered fire regimes through their effects on fire spread, anthropogenic ignitions, and fire suppression. For example, in the prairies and savannas of the central and southwestern U.S., it is believed that road construction in the late 19th and early 20th centuries reduced fire spread through creation of fuel breaks (Covington and Moore 1994, Archer 1994, McPherson 1997, Brown and Sieg 1999). There are numerous accounts from early settlers of prairie fires that burned over several 1000s of square kilometers (e.g., Higgins 1986), and roads may have subsequently reduced the spread of such fires. The hypothesis of reduced fire spread due to fire breaks created by roads may apply to the parklands and lower elevation grasslands of the AR, but roads are unlikely to be effective fire breaks for stand-replacing fires in dense forest of the subalpine zone. Indeed, some roads and railroads may have increased the frequency of intentional and accidental fire ignition by humans along travel routes (e.g., Jack 1900, Veblen and Lorenz 1991). Conversely, improved access due to roads may have reduced the size of fires by facilitating fire suppression.

Roads and logging also affect patterns of disturbance associated with wind storms by increasing the susceptibility of trees to windthrow where sharp edges are created (Alexander 1964). Where road construction leaves abundant logging debris on the ground, the hazard of SB outbreaks is known to increase (Schmid and Frye 1977). Studies specific to the AR and PSI are needed to evaluate the potential ecological impacts of roads on disturbance regimes.

6.4 Euro-American Influences on Fire Regimes

Based on tree-ring records of fire, two general temporal trends in fire occurrence for the FR and nearby areas in the southern Rockies are evident : 1) a period of increased fire occurrence from the mid- to late 19th century; and 2) a dramatic decline in fire frequency since the early 1900s (Veblen 2000). These trends are of variable strength in different forest types and geographical areas in the FR. In the following two sections we examine the evidence for these two patterns in different habitat types, the possible contribution of human activities to these two trends, and the ecological consequences of these changes in fire regimes.

6.4.1. Human Influences on the 19th Century Increase in Fire

Increases in fire occurrence during the second half of the 19th century have been documented for many areas in the southern Rocky Mountains, including the Front Range (Rowdabaugh 1978, Laven et al. 1980, Skinner and Laven 1983, Zimmerman and Laven 1984, Goldblum and Veblen 1992, Kipfmueller 1997, Donnegan 1999, Veblen et al. 2000, Brown and Carpenter 2001). The pattern of increased fire occurrence during the latter half of the 19th century is well documented for the ponderosa pine and Douglas-fir/mixed conifer cover type. Over a relatively broad elevational range of the montane zone this trend towards more frequent fire years, as well as increased frequency of more widespread fires, is reflected by the percentages of fire-scar sample sites recording fire in the northern and southern Front Range (Fig. 5.5). Although less extensively studied, there also has been an increase in fire occurrence in the subalpine zone, especially in the lower elevational part of this zone, during the second half of the 19th century (Zimmerman and Laven 1984, Kipfmueller and Baker 2000, Sibold 2001, Sibold et al. unpublished m.s.).

The latter half of the 19th century was a time of climatic variation favorable to extensive fires (see Section 5.2.3.2) and also a time of increased anthropogenic ignitions. As previously discussed, there is some evidence of an increase in intentional burning by the Utes in addition to abundant evidence of burning by white settlers during this period (see Section 3.4). The 1859 mineral discoveries initiated an influx of Euro-American settlers who intentionally or accidentally set many fires (Veblen and Lorenz 1986; see section 3.4). During the Euro-American settlement period (c. 1850 to 1910) in Colorado, fires were frequently set by Euro-Americans accidentally or to facilitate prospecting, to justify salvage logging, or to clear brush for ranching (Tice 1872, Fossett 1880, Jack 1900, Sudworth 1900, Wier 1987). While it is difficult to exactly determine the roles played by climatic *versus* land-use changes, the historical descriptions of fires set by Euro-American settlers strongly imply that people contributed to the high *frequency* of fire over much of the FR during the second half of the 19th century.

Despite the widely documented abundance of human-set fires, the increase in fire *extent* after the mid-19th century appears to have been controlled primarily by climatic conditions more favorable to widespread fires. As described in Section 5.3.2 the mid- and late-19th century was a period that was climatically more conducive to widespread fire. We believe that the increase in ignitions by humans probably did not greatly increase the area burned over what it would have been if the only source of ignition was lightning. Lightning is sufficiently abundant in this landscape so that ignitions are not likely to be limiting to fire occurrence. Nevertheless, the frequency of fire may have been increased by human activities during the settlement period, and the location of particular fires would have been influenced by humans.

More important than the increase in anthropogenic ignitions in the latter half of the 19th century is the fact that climatic variability was more conducive to fire spread during this period. This was a time period severe droughts reflected by synchronous fire years occurring over widespread areas of the Rocky Mountain region. Years of widespread fire are less frequent with increasing elevation from the lower montane through the upper montane and into the subalpine zones in the Front Range (Veblen et al. 2000, Donnegan et al. 2000, Sibold et al. unpublished m.s.). The greater dependence of widespread fire at higher elevation on drought implies that human influences on fires in the upper montane and subalpine zone was less than at lower elevation. Although the documentary evidence suggests that while fire frequency may have been increased by human activities in the second half of the 19th century the spread of these fires to large areas of subalpine forest in the area of FR was facilitated primarily by drought. Even if the areas burned in the late 19th century were all from fires ignited by humans, the occurrence of extensive fires during these very dry years does not appear to be outside of the HRV. For example, in RMNP the area burned in the 17th century appears to be comparable to the area burned in the late 19th century (Sibold et al. unpublished m.s.).

The widespread fires of the second half of the19th century, whether natural or caused by humans, have profoundly influenced the contemporary landscape of the FR. Extensive fires during the late 19th century were largely responsible for the abundant even-aged stands of quaking aspen, lodgepole pine, Engelmann spruce, and subalpine fir that originated across much of the subalpine zone and of Douglas-fir and ponderosa pine in much of the montane zone of the FR (Fig. 6.2 and 6.3; Moir 1969, Veblen and Lorenz 1986, Kaufmann et al. 2000, Ehle and Baker 2003, Sibold et al. unpublished m.s., Sherriff 2004, Sherriff and Veblen unpublished m.s.). Despite the great importance to modern forest structure of the late-19th century period of increased fire occurrence, such a large amount of burning is not believed to be unusual in comparison with the previous several centuries. Such high variation in the extent of fire in the FR appears to be related primarily to climatic variability, and the contribution of humans to fire extent in the late 19th century was probably secondary to the effects of climate.

6.4.2. The Decline in Fire Frequency during the 20th Century

As discussed in the Introduction, one of the key objectives of this HRV assessment is to evaluate key premises about fire history and fire suppression that often drive resource management decisions and policies. Thus, for major forest types of the FR we must address the questions: Have historic fire regimes been significantly altered by fire suppression during the 20th century? Has fire suppression resulted in unnatural accumulations of fuel that in turn will permit unnaturally severe fires? These questions must be addressed separately for the montane zone versus the subalpine zone, and within each of those elevation zones variations must be considered according to forest types.

Montane zone

A pattern of reduced fire frequency during the 20th century has been consistently found in studies of montane and upper montane fire history from the northern Front Range to the Sangre de Cristo Range (Rowdabaugh 1978, Laven et al. 1980, Skinner and Laven 1983, Goldblum and Veblen 1992, Alington 1998, Brown et al. 1999, Veblen et al. 2000; Fig. 5.4 and 5.5). The modern fire exclusion period beginning in the early 1900s refers to both suppression of lightning-ignited fires and cessation of intentional burning by humans. Fuel reductions due to heavy grazing in the late 19th and early 20th centuries also may have contributed to the decline in fire frequency near the turn of the century, which in many studies pre-dates effective fire-suppression technology by one or several decades (Veblen 2000).

In the northern Front Range, at sites below 2100 m (6880 ft) the available fire history data indicates that the pre-20th century fire regime was characterized by frequent fires (Veblen et al. 2000, Sherriff 2004, Sherriff and Veblen unpublished m.s.). Fire frequency was relatively high, sometimes recurring to the same tree in fewer than 10 years (Fig. 5.3a). Furthermore, many fire-scarred trees survived several to over 10 scar-producing fires. These low-intensity fires maintained ponderosa pine stands as open woodlands which following fire exclusion at the beginning of the 20th century increased dramatically in density as indicated both by tree ages in these stands and by historical photographs (Mast et al. 1988, Veblen and Lorenz 1991, Sherriff and Veblen unpublished m.s.). Other factors, such as logging, livestock effects, and climatic variation, probably contributed to the increase in stand densities in the modern landscape but without fire exclusion these factors would not have resulted in the observed increases in stand densities. Approximately 20% of the ponderosa pine cover type of the northern Front Range had an historic fire regime of primarily surface fires of moderately short return intervals (i.e. return intervals of 10 to 30 years in stands of 100 ha) (Sherriff 2004, Sherriff and Veblen unpublished m.s.). Substantial increases in stand densities in the lower montane zone coincides with the reduced fire frequency of the 20th century, and many such sites now could carry a crown fire where previously only surface fires occurred. Thus, in the lower montane zone of the northern

Front Range vegetation treatments and prescribed burning can simultaneously achieve the stated management goals of reducing the hazard of severe fires and restoring the landscape to a condition typical of pre-settlement times (e.g., City of Boulder 1999, Brown et al. 2001).

In the northern Front Range, at higher elevations within the ponderosa pine cover type, there is abundant evidence that forest structures were shaped primarily by infrequent stand-replacing fires rather than frequent surface fires. The pattern of post-fire cohorts dominating the age structures of these forests, as well as the scarcity of trees scarred by multiple fires, indicate that these forests were shaped primarily by stand-replacing rather than surface fires (Veblen and Lorenz 1986, Mast et al. 1998, Ehle and Baker 2003, Sherriff and Veblen unpublished m.s.). In the northern Front Range age structures from widely separated sites indicate that many ponderosa pine stands experienced recruitment episodes following a relatively small number of years of very widespread fire scars: 1654, 1786, 1813, and numerous years in the second half of the 19th century (Veblen and Lorenz 1986, Mast et al. 1998, Ehle and Baker 2003, Sherriff 2004). This implies that stand-replacing fires occurred synchronously over large portions of the montane zone. Furthermore, there are abundant landscape photographs from the late 19th century depicting both dense stands in the upper montane zone long before there were any effects of fire suppression and large areas of severe fires (Appendix 2, Veblen and Lorenz 1991). Thus, for most of the montane zone of ponderosa pine in the northern Front Range, we conclude that the historic fire regime included widespread, stand-replacing fires. For the northern Front Range, approximately 80% of the ponderosa pine cover type had fire return intervals of > 30 years, and apparently those forests were shaped primarily by stand-replacing fires (Sherriff 2004, Sherriff and Veblen unpublished m.s.).

At a stand scale of c. 100 ha long fire-free intervals (even as long as 100 years) were part of the historic fire regime for the upper montane zone and most of the area of the ponderosa pine cover type of the northern Front Range (Fig. 5.3b; Sherriff 2004, Sherriff and Veblen unpublished m.s.). However, at a landscape scale of many 1000s of ha, the 20th century is time of lower fire occurrence in comparison with previous centuries (Veblen et al. 2000; Fig. 5.5). This means that although for individual stands of c. 100 ha in the upper montane zone the relative lack of fire during most of the 20th century is not outside the longer-term HRV, although for the landscape as a whole fire occurrence has been significantly altered during the 20th century period of fire suppression. In particular, the reduction in the area burned by stand-replacing fires in the 20th century is the greatest contrast with the historic fire regime.

Has this reduction in the extent of stand-replacing fires in the upper montane zone of the northern Front Range resulted in a landscape that is significantly outside the long-term HRV in terms of stand ages? Fewer stand-replacing fires in the upper montane zone have resulted in a smaller extent of young, post-fire ponderosa pine stands than expected in the absence of fire suppression. However, this may be partially balanced by young patches originating after widespread cutting of older trees during the 1970s mountain pine beetle outbreak (Fig. 6.2) At a landscape scale, the relative proportion of stands originating prior to c. 1900 is high, but this strongly reflects the effects of widespread fires in the 19th century creating open areas for regeneration of ponderosa pine and Douglas-fir. The previous occurrence of widespread, severe fires in association with severe droughts (Veblen et al. 2000, Sherriff 2004) suggests that wide swings in stand ages at a landscape scale are an inherent feature of the upper montane zone. Nevertheless, if fire suppression continues to prevent stand-replacing fires in the upper montane zone age structures eventually will exceed the HRV of the past several centuries. However, it is likely that in future droughts, significant areas of the upper montane zone of the northern Front Range will burn in severe and widespread fires as occurred in the 2002 Hayman fire in the southern Front Range. Such fires, whether ignited by lightning or humans, will trend the age structures back towards historical conditions.

Although we have stressed that primarily stand-replacing fires have shaped ponderosa pine stands in the upper montane zone in the northern Front Range, at these elevations there also were some surface fires (Veblen et al. 2000). It is difficult to estimate the areal extent of these surface fires. It is likely that in meadows along forest edges grass fuels potentially would have supported more frequent surface fires than in dense ponderosa pine or ponderosa pine-Douglas-fir stands. It is likely that surface fires killed tree seedlings creating a positive feedback in which open stands or meadows could have been maintained by repeated surface fires if the interval between fires was not too great (i.e. less than a few decades). Field observations and historical photographs (Veblen and Lorenz 1991) show that some areas of grasslands have been partially invaded by trees. However, the magnitude of such tree invasions appears to be much less than in the lower montane zone. Suppression of low-intensity surface fires (as well as climate variability, livestock impacts, and other disturbances) may have contributed to these limited areas of tree invasions of grasslands in the upper montane zone.

For the upper montane zone in the northern FR, tree ages do not support the idea that fire suppression has resulted in unusual abundances of young ponderosa pine or Douglas-fir in forested areas of the upper montane zone (Sherriff 2004). Tree age data from > 3200 trees at 23 sample sites from the lower montane through the upper montane zone in the northern Front

Range were used to test the idea that exclusion of low-severity fires had permitted the survival of an unusually high number of tree seedlings in comparison with the period prior to fire exclusion. If fire suppression has resulted in unusually dense stands, then trees that established since effective fire suppression in c. 1920 should be abundant. However, in none of the stands at 2200 m or higher elevation did trees that established after 1920 account for more than 20% of the tree population. Instead, over 50% of the tree establishments date from the 1841-1920 period (equal in length to the 1921 to 2000 period of fire exclusion) of increased fire occurrence and livestock and mining disturbances associated with Euro-American establishment (Sherriff 2004). In both the lower and upper montane zone, there was no evidence that during the post-1920 period Douglas-fir had invaded stands in which it had been absent or unimportant in the 19th century. Thus, these tree age data do not support the interpretation for the Cheesman Lake area that during the 1900s there was a widespread conversion of stands formerly dominated purely by ponderosa pine to mixed stands with Douglas-fir. We stress that for the upper montane zone of the ponderosa pine cover type in the northern Front Range, stands with high densities of trees, often including Douglas-fir, have not resulted from suppression of formerly frequent surface fires.

As discussed in section 5.2.3.1, these interpretations of the consequences of reduced 20th century fire occurrence for the montane zone in the northern Front Range (Veblen et al. 2000, Sherriff 2004, Ehle and Baker 2003) share both similarities and differences with the interpretations for ponderosa pine forests in the southern Front Range (Kaufmann et al. 2000, 2001 et al. 2001). In both areas there is strong evidence that stand-replacing fires were an important component of the historic fire regime. Both sets of studies found that low-severity fires were not frequent in the historic fire regime. In the Cheesman Lake studies emphasis is placed on modeling results indicating a major shift from open areas in 1900 to stands of closed canopies in the modern landscape. It is not certain if the open areas stressed in the Cheesman Lake studies would have been maintained by repeated low-severity fires or if they were treeless due to the effects of severe fires followed by a lack of regeneration. Open areas existed in the historic montane landscape of northern Front Range too, but no one has attempted to precisely estimate the area of open areas at a particular date. Given the high degree of variation in the fire regime of the montane zone at multi-decadal and centennial time scales and the variability in the success of post-fire tree regeneration related to fire severity and probably climatic variability, we suggest that the proportion of the montane landscape in open versus forested condition fluctuated widely over the past several centuries.

Has 20th century fire suppression resulted in the potential for larger or more severe fires in the montane zone of ponderosa pine forests? Based on knowledge of the historic fire regime and past forest conditions, fire suppression in the lower montane zone of ponderosa pine forests has altered fuel conditions so that potentially sites can support more severe and more extensive fires. Increased ladder fuels and more contiguous fine fuels in the canopy now will support crown fires where formerly the predominant fire type was non-lethal, surface fire.

In contrast, in the mid and upper montane zone, this pattern of change applies to a small fraction of the landscape. In the upper montane zone of both the ponderosa pine and Douglas-fir / mixed conifer cover types, fire suppression has had a much smaller effect on fire extent and severity. For example, after considering all the evidence on fire history for the Front Range (both north and south), a review team concluded as follows for the 2002 Hayman fire in the montane zone of Pike N.F.:

> "Comparing the 2002 Hayman Fire with the historical fire record developed in the Cheesman Lake study area and elsewhere in the Front Range, we conclude that the size (total area burned) of the Hayman Fire was not unusual. ...The fact that portions of the Hayman Fire were high-severity and stand-replacing also was not unusual; many historical fires contained a significant stand-replacing component. However, the size and homogeneity of patches of high-severity, stand-replacing fire in 2002 clearly were unprecedented in the Cheesman study area during the last 700 years. ..." (Romme et al. 2003a).

For the Cheesman Lake study area, increased canopy closure during the 20th century in comparison with a much larger percentage of the landscape in openings created by high-severity fires, is believed to have contributed to the severity of the 2002 Hayman fire (Romme et al. 2003b, p. 168). This report concludes for the montane zone of ponderosa pine forests in the Cheesman Lake area that there is "strong evidence that current fire effects and landscape structure are well outside the historical range of variability. In other areas, however (such as much of Boulder County), fires as large and severe as the Hayman Fire appear to be a component of the historical range of variability." (Romme et al. 2003b, p. 170). A key conclusion of the report on the Hayman fire is that the importance of the extreme drought to the severity of the fire "cannot be overemphasized" (Romme et al. 2003a, p. 162). This is consistent with the observation that "fuel modifications generally had little influence on the severity of the Hayman Fire during its most significant run on June 9" (Finney et al. 2003, p. 105), and that the low fuel moisture and weather in combination with the large size of the fire overwhelmed most fuel treatment effects in areas that burned on the day of the fire's most significant run.

Overall, in the Cheesman study area, the effects of fire suppression appear to have contributed to the size and severity of the 2002 Hayman fire. However, that event was driven primarily by extreme weather, and there is abundant evidence that high severity fires have burned large areas of montane forest in the Front Range prior to any effects of fire suppression. Although there is uncertainty of the magnitude of increased potential for fires of greater extent or severity that can be related to fire exclusion, there is a consensus among all researchers that severe and widespread fires associated with extreme weather are a natural feature of the historic fire regime of the montane zone of the FR.

Subalpine zone

In the subalpine zone, due to the long intervals between successive fires in the historic fire regime it is difficult to ascertain if fire suppression has had a significant impact on fire occurrence in these high elevations. Tree-ring records of fire in subalpine forests in northern Colorado do show fewer fire years for most of the 20th century in comparison with the second half of the 19th century (Sibold 2001, Kulakowski and Veblen 2002, Kulakowski et al. 2003, Sibold et al. unpublished m.s.). However, it is not clear how much of this reduction may be attributed to fire suppression activities as opposed to climatic variability or the occurrence of young less fire-prone forests due to abundant post-fire stand establishment in the late 19th century. During the 20th century many fires have been suppressed in subalpine forests that would have spread to larger areas in the absence of suppression. However, there is no way of knowing for sure how large an area would have burned in the absence of these suppression efforts. For large areas of spruce-fir and lodgepole pine forests of Yellowstone and Rocky Mountain National Parks, Romme and Despain (1989) and Clagg (1975) previously have questioned if fire suppression has significantly altered the natural fire regimes. In contrast, in the Medicine Bow Range of southern Wyoming, Kipfmueller and Baker (2000) attribute to fire suppression a decrease in the area burned by stand-replacing fire after the beginning of fire suppression in c. 1912.

In subalpine forests of Colorado's Front Range, fire suppression efforts in National Forests have been substantial and most recent fires have been relatively small. Few large fires occurred in the FR area after c. 1910 and potentially this is explained by fire suppression. However, how much has fire suppression reduced the area burned over what it would have been in the absence of fire suppression? We believe that it is impossible to precisely determine the effectiveness of fire suppression in the absence of long-term experiments. In other words, this question can only be fully resolved by monitoring fires in identical large areas with and without fire suppression activities. Such a large-scale and long-term experiment is impractical, and, instead the relevant question to examine is: *Has fire occurrence since the beginning of fire suppression in the early 20th century been inside or outside the long-term historic range of variability of fire occurrence?*

The most extensive fire history study conducted in subalpine forests in Colorado shows that the 20th century period of relatively little fire occurrence is not outside the historic range of variability for RMNP (Sibold 2001, Sibold et al. unpublished m.s.). For the 30,000 hectare study area spruce-fir and lodgepole pine forests, taken as a whole and also for individual watersheds of c. 6000 hectares, between c. 1600 and 1850 A.D. there were 80 to 100 year periods of no or little fire comparable to the pattern of fire in the 20th century. Likewise, in c. 4000 ha study areas in Routt and White River National Forests, fire history studies often show intervals of > 100 years without large fire events (Kulakowski and Veblen 2002, Kulakowski et al. 2003, Howe and Baker 2003). Because of the high variability in areas burned at a centennial scale and the long fire intervals typical of subalpine forests, the 20th century period of low fire occurrence is not outside the historic range of variability for these ecosystems. This generalization for the subalpine zone as a whole needs to be evaluated for the major cover types of this zone.

Spruce-fir cover type. Particularly for the wetter spruce-fir habitats, the 20th century pattern of low fire occurrence is not atypical because large percentages of this forest type have not burned in the last 400 years (Sibold et al. unpublished m.s.). If a reduction in occurrence of stand-replacing fires has occurred in the FR, it is unlikely that it is reflected in stand conditions of the spruce-fir cover type. This tentative interpretation is based on the slow rates of stand development and the long natural intervals between widespread stand-replacing fires in the subalpine zone relative to the short period of fire exclusion. Furthermore, the wide range of stand-origin dates for spruce-fir in the FR RIS data suggest that stand-initiating disturbance (either fires or logging) has not ceased during the 20th century (Fig. 6.2 and 6.3)

Lodgepole pine cover type. For lower elevation forests of lodgepole pine it is more likely that fire suppression activities may have significantly reduced the area burned in the 20th century over what it would have been in the absence of suppression. These forests are more flammable than high elevation spruce-fir forests and are more accessible for suppression activities. While the fire-free period of the 20th century is not outside of HRV for the subalpine zone taken as a whole, suppression activities may have reduced the area burned in lodgepole pine forests. If so, then the area of young (< 80 years old) lodgepole pine forest may be smaller than what it would have been in the absence of fire suppression. The age structure of this forest type is heavily influenced by the occurrence of fires in the late 1800s (Fig. 6.2 and 6.3), which resulted in young less fire-prone forests dominating this cover type for most of the 20th century. Thus, the relatively small number of lodgepole pine stands that originated since the 1920s may be partially attributable to fuel structures less favorable to fire spread under all but the most extreme drought conditions.

During the reference period, stand-replacing fires, rather than surface fires, were the modal fire type that shaped the structure of most lodgepole pine stands. Thus, there is no evidence that tree densities have increased at stand scales because of fire suppression. Reduced frequency of stand-replacing fires in the lower subalpine zone of lodgepole pine may have allowed a shift in dominance from lodgepole pine towards spruce-fir in some habitats. This would have to be determined on a site-by-site basis because there are habitats in the Front Range where lodgepole pine is self-replacing and other habitats where it is seral to spruce and fir (Peet 1981). At a regional scale, however, increased burning during the late 1800s probably substantially increased the area of the lodgepole pine cover type in the FR. For example, in both the upper montane and lower subalpine zone many of the even-aged lodgepole pine stands of the FR originated during the late 19th century period of increased burning (Moir 1969, Veblen and Lorenz 1986). Given the high variance in the extent of severe fires in the lodgepole pine cover type at a centennial time scale (Sibold et al. unpublished m.s.), the relative abundance of the lodgepole pine cover in the FR is probably well within the historic range of variability.

Aspen cover type. Given the lack of fire history research that has focused specifically on the aspen cover type, we tentatively conclude that most of the habitat of aspen had an historic fire regime of stand-replacing fires similar to the surrounding stands of lodgepole pine and spruce-fir. Given its seral status in some habitats and the high variance in fire occurrence historically in the subalpine and upper montane zone, it is likely that the extent of aspen in the landscape would have fluctuated greatly over the past several hundred years.

In the FR, fires are essential in preventing conifers from successionally replacing aspen in many areas and therefore to the persistence of aspen in landscape. In the FR, in comparison with western Colorado, a larger percentage of aspen stands appear to be seral rather than self-replacing. For the FR it is reasonable to hypothesize that the extensive burning in the upper montane and lower subalpine zones during the late 19th century, triggered high rates of aspen regeneration. Furthermore, reduction of browsing by native herbivores due to hunting pressure as well as logging in the early 20th century would be expected to be favorable to aspen regeneration. These factors may have resulted in a substantial increase in the amount of aspen in the FR by the mid-20th century in comparison with its extent in the mid-19th century. Regional scale data for the FR are not available to test this hypothesis.

Given the seral status of aspen in a significant portion of the FR, it is likely that continued, effective fire suppression would result in some reduction in its extent. However, current stand ages of aspen in the RIS data base, a broad-scale survey of aspen stands in the northern Front Range (Kashian et al. 2004), and in RMNP do not imply that aspen is in significant decline in the FR.

7. SUMMARY OF HISTORICAL RANGE OF VARIABILITY BY COVER TYPE

In this final chapter we summarize the key findings about historical range of variability for each major cover type (Table 7.1). We also briefly mention some possible management implications of our findings, but we recognize that such decisions also depend on socio-economic, political, and other factors outside the scope of this assessment.

7.1. Ponderosa Pine Cover Type

In comparison with reference conditions, many ponderosa pine stands in the FR are denser, have fewer large trees and snags, and are more homogeneous in tree age and size. The abundance of ponderosa pine stands older than c. 150 years at both landscape and regional scales is less than what it was prior to logging in the late 19th century. The accessibility of most lower elevation forests to harvest facilitated the removal of larger and older trees from stands which has resulted in less structural diversity in stands and across landscapes.

During the 20th-century period of fire exclusion, many formerly open park-like stands in the *lower montane zone* (i.e. below c. 2100 m; 6880 ft) have converted to dense young stands.

Although the extent of open park-like stands has clearly declined at lower elevations, some small dense patches also existed during the reference period. In much of the lower montane habitat of ponderosa pine, increased stand densities have resulted in increased probability of crown fires in comparison with historic low-severity fires, and probably have also increased susceptibility to some pathogens in comparison to the reference period. There is some uncertainty about what proportion of the ponderosa pine cover type fits a general model of conversion from open to dense stand structures that can be attributed to fire exclusion, but for the northern Front Range an estimated 20% of the ponderosa pine zone fits this lower montane model.

In most of the mid and upper montane zone of ponderosa pine, the historic fire regime was highly variable spatially and temporally including both high-severity and low-severity fires. The fire regime was dominated by severe, stand-replacing fires that were the main fire type that shaped stand structures. In the mid and upper montane zone low-severity fires at short enough intervals to prevent ponderosa pine seedlings from surviving to maturity played a relatively minor role. Areas of meadow (perhaps with more frequent low-severity fires) and open, severely burned areas were juxtaposed with extensive areas of dense ponderosa pine. Thus, within the ponderosa pine cover type, site-specific knowledge is needed to determine if a proposed vegetation treatment would move the vegetation condition closer to or further from historic conditions. We stress that a single prescription aimed at creating a uniformly low tree density in ponderosa pine stands is not consistent with knowledge of the importance of high-severity fires in the mid and upper montane zone.

Currently dense stands of ponderosa pine in the mid and upper montane zone primarily reflect episodes of tree regeneration following widespread stand-replacing fires in the second half of the 19th century, regeneration after logging and disturbances associated with mining. We stress that these dense stands for the most part are not the result of exclusion of formerly frequently surface fires. Dense post-fire stands were an inherent feature of the historic landscape of the mid and upper montane zone of ponderosa pine forests.

Mistletoe infestation and mountain pine beetle outbreaks have been widespread during the 20th century in this cover type, but they also occurred during the reference period. In the lower montane zone, at a stand level, fire exclusion has probably increased susceptibility of ponderosa pine to insect and pathogen infestation. At a landscape or regional scale, however, pre-1900 data on insect and pathogen levels are insufficient for identifying any differences in the extent or severity of these biotic disturbances between the modern and the reference periods.

In some ponderosa pine forests, mostly at low elevations, fire history data and tree age structures clearly show major increases in stand densities during the 20th century period of fire exclusion. In such stands, restoration of original stand structures eventually may be attained through a combination of silvicultural removal of small diameter trees (with retention of older trees) and restoration of more frequent low-intensity prescribed burns. Such ecological restoration is compatible with reduction of hazard of catastrophic fire and insect outbreak, but must also consider the risk of exotic weed invasions following burns, as well as smoke hazards to the public. Ecological restoration at higher elevation sites of ponderosa pine should consider temporally and spatially variable fire regimes of these habitats. Since most of the mid and upper montane zone of ponderosa pine was characterized by relatively dense rather than open stands, thinning is not a suitable prescription for ecological restoration.

7.2. Douglas-Fir / Mixed Conifer Cover Type

The Douglas-fir / mixed conifer cover type also has been drastically transformed by logging. Today's forests have very few old trees or snags, are dense, and are less diverse in size and age structure than what we believe the reference conditions were. The impact of logging on forest structure is certain, but large trees also tend to be selected by bark beetles that have killed large numbers of Douglas-fir.

The Douglas-fir cover type had an historic fire regime dominated primarily by high severity fires. The relative importance of low-severity fires in this cover type appears to have been less than in the ponderosa pine cover type. There is no evidence to support the view that suppression of formerly frequent surface fires has resulted in a major increase in the abundance of young Douglas-fir. Reduction in stand-replacing fires would be expected to eventually favor increased dominance by Douglas-fir in mixed stands with ponderosa pine. However, in the northern Front Range, extensive data on tree ages do not indicate that there has been a significant shift from ponderosa pine towards dominance by Douglas-fir during the twentieth century.

The extensive burning and logging of Douglas-fir forests during the latter half of the 19th century synchronized regeneration over large areas of the upper montane zone. The combined results from these early settlement activities are more homogeneous age and size structures across landscape and regional scales. Reduced diversity of stand ages at a landscape scale may have implications for wildlife habitat, insect outbreaks, and fire risk. However, it is likely that this landscape has previously experienced episodes of large, severe fires in association with extreme drought that would have similarly synchronized stand ages over large areas.

Regionally, outbreaks of western spruce budworm have been synchronous and extensive during the latter half of the 20^{th} century, and it has been hypothesized that land-use history has increased the severity and/or synchrony of such outbreaks. However, tree-ring reconstructions indicate synchronous and severe budworm outbreaks over extensive areas also occurred during the reference period at least as early as the 17^{th} century. Human-caused changes in stand structures certainly affect susceptibility to budworm attack at the stand level. However, at a regional scale the relative importance of climatic variation in favoring outbreaks appears to be greater than any effects related to the history of land use or fire suppression. Current knowledge does not support the view that 20^{th} century budworm outbreaks have increased significantly in frequency or severity in comparison with reference conditions. At a regional scale, elimination of severe insect outbreaks is neither possible nor consistent with reference conditions.

The relative lack of old trees in the Douglas-fir cover type, due both to logging and increased burning in the 19^{th} century may be outside the HRV for this type. This implies that for purposes of ecological restoration, protection of old Douglas-fir (and old ponderosa pine) would be necessary to assure a greater diversity of age structures in the future. In the northern Front Range, young Douglas-fir do not appear to be unusually abundant in comparison with historic patterns. Aggressive thinning to create extensive areas of park-like stands of Douglas-fir to be maintained by frequent surface fires is not consistent with the historic range of variability. Thus, the Douglas-fir cover type is an example of where aggressive fuels reduction would have to be justified primarily on the basis of fire hazard to humans and structures rather than as ecological restoration.

7.3. Shrubland Cover Types

No comprehensive study that compares reference and present conditions has been conducted in the Gambel oak or mountain mahogany shrublands. These community types have probably been significantly influenced by livestock grazing. Due to their location at lower elevations in the foothills, large percentages of these community types have been converted to urban and exurban land use. Based on research conducted in southwestern Colorado, Gambel oak shrublands appear to have historic fire regimes of relatively infrequent, stand-replacing fires. Years of widespread fire in this cover type appear to depend more on extreme drought than on fuel accumulation. It is unlikely that fire suppression has greatly altered the fire regimes of Gambel oak woodlands. However, we caution that in the absence of fire history studies conducted in Gambel oak shrublands in the FR that this conclusion is tentative.

7.4. Pinyon-Juniper Woodland Cover Type

Pinyon-Juniper woodlands also due to their location at low elevation have been heavily used for ranching purposes, and more recently for exurban development. Ecological studies are not available for this community type in the PSI, and generalizations must be based on studies conducted in southwestern Colorado. Studies of pinyon-juniper woodlands in southwestern Colorado indicate that the historic fire regime was dominated by severe, stand-replacing fires rather than by non-lethal surface fires. Thus, any effects of 20th century fire suppression do not include a major shift in fire type from surface to stand-replacing fires or an unprecedented accumulation of fuels. Even if the total area burned during the 20th century was reduced by fire suppression, the alteration of the fire regime has been relatively slight because of the long intervals between severe fires.

7.5. Quaking Aspen Cover Type

The aspen cover type has been widely affected by logging, grazing, and, at a stand scale, by altered fire regimes. The heavy livestock grazing in FR during the late 19th and early 20th centuries, may have impacted aspen communities during that time period but did not prevent a large number of stands from originating during that period. Locally, browsing by elk has been observed to impede aspen regeneration but at broader spatial scales herbivory by native or introduced animals does not appear to endanger the future presence of aspen in the FR.

At many sites, fire favors regeneration of aspen and retards its successional replacement by conifers. At other sites, however, aspen forms self-replacing stands due to site conditions, lack of seed of the conifers, and/or fire regime. The extent and defining conditions of seral and self-replacing stands have not been determined for the FR. In addition, we have no information on the extent or condition of aspen stands prior to the late 19th century for comparison with present distribution and age structures. It is probable that logging and burning during the late 19th and early 20th centuries increased the area of aspen over its former extent. Thus, although there are many sites in the landscape today where conifers are successionally replacing aspen, this trend may be a return to conditions more typical of the reference period. Recent assessments of the regeneration status of aspen in the northern Front Range have concluded that the species is not in imminent danger of drastic decline.

Logging may have reduced the extent of older, self-replacing aspen stands and resulted in their replacement by young aspen-dominated stands. Conversely, regeneration after logging probably also accounts for many of the young aspen stands in the FR. At a regional scale, it is not known to what extent the regeneration of aspen depends on disturbance by fire or by logging. We stress that the available evidence on the regeneration status of aspen does not support proposed acceleration of logging in order to create a missing younger age class. Given the scarcity of old aspen stands, logging of old stands would further deviate age structure from reference conditions.

7.6. Lodgepole Pine Cover Type

The lodgepole pine cover type has been widely affected by logging but perhaps less dramatically so than the ponderosa pine and Douglas-fir cover types of the montane zone. Logging of large old lodgepole pine may have contributed to the scarcity of stands older than 200 years in the RIS database. However, extensive burning during the late 19th century clearly has resulted in an abundance of post-fire stands of 100 to 140 years of age in the FR.

The historic fire regime of lodgepole pine forests consisted of mainly severe, stand-replacing fires. Low-severity fires occurred infrequently within the same stand and affected small percentages of the lodgepole pine cover type. The rare occurrences of low-severity fire in lodgepole pine did not have major effects on tree mortality or recruitment patterns. We stress there is no evidence that currently dense stands of lodgepole pine can be related to reduced occurrence of surface fires. Instead, high stand densities are an inherent feature of the development of these forests following their characteristically severe burning.

Although at a stand scale, fire-free intervals of a century or more are well within the range of intervals in the historic regime, at a landscape scale fire suppression may have decreased the abundance of young stands relative to what would be expected in the absence of fire exclusion. Nevertheless, past episodes of widespread and severe fires associated with drought implies that the age structure of this cover type varied dramatically over the past several centuries. Following regionally extensive fires in the past there were probably vast areas of similarly aged lodgepole pine stands, analogous to the modern landscape that strongly reflects the great extent of late-19th century fires. If 20th century fire suppression has reduced the number of young post-fire lodgepole pine stands, regeneration (including planting) after logging during the 20th century has partially offset that trend.

The main pathogens and insect pests of this cover type, mistletoe and mountain pine beetle, probably underwent substantial fluctuations in their extents in relation to past episodes of burning and synchronization of stand ages. At a stand scale, susceptibility of lodgepole pine stands to these pathogen and insect outbreaks can be related to stand ages determined by logging and fire suppression. However, at a regional scale these biotic disturbances are probably within their widely fluctuating natural range.

7.7. Limber Pine and Bristlecone Pine Cover Types

The limber pine and bristlecone pine cover types occupy relatively small areas in the FR. Although some stands were affected by early logging and grazing, many stands are located at high elevation sites less affected by human activities. Extraction of logs, for example along railroad rights-of-ways, for construction and for fuelwood during the late 1800s must have altered forest structures, but neither type has been the object of major logging operations. Fire history in this cover type shows a modest decline in the occurrence of small, low-severity fires in areas near South Park in Pike N.F. and no decline at remote treeline sites in the northern Front Range.

White pine blister rust has entered the FR, and there is a high probability that abundance of one or both of these pine species will significantly decline in the near future. Given the absence of any other known pathogen that would have caused widespread mortality in these pine species, such an infestation by an introduced pathogen will be a major departure from the historical range of variability.

7.8. Engelmann Spruce-Subalpine Fir Cover Type

The spruce-fir cover type has been severely affected by logging, including both selective cutting during the 19th century and commercial clearcut logging during the mid-20th century. Logging has reduced the extent of older stands compared to reference conditions but the magnitude of this reduction is unknown. Large, severe fires in this cover type are highly dependent on extreme weather conditions that occur infrequently. The historic fire regime is one of high variance so that large portions of the landscape would burn in single events, and then long intervals (sometimes centuries) would pass before the next extensive fire at the same site. Given the long fire-free intervals typical of this cover type, the 20th century fire regime does not appear to have deviated significantly from the historical range of variability.

Analogously, there is no reason to believe that current levels of insect and pathogen infestation are outside the historical range of variability in the spruce-fir cover type. We stress that there is no evidence to support the view that fire suppression in the spruce-fir cover type has resulted in significant increases in pathogen or insect problems during the 20th century. Overall, the current fire regime and forest conditions, except for the effects of logging, are believed to have changed relatively little from the reference period.

7.9. Other Vegetation Types

Although outside the scope of this study, a few comments on changes in non-forest vegetation types are appropriate. At alpine treeline, both the krummholz zone of spruce-fir and the tundra are considered fragile ecosystems in terms of their slow recovery following any type of human-caused disturbance (Bowman et al. 2002). The krummholz zone has not experienced widespread human impacts primarily because of its remote location. However, near mining areas cutting of fuelwood in the late 19th and early 20th century has had a lasting impact. Domestic cattle and sheep have grazed alpine areas in the Front Range and locally have had effects on plant community composition (Bowman et al. 2002). Recreational use of the alpine zone and increased road access has had severe, but localized, impacts on these high elevation ecosystems. Tundra ecosystems are particularly susceptible to the effects of atmospheric deposition from air pollution due to their low capacity to buffer atmospheric inputs. Increased nitrate concentrations have been measured in tundra ecosystems of the Front Range in the late 20th century and are having significant impacts on tundra communities as well as downstream aquatic communities (Bowman et al. 2002). Clearly, the biogeochemistry of alpine ecosystems is now outside the historic range of variability.

Riparian vegetation has probably been more modified by human activities than any other community types in the Front Range. At low elevations, bottom lands have often been converted to agriculture and home sites and also provide natural routes for road and railroad construction (Peet 2000). Non-native species tend to be more abundant in riparian vegetation than in adjacent forest communities (Wohl 1992). In the Front Range, extensive mining has directly changed stream courses, sediment loads, and water quality profoundly altering riparian communities (Wohl 1992). Impacts have been greatest in the montane zone but even high elevation riparian areas have been affected by mining. Riparian vegetation is often preferred habitat for livestock, and grazing and browsing is often severe in these communities. It is likely that most of riparian communities of the Front Range are outside their HRV due to chronic impacts from livestock, construction, and past mining activities.

Although also outside the scope of this assessment of forest conditions, it is likely that some montane and subalpine meadows and sagebrush-dominated parklands are outside of their HRV where they are heavily grazed or where exotic plant species have invaded. In some of these communities, exotic plant species are now significant components of these communities and in the montane meadows and parklands there has have been a long history of livestock grazing (Seastedt 2002).

8.0. OVERVIEW AND RESEARCH NEEDS

Findings presented in this report are expected to be used as baseline components in an integrated forest management planning process. The recent Committee of Scientists report (1999) suggests that sustainable ecosystems must be at the heart of any consideration of what constitutes sustainable economic and societal goals, and that management to attain sustainability must be based on knowledge of the historical processes that have shaped ecosystems. Without a sustainable ecosystem, natural resources that meet social and economic needs may be compromised in the future. We recognize that knowledge of historical range of variability is only one element in the management decision process (Wagner et al. 2000, Committee of Scientists 1999).

This assessment of HRV of the Front Range forest ecosystems does not assume that prior to massive Euro-American settlement there were no human impacts. Instead, we describe what is known about Native American land-use practices, and discuss their possible influences in different habitats. Greatest impacts by Native Americans probably occurred in low-elevation areas such as at the ecotone between the Plains Grassland and ponderosa pine woodlands. Although Native Americans may have had a slight influence over a large percentage of the forested landscape, we did not find evidence that Native American burning or any other land-use practice was a driving factor in shaping most of the forested area. Thus, human activities prior to permanent settlement by Euro-Americans in the mid-19th century appear to have had a slight, although spatially variable, impact on the FR landscape.

Changes in forest ecosystem conditions in the FR over the past century and a half can be attributed to both human and natural (mainly climatic) influences. In the early settlement period (c. 1860 to 1910) unregulated timber harvest, and livestock grazing were the most significant anthropogenic influences on the FR. Livestock grazing was probably the most pervasive impact, especially in grassland ecosystems, but at low elevations logging had a dramatic and long-lasting effect on forest structures. Euro-American settlers were responsible for some fires, and in particular areas and habitats fires probably occurred at a somewhat elevated frequency. However, given the dependence of widespread fires in the subalpine zone on extreme drought and the abundance of natural lightning ignitions, it does not appear that Euro-American settlers played a very important role in the increase in burning that occurred near the end of the 19th century. Instead, that episode of widespread burning, which has profoundly affected the modern landscape was largely facilitated by natural climatic variability.

Climatic variability appears to also be the primary driver of regional-scale outbreaks of forest insect pests. Outbreaks of bark beetles and spruce budworm have been significant in the second half of the 20th century, and it is tempting to assume that they are undesirable consequences of past land-use practices including fire suppression. However, the documentary and tree-ring records indicate that all the major insect pests of the FR experienced severe and widespread outbreaks prior to any effects of fire suppression. Long-term tree-ring studies have failed to find significant differences between the patterns of late 20th century and pre-20th century budworm outbreaks. Although at a stand scale forest structure affects susceptibility to the major insect pest, at a regional scale insect outbreaks appear to be more strongly driven by climatic variability than by the history of land use that might have affected forest structure.

The most significant findings of this assessment involve the questions stated in the Introduction (see Section 1.3) about how modern fire exclusion has affected fire regimes and forest conditions. The certainty with which we have been able to answer those questions varies with forest type (Table 7.1). There is a high level of certainty that for the spruce-fir cover type of the FR current fire regimes and forest conditions are not outside their historic range of variability. At the opposite end of the elevation gradient, there is a high degree of certainty that in the lower montane ponderosa pine forests, exclusion of moderately frequent non-lethal fires has coincided with substantial increases in stand densities and fuel configurations. In the mid and upper montane zone the greater complexity of the historic fire regime makes it more difficult to arrive at generalizations. However, one highly important generalization about the ponderosa pine and Douglas-fir cover types of the mid and upper montane zones is that large, high-severity fires were an important component of their historic fire regimes.

During the latter half of the 20th-century, landscape-scale patterns have been significantly altered by logging and road construction. The greatest consequences of the landscape fragmentation are likely to be for wildlife populations, but this is an area for which there is relatively site-specific research in the FR.

One of the most significant overall findings of this report, is the demonstration that generalizations about historic range of variability cannot be applied indiscriminately across different forest cover types. Premises and broad generalizations must be critically assessed for distinct forest ecosystem types. Even within the same forest ecosystem type (e.g. the ponderosa pine cover type) there sometimes is ecologically significant variation related to elevation and/or topographic influences on historic fire regimes. Thus, a main message of this assessment is that a single management prescription will not fit all forest cover types. Data specific to the area to be managed must be evaluated to determine the suitability of general management prescriptions for goals related to fire hazard mitigation and ecological restoration as well as many other potential management goals.

One of our aims is to identify critical areas where further is needed for informing resource planning and decision making. The following are such areas of needed research:

1) Research is needed in the ponderosa pine cover type to better relate variations in historic fire regime to abiotic habitat. Specifically, a broad-scale analysis of fire and potential local drivers of fire regime similar to recent work in the AR (e.g. Sherriff and Veblen unpublished m.s.) needs to be conducted in the PSI across the full elevational range of ponderosa pine.

2) Research is needed on past forest conditions throughout the Douglas-fir and mixed conifer cover type in the FR.

3) Further research is needed on the history of insect outbreaks in the FR. Although substantial tree-ring work has been conducted on the history of budworm outbreaks, there is a need to expand that data set. There have been no tree-ring based reconstructions of Mountain Pine Beetle or Spruce Beetle outbreaks.

4) The relationship of the seral status of aspen to habitat factors is an urgent research need. Both seral and self-replacing stands are known to exist, but a large-scale survey of habitat factors in relationship to stand age structures should be a high priority. Work in the PSI to complement the aspen assessment currently underway in the AR should be a high priority.

5) Studies are needed on the effects of livestock grazing and herbivory by native ungulates across a range of ecosystem types in the FR. Historical approaches to this topic are probably not feasible but experimental approaches with permanent plots and animal exclosures could

yield highly useful results.

6) Research is needed on how management activities and resource use (logging, grazing, recreation) affect the spread of invasive alien species.

7) Studies specific to the FR are needed to determine the effects of fragmentation associated with logging and road construction on wildlife populations.

8) Studies specific to the FR are needed for fire history and successional dynamics in pinyon-juniper woodlands, Gambel oak shrubland, and other shrubland types.

9. LITERATURE CITED

Adams, D.K. and A.C. Comrie. 1997. The North American monsoon. Bulletin of the American Meteorological Society 78:2197-2213.

Agee, J.K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C.

Agee, F.B., and J.M Cuenin. 1924. History of Cochetopa National Forest. The Salida Mail, Salida, Colorado.

Alexander, R.R. 1964. Minimizing windfall around clear cuttings in spruce-fir forests. Forest Science 10:130-142.

Alexander, R.R. 1984. Natural regeneration of Engelmann spruce after clear-cutting in the Central Rocky Mountains in relation to environmental factors. USDA Forest Service, Fort Collins, CO. Research paper RM-254.

Alexander, R.R. 1987. Ecology, silviculture and management of the Engelmann spruce-subalpine fir type in the central and southern Rocky Mountains. USDA Forest Service, Agricultural Handbook No. 659.

Alington, K. 1998. Fire history and landscape pattern in the Sangre de Cristo mountains, Colorado. Ph.D. Dissertation, Colorado State University, Fort Collins, Colorado.

Allen, C.D., and D.D. Breshears. 1998. Drought-induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings of the National Academy of Sciences 95:14839-14842.

Allen, T.F.H., and T.W. Hoekstra. 1992. Toward a unified ecology. Columbia University Press, New York, New York.

Allen, C.D., J.L. Betancourt, and T.W. Swetnam. 1998. Landscape changes in the southwestern United States: techniques, long-term data sets, and trends. *in* T. Sisk, editor, Perspectives on the land use history of North America: a context for understanding our changing environment. USGS Biological Science Report USGS/BRD/BSR-1998-0003:71-84.

Allen, R.B. and R.K. Peet. 1990. Gradient analysis of forests of the Sangre de Cristo Range, Colorado. Canadian Journal of Botany 68:193-201.

Amman, G.D. 1977. The role of the mountain pine beetle in lodgepole pine ecosystems: impact on succession. Pages 3-18 *in* W.J. Mattson, editor, Arthropods in forest ecosystems. Proceedings in the Life Sciences. Springer-Verlag, New York, New York.

Amman, G.D. and K.C. Ryan. 1991. Insect infestation of fire-injured trees in the Greater Yellowstone Area. USDA Forest Service, Research Note INT-398, Ogden, Utah.

Anderson, L., C.E. Carlson, and R.H. Wakimoto. 1987. Forest fire frequency and western spruce budworm in western Montana. Forest Ecology and Management 22:251-260.

Aplet, G.H., R.D. Laven, and F.W. Smith. 1988. Patterns of community dynamics in Colorado Engelmann spruce and subalpine fir forests. Ecology 69:312-319.

Archer, S. 1994. Woody plant encroachment into southwestern grasslands and savannas: rates, patterns, and proximate causes. Pages 13-68 *in* M. Vavra, W.A. Laycock, & R.D. Pieper, editors, Ecological implications of livestock herbivory in the West. Society of Range Management, Denver, Colorado.

Auclair, A.N.D. 1993. Extreme climatic fluctuations as a cause of forest dieback in the Pacific Rim. Water, Air and Soil Pollution 66:207-229.

Averill, R.D., L. Larson, J. Saveland, P. Wargo, J. Williams, M. Bellinger. 1995. Disturbance processes and ecosystem management. USDA Forest Service Research Publication. http://www.fs.fed.us/research/publications/disturb.htm

Bailey, R.G. 1995. Description of the ecoregions of the United States, second edition. USDA, Forest Service Misc. Publ. 1391. United States Printing Office, Washington D.C.

Bailey, R.G. 1997. Map: Ecoregions of North America. USDA Forest Service in cooperation with The Nature Conservancy and the U.S. Geological Survey, Washington D.C.

Baker, F.S. 1925. Aspen in the central Rocky Mountain region. USDA, Department Bulletin 1291.

Baker, H.G. 1978. Invasion and replacement in Californian and neotropical grasslands. Pages 368-384 *in* J.R. Wilson, editor, Plant relations in pastures. CSIRO, East Melbourne, Australia.

Baker, W.L. 1989. Macro- and micro-scale influences on riparian vegetation in western Colorado. Annals of Association of American Geographers. 79:65-78.

Baker, W.L. 1990. Species richness of Colorado riparian vegetation. Journal of Vegetation Science 1:119-124.

Baker, W.L. 1992. Structure, disturbance, and change in the bristlecone pine forests of Colorado, USA. Arctic and Alpine Research 24:17-26.

Baker, W.L. 2002. Indians and fire in the U.S. Rocky Mountains: the wilderness hypothesis renewed. Pages 41-76 in T.R. Vale (ed.), Western wildnersess: Fire, native peoples, and the natural landscape. Island Press. Washington, D.C.

Baker, W.L. and G.K. Dillon. 2000. Plant and vegetation responses to edges in the Southern Rocky Mountains. Pages 221-246 *in* R.L. Knight, F.W. Smith, S.W. Buskirk, W.H. Romme and W.L. Baker, editors, Forest fragmentation in the Southern Rocky Mountains. Colorado University Press, Boulder, Colorado.

Baker, W.L., P.H. Flaherty, J.D. Lindemann, T.T. Veblen, K.S. Eisenhart, and D. Kulakowski. 2002. Effect of vegetation on the impact of a severe blowdown in the Southern Rocky Mountains, USA. Forest Ecology and Management.

Baker, W. L. 2003. Fires and climate in forested landscape in the U.S. Rocky Mountains. Pages 120-157 in T.T. Veblen, W.L. Baker, G. Montenegro, and T.W. Swetnam, editors. Fire and climatic change in temperate ecosystems of the western Americas. Springer, New York.

Baker, W.L. and D. Ehle. 2001. Uncertainty in surface-fire history: The case of ponderosa pine forests in the Western United States. Canadian Journal of Forest Research 31:1205-1226.

Baker, W.L., P.H. Flaherty, J.D. Lindemann, T.T. Veblen, K.S. Eisenhart, and D. Kulakowski. 2001. Effect of vegetation on the impact of a severe blowdown in the Southern Rocky Mountains, USA. Forest Ecology and Management 154: 1-13.

Baker, W.L., P.H. Flaherty, J.D. Lindemann, T.T. Veblen, K.S. Eisenhart, and D. Kulakowski. Effect of vegetation on the impact of a severe blowdown in the Southern Rocky Mountains, USA. Forest Ecology and Management.

Baker, W.L. and R.L. Knight. 2000. Roads and forest fragmentation in the Southern Rocky Mountains. Pages 97-122 *in* R.L. Knight, F.W. Smith, S.W. Buskirk, W.H. Romme and W.L. Baker, editors, Forest fragmentation in the Southern Rocky Mountains. Colorado University Press, Boulder, Colorado.

Baker, W.L., J.A. Munroe and A.E. Hessl. 1997. The effects of elk on aspen in the winter range in Rocky Mountain National Park. Ecography 20:155-165.

Baker, W.L. and T.T. Veblen 1990. Spruce beetles and fires in the nineteenth century subalpine forests of western Colorado. Arctic and Alpine Research 22:65-80.

Barrett, S.W. 1980a. Indian fires in the pre-settlement forests of western Montana. Pages 35-41 *in* M.A. Stokes and J.H. Dieterich, editors, Proceedings of the Fire History Workshop. USDA Forest Service, General Technical Report RM-81, Fort Collins, Colorado.

Barrett, S.W. 1980b. Indians and fire. Western Wildlands 6:17-21.

Barrett, S.W. and S.F.Arno. 1999. Indian fires in the northern Rockies: Ethnohistory and ecology. Pages 50-64 in R. Boyd, editor, Indians, fire, and the land in the Pacific Northwest. Oregon State University Press, Portland, Oregon.

Barry, R.G. 1992. Mountain weather and climate. Routledge, New York, New York.

Barry, R.G., G. Kiladis and R.S. Bradley. 1981. Synoptic climatology of the western United States in relation to climatic fluctuations during the twentieth century. Journal of Climatology 1:97-113.

Bartlett-Howe, E.J. 2001. Natural disturbance history of a subalpine watershed in the Park Range, Colorado. Masters Thesis, University of Wyoming, Laramie.

Bartlett, R.A. 1962. Great Surveys of the American West. University of Oklahoma Press, Norman, Oklahoma.

Beauvais, G.P. 2000. Mammalian responses to forest fragmentation in the Central and Southern Rocky Mountains. Pages 177-200 *in* R.L. Knight, F.W. Smith, S.W. Buskirk, W.H. Romme and W.L. Baker, editors, Forest fragmentation in the Southern Rocky Mountains. Colorado University Press, Boulder, Colorado.

Bebi, P., D. Kulakowski, and T.T. Veblen. 2003. Interactions between fire and spruce beetle in a subalpine Rocky Mountain forest landscape. Ecology 84:362-371.

Benedict, J.B. 1973. Chronology of cirque glaciation, Colorado Front Range. Quaternary Research 3:584-599.

Benedict, J.B. 1975. The Albion Boardinghouse site: archaic occupation of a high mountain valley. Southwestern Lore 41:1-12.

Bennetts, R.E., G.C. White, F.G. Hawksworth, and S.E. Severs. 1996. The influence of dwarf mistletoe on bird communities in Colorado ponderosa pine forests. Ecological Applications 6:899-909.

Bentz, B.J., J.A. Logan, and G.D. Amman. 1991. Temperature-dependent development of the mountain pine beetle (Coleoptera, Scolytidae) and simulation of its phenology. Canadian Entomologist 123: 1083-1094.

Bentz, B.J., J.A. Logan, and J.C. Vandygriff. 2001. Latitudinal variation in *Dendroctonus ponderosae* (Coleoptera: Scolytidae) development time and adult size. Canadian Entomologist 133: 375-387.

Betancourt, J. L. 1987. Paleoecology of pinyon-juniper woodlands: summary. In: Everett, R. L., ed. Proceedings, Pinyon-juniper conference: 1986 January 13-16; Reno, NV. Gen. Tech. Report. INT-215. Ogden, UT: USDA Forest Service, Intermountain Research Station: 129-139.

Betancourt, J.L. 1990. Late quaternary biogeography of the Colorado Plateau. *in* Betancourt, J.L., T.R. Van Devender, and P.S. Martin, editors, Packrat middens: the last 40,000 years of biotic change. University of Arizona Press, Tucson, Arizona.

Betancourt, J.L., E.A. Pierson, K. Aasen-Rylander, J.A. Fairchild-Parks, & J.S. Dean. 1993. Influence of history and climate on New Mexico pinyon-juniper woodlands. Pages 42-62 *in* E.F. Aldon and D.W. Shaw, editors, Proceedings: managing pinyon-juniper ecosystems for sustainability and social needs. USDA, Forest Service General Technical Report RM-236.

Betancourt, J.L., W.S. Schuster, J.B Mitton, and R.S. Anderson. 1991. Fossil and genetic history of a pinyon pine (*Pinus edulis*) isolate. Ecology 72:1685-1697.

Betancourt, J.L., T.R. Van Devender, and P.S. Martin. 1990. Packrat middens: the last 40,000 years of biotic change. University of Arizona Press, Tucson, Arizona.

Black, R.C. 1969. Island in the Rockies. Country Printer, Granby, Colorado.

Botkin, D.B. 1990. Discordant harmonies: a new ecology for the twenty-first century. Oxford University Press. New York, New York.

Bowman, W.D., D.M. Cairns, J.S. Baron, and T.R. Seastedt. 2002. Islands in the sky: Alpine and Treeline Ecosystems of the Rockies. Pages 183-202 in: J.S. Baron, editor, Rocky Mountain futures: An ecological perspective. Island Press, Washington, D.C.

Bradley, R.S. 1976. Precipitation History of the Rocky Mountain States. Westview Press. Boulder, Colorado.

Brown, H. E. 1958. Gambel oak in west central Colorado. Ecology. 39: 317-327.

Brown, J.K. 1995. Fire regimes and their relevance to ecosystem management. Proc. Soc. Am. For. , pages 171-178.

Brown, P.M., and A.T. Carpenter. 2001. Historical Range of Variability Assessment for Caribou Ranch Open Space, Boulder County, Colorado. Phase 2: Fire History and Stand Structure in Ponderosa Pine Woodlands at Caribou Ranch. Unpublished report, Boulder County Open Space, Colorado.

Brown, P.M., D.R. D'Amico, A.T. Carpenter, and D.M. Andrews. 2001. Restoration of montane ponderosa pine forests in the Colorado Front Range: A Forest Ecosystem Management Plan for the City of Boulder. Ecological Restoration 19:19-26.

Brown, P.M., M.R. Kaufmann, and W.D. Shepperd. 1999. Long-term, landscape patterns of past fire events in a montane ponderosa pine forest of central Colorado. Landscape Ecology 14:513-532.

Brown, P.M., M.G. Ryan, and T.G. Andrews. 2000. Historical fire frequency in ponderosa pine stands in research natural areas, Central Rocky Mountains and Black Hills, US. Natural Areas Journal 20:133-139.

Brown, P.M. and W.D. Shepperd. 2001. Fire history and fire climatology along a 5^{o} latitude gradient in Colorado and Wyoming, USA. Paleobotanist 50: 133-140.

Brown, P.M, and C.H. Sieg. 1999. Historical variability in fire at the ponderosa pine - northern Great Plains prairie ecotone, southeastern Black Hills, South Dakota. Écoscience 6: 539-547.

Brubaker, L.B. 1986. Responses of tree populations to climatic change. Vegetatio 67:119-130.

Buchholtz, C.W. 1983. Rocky Mountain National Park: A history. Colorado Associated University Press, Boulder, Colorado.

Buechling, A. and W.L. Baker. 2004. A fire history from tree rings in a high-elevation forest of Rocky Mountain National Park. Can. J. For Res.34: 1259-1273.

Butler, D.R., G.P. Malanson, and S.J. Walsh. 1992. Snow-avalanche paths: conduits from the periglacial-alpine to the subalpine-depositional zone. *in* A. Abrahams and J. Dixon, editors, Periglacial geomorphology. John Wiley Publishers, London.

Calhoun, R.E. 1956. The age of exploration: 1806-1858. Pages 1-7 *in* H.L. Carter, editor, The Pikes Peak region: a sesquicentennial history. Historical Society of the Pikes Peak Region, Colorado Springs, Colorado.

Carelton, A.M., D.A. Carpenter and P.J. Weser. 1990. Mechanisms of interannual variability of the southwest United States summer rainfall maximum. Journal of Climate 3:999-1015.

Carrara, P.E. 1979. The determination of snow avalanche frequency through tree-ring analysis and historical records at Ophir, Colorado. Geological Society of America Bulletin 90: 773-780.

Casells, E.S. 1997. The archeology of Colorado. Revised Edition. Johnson Books, Boulder, Colorado.

Cates, R.H. and H. Alexander. 1982. Host resistance and susceptibility. Pages 212-263 *in* J.B. Mitton and K.B. Sturgeon, editors, Bark beetles in North American conifers. University of Texas Press, Austin, Texas.

Cayan, D.R. 1996. Interannual climate variability and snowpack in the western United States. Journal of Climate 9:928-948.

Changnon, D., T.B. McKee, and N.J. Doesken. 1990. Hydroclimatic variability in the Rocky Mountain region. Department of Atmospheric Science, Colorado State University, Climatology Report No. 90-3.

Chase, T.N., R.A. Pielke, Sr., T.G.F. Kittel, J.S. Baron, and T.J. Stohlgren. 1999. Impacts on Colorado Rocky Mountain weather and climate due to land use changes on the adjacent Great Plains. Journal of Geophysical Research 104:16,673-16,690.

Christensen, N.L., A.M. Bartuska, J.H. Brown, S. Carpenter, C. D'Antonio, R. Francis, J.F. Franklin, J.A. MacMahon, R.F. Noss, D.J. Parsons, C.H. Peterson, M.G. Turner, and R.G. Woodmansee. 1996. The report of the Ecological Society of America committee on the scientific basis for ecosystem management. Ecological Applications 6:665-691.

Chronic, H. 1998. Roadside Geology of Colorado. Mountain Press Publishing Company, Missoula, Montana.

Chronic, J. and H. Chronic. 1972. Prairie, peak, and plateau: a guide to the geology of Colorado. Colorado Geological Survey, Denver, Colorado.

City of Boulder. 1999. Forest ecosystem management plan, Open Space and Mountain Parks. City of Boulder, Colorado.

Clagg, H.B. 1975. Fire ecology in high-elevation forests in Colorado. Masters Thesis, Colorado State University, Fort Collins, Colorado.

Clements, G.E. 1910. The life history of lodgepole burn forests. USDA Forest Service, Bulletin 79:7-56.

Cole, K.L. 1985. Past rates of change, species richness, and a model of vegetational inertia in the Grand Canyon, Arizona. American Naturalist 125:289-303.

Committee of Scientists. 1999. Sustaining the people's lands: recommendations for stewardship of the National Forests and Grasslands into the next century. USDA, Washington, D.C., http://www.fs.fed.us/news/science/.

Cook, E.R., D.M. Meko, D.W. Stahle, and M.K. 1998. Reconstruction of past drought across the coterminous United States from a network of climatically-sensitive tree-ring data. Available online at http://www.ngdc.noaa.gov/paleo/drought.html.

Cook, J.E. 1996. Implications of modern successional theory for habitat typing: A review. Forest Science 42: 67-75.

Cooke, W.B. 1955. Subalpine fungi and snowbanks. Ecology 36:124-130.

Cooper, C.F. 1960. Changes in vegetation, structure, and growth of southwestern pine forests since white settlement. Ecological Monographs 30:129-164.

Couzin, J. 1999. Landscape changes make regional climate run hot and cold. Science 283: 317-319.

Covington, W.W. 2000. Helping western forests heal. Nature 408: 135-136.

Covington, W.W., and M.M. Moore. 1994. Southwestern ponderosa forest structure: changes since Euro-American settlement. Journal of Forestry 92:39-47.

Crossen, F. 1978. The Switzerland Trail of America. Pruett Press, Boulder, Colorado.

Crossley, J.W. 1996. Managing ecosystems for integrity: Theoretical considerations for resource and environmental managers. Society and Natural Resources: 465-481.

Crowley, T.J. 2000. Causes of climate change over the past 1000 years. Science 289: 270-277.

Crum, S. 1996. People of the Red Earth. Ancient City Press, Santa Fe, New Mexico.

Cryer, D.H. and J.E. Murray. 1992. Aspen regeneration and soils. Rangelands 14: 223-236.

Dahms, C.W.,and B.W. Geils (technical editors). 1997. An assessment of forest ecosystem health in the Southwest. USDA Forest Service, General Technical Report RM-295.

Debyle, N.V. 1985. Wildlife. Pages 135-152 *in* N.V. DeByle and R.P. Winokur, editors, Aspen: ecology and management in the western United States. USDA Forest Service, General Technical Report RM-119.

DeLeo, G.A. and S. Levin. 1997. The multifaceted aspects of ecosystem integrity. Conservation Ecology 1: 1-3.

Denevan, W.M. 1992. The native population of the Americas in 1492. 2nd edition. University of Wisconsin Press, Madison, Wisconsin.

Diaz, H.F. and G.N. Kiladis. 1992. Atmospheric teleconnections associated with the extreme phases of the Southern Oscillation. Pages 7-28 *in* H.F. Diaz and V. Markgraf, editors, El Niño: historical and paleoclimatic aspects of the Southern Oscillation. Cambridge University Press, Cambridge.

Donnegan, J. 1999. Climatic and human influences on fire regimes in Pike National Forest. Ph.D. Dissertation, University of Colorado, Boulder, Colorado.

Donnegan, J. and A.J. Rebertus. 1999. Rates of succession and mechanisms of subalpine forest succession along an environmental gradient. Ecology 80: 1370-1384.

Donnegan, J., T.T. Veblen, and J.S. Sibold. 2001. Climatic and human influences on fire history in Pike National Forest, central Colorado. Canadian Journal of Forest Research 31: 1526-1539

Doyle K.M., K.H. Knight, D. L. Taylor, W.J. Barmore, and J.M. Benedict. 1998. Seventeen years of forest succesion following the Waterfalls Canyon Fire in Grand Teton National Park, Wyoming. International Journal of Wildland Fire 8: 45-55.

Dull, R.A. 1999. Palynological evidence for 19th century grazing-induced vegetation change in the southern Sierra Nevada, California, USA. Journal of Biogeography 26: 899-912.

Dunwiddie, P.W. 1977. Recent tree invasion of subalpine meadows in the Wind River Mountains, Wyoming. Arctic and Alpine Research 9:393-399.

Ehle, D.S. 2001. Spatial and temporal patterns of disturbance and ponderosa pine forest structure in Rocky Mountain National Park. Masters Thesis, University of Wyoming, Laramie, Wyoming.

Ehle, D.S., and W.L. Baker. 2003. Disturbance and stand dynamics in ponderosa pine forests in Rocky Mountain National Park, USA. Ecological Monographs 73 (4): 543-566.

Eisenhart, K.S. 1999. Dendrochronological identification of spruce bark beetle outbreaks in northwestern Colorado. Masters Thesis, University of Colorado, Boulder, Colorado.

Eisenhart, K. and T. T. Veblen. 2000. Dendrochronological identification of spruce bark beetle outbreaks in northwestern Colorado. Canadian Journal of Forest Research 30: 1788-98.

Eisenhart, K. 2004. Pinyon-juniper woodlands through thick and thin: Natural variability and woodland stand development on the Uncomphagre Plateau in western Colorado. Ph.D. thesis, University of Colorado, Boulder, Colorado.

Elias, S. 1996. Late Pelistocene and Holocene seasonal temperatures reconstructed from fossil beetle assemblages in the Rocky Mountains. Quaternary Research 46:311-318.

Evans, R.A. 1988. Management of pinyon-juniper woodlands. USDA Forest Service, General Technical Report INT-249.

Fall, P.L. 1997. Fire history and composition of the subalpine forest of western Colorado during the Holocene. Journal of Biogeography 24:309-325.

Fall, P.L. 1988. Vegetation dynamics in the southern Rocky Mountains: Late Pleistocene and Holocene timberline fluctuations. Ph.D. Dissertation, University of Arizona, Tucson, AZ.

Feeney, S.R., T.E. Kolb, W.W. Covington, and M.R. Wagner. 1998. Influence of thinning and burning restoration treatments on presettlement ponderosa pines at the Gus Pearson Natural Area. Canadian Journal of Forest Research 28:1295-1306.

Feiler, E. 1994. A 6000-year record of vegetation change from West Carrizo Canyon, southeastern Colorado. Masters Thesis, Northern Arizona University, Flagstaff, Arizona.

Feiler, E.J. and R.S. Anderson. 1993. The paleoecology of Dome Creek Meadow, Bear River Corridor, Garfield County, Colorado. Unpublished report to the USDA Forest Service, Steamboat Springs, Colorado.

Ferrell, M.H. 1981. Colorado & Southern Narrow Gauge. Pruett Publishing Company, Boulder, Colorado.

Finney, M. A., R. Bartlette, L. Bradshaw, K. Close, B. M. Collins, P. Gleason, W. M. Hao, P. Langowski, J. McGinely, C. W. McHugh, E. Martinson, P. N. Omi, W. Shepperd, and K. Zeller. 2003. Fire behavior, fuel treatments, and fire suppression on the Hayman Fire. Pages 59-96 in R. T. Graham, editor. Hayman Fire Case Study Analysis. U.S. Department of Agriculture, Forest Service, Rocky MountainResearch Station, Ogden, UT.

Flaherty, P.H. 2000. Effects of pre-disturbance forest vegetation on the impacts of a catastrophic blowdown in a Rocky Mountain subalpine landscape. Masters Thesis, University of Wyoming. Laramie, Wyoming.

Floyd, M. L., W. L. Romme, and D. D. Hanna. 2000. Fire history and vegetation pattern in the Mesa Verde National Park, Colorado, USA. Ecological Applications 10: 1666-1680.

Floyd, L., M. Colyer, D.D. Hanna, and W.H. Hanna. 2003. Gnarly old trees: Canopy characteristics of old-growth pinyon-juniper woodland. Pages 11-30 in L. Floyd, editor, Ancient pinyon-juniper woodlands: A natural history of Mesa Verde country. Colorado University Press, Boulder, Colorado.

Fossett, F. 1880. Colorado—Its gold and silver mines, farms and stock ranges, and health and pleasure resorts. C.G. Crawford, New York, New York.

Foster Wheeler Environmental Corporation. 1999. Landscape assessment: Upper South Platte Watershed. Report to the USDA Forest Service, Colorado State Forest Service, Denver Water Board and U.S. Environmental Protection Agency, http://www.fs.fed.us/r2/psicc/spl/south_ platte_web/index.htm.

Fritts, H.C. 1976. Tree rings and climate. Academic Press, London.

Fritts, H.C. and T.W. Swetnam. 1989. Dendroecology: a tool for evaluating variations in past and present forest environments. Advances in Ecological Research 19:111-188.

Fritz, P.S. 1933. Mining districts of Boulder County, Colorado. Ph.D. thesis, University of Colorado, Boulder, Colorado.

Frye, R.H., and T.D. Landis. 1975. Mountain pine beetle and dwarf mistletoe, Lake Creek Area, San Carlos Ranger District, Pike-San Isabel National Forest. USDA Forest Service, Biological Evaluation Region 2, R2-75-4.

Frye, R.H., Flake, H.W. and Germain, C.J. 1974. Spruce beetle winter mortality resulting from record low temperatures in Arizona. Environmental Entomologist, 3, 752-754.

Fujita, T.T. 1989. The Teton-Yellowstone tornado of 21 July 1987. Monthly Weather Review 117:1913-1940.

Gilbert, P.F., O.C. Wallmo, and R.B. Gill. 1970. Effects of snow depth on mule deer in Middle Park, Colorado. Journal of Wildlife Management 34:15-23.

Glenn-Lewin, D.C., R.K. Peet and T.T. Veblen, editors. 1992. Plant succession: theory and prediction. Chapman and Hall, London.

Goldblum, D. and T.T. Veblen. 1992. Fire history of a ponderosa pine/Douglas-fir forest in the Colorado Front Range. Physical Geography 13:133-148.

Gottfried, G.J., T.W. Swetnam, C.D. Allen, J.L. Betancourt, and A.L. Chung-MacCoubrey. 1995. Pinyon-Juniper Woodlands. Pages 95-132 *in* D.M. Finch and J.A. Tainter, technical editors, Ecology, diversity, and sustainability of the Middle Rio Grande Basin. USDA Forest Service, General Technical Report RM-268.

Gray, S.T., J.L. Betancourt, C.L. Fastie, and S.T. Jackson. 2003. Patterns and sources of multidecadal osciallations in drought-sensitive tree-ring records from the central and southern Rocky Mountains. Geophysical Research Letters 30:1-4.

Gray, S.T., L.J. Graumlich, J.L. Betancourt, and G.T. Pederson. 2004. A tree-ring based reconstruction of the Atlantic Multidecadal Oscillation since 1567 A.D. Geophysical Research Letters 31:1-4.

Graybill, D.A. 1989. Analysis of growth trends and variation in conifers from the Colorado Front Range. A final report submitted to Western Conifers Research Cooperative, Forest Response Program, Environmental Protection Agency.

Greenland, D., J. Burbank, J. Key, L. Klinger, J. Moorhouse, S. Oaks and D. Shankman. 1985. The bioclimates of the Colorado Front Range. Mountain Research and Development 5:251-262.

Grimm, M.M., E.E. Wohl, and R.D. Jarrett. 1995. Coarse-sediment distribution as evidence of an elevation limit for flash flooding, Bear Creek, Colorado. Geomorphology 14:199-210.

Grissino-Mayer, H.D. 1995. Tree-ring reconstructions of climate and fire history at El Malpaís National Monument, New Mexico. Ph.D. Dissertation, University of Arizona, Tucson, Arizona.

Grissino-Mayer, H.D., W. H. Romme, M. L. Floyd, and D. D. Hanna. 2004. Climatic and human influences on fire regimes of the southern San Juan Mountains, Colorado, USA. Ecology 85: 1708-1724.

Grove, J.M. 1988. The Little Ice Age. Methuen & Co. Ltd., New York, New York.

Gruell, G.E. 1985. Indian fires in the interior west: a widespread influence. USDA Forest Service, General Technical Report INT-182.

Hadley, K.S. 1994. The role of disturbance, topography, and forest structure in the development of a montane forest landscape. Bulletin of the Torrey Botanical Club 121:47-61.

Hadley, K.S. and T.T. Veblen. 1993. Stand response to western spruce budworm and Douglas-fir bark beetle outbreaks, Colorado Front Range. Canadian Journal of Forest Research 23:479-491.

Hafen, L.R. and A.W. Hafen. 1956. Rufus B. Sage, his letters and papers, 1836-1847. Volume 2. The Arthur H. Clark Company, Glendale, California.

Harbour, M. 1982. The Tarryall Mountains and the Puma Hills: A History. Century One Press, Colorado Springs, Colorado.

Hansen, E. M., B.J. Bentz, and D.L. Turner. 2001. Temperature-based model for predicting univoltine brood proportions in spruce beetle (Coleoptera: Scolytidae). The Canadian Entomologist 133: 827-841.

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P.Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaemper, K. Cromack, Jr., and K.W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Pages 133-202 *in* A. MacFadyen and E.D. Ford, editors, Advances in Ecological Research, Volume 15. Academic Press, London.

Harper, K. T., F. J. Wagstaff, and L. M. Kunzler. 1985. Biology and management of the Gambel oak vegetative type: a literature review. USDA Forest Service. General Technical Report INT-179. Ogden, UT.

Hawksworth, F.G., 1961. Dwarf mistletoe of ponderosa pine in the Southwest. USDA Forest Service Technical Bulletin 1246.

Hebertson, E.G. 2004. Snow avalanche disturbance in intermountain spruce-fir forests and implications for the spruce bark beetle (Coleoptera: Scolytidae). Ph.D. thesis, Utah State University, Logan, Utah.

Hessl, A.E. and W.L. Baker. 1997a. Spruce and fir regeneration and climate in the forest-tundra ecotone of Rocky Mountain National Park, Colorado, USA. Arctic and Alpine Research 29:173-183.

Hessl, A.E. and W.L. Baker. 1997b. Spruce-fir growth form changes in the forest-tundra ecotone of Rocky Mountain National Park, Colorado, USA. Ecography 20: 356-367.

Higgins, K.F. 1986. Interpretation and compendium of historical fire accounts in the Northern Great Plains. USDI Fish and Wildlife Service, Resource Publication 161, Washington, D.C.

Higgins, R.W., K.C. Mo and Y. Yao. 1998. Interannual variability of the U.S. summer precipitation regime with emphasis on the southwestern monsoon. Journal of Climate 11:2582-2606.

Hinds, T.E. 1976. Aspen mortality in Rocky Mountain campgrounds Forest Service, Research Paper RM-164.

Hinds, T.E. 1985. Diseases. Pages 87-114 *in* N.V. DeByle and R.P. Winokur, editors, Aspen: ecology and management in the western United States. USDA Forest Service, General Technical Report RM-119.

Hinds, T.E., F.G. Hawksworth, and R.W. Davidson. 1965. Beetle-killed Engelmann spruce: its deterioration in Colorado. Journal of Forestry 63:536-542.

Holling, C.S. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. Ecological Monographs 62:447-502.

Holling, C.S., and G.K. Meffe. 1996. Command and control and the pathology of natural resource management. Conservation Biology 10:328-337.

Holway, J.G., and R.T. Ward. 1963. Snow and meltwater effects in an area of Colorado alpine. American Midland Naturalist 69:189-197.

Hopkins, A.D. 1909. Practical information on the Scolytid beetles of North American forests. I. Bark beetles of the Genus *Dendroctonus*. USDA Bureau of Entomology Bulletin 83-1, United States Government Printing Office, Washington, D.C.

Howe, E. and W.L. Baker. 2003. Landscape heterogeneity and disturbance interactions in a subalpine watershed in northern Colorado, USA. Annals of the Association of American Geographers 93: 797-813.

Huckaby, L.S., M.R. Kaufmann, J.M. Stoker, and P.J. Fornwalt. 2001. Landscape patterns of montane forest age structure relative to fire history at Cheesman Lake in the Colorado Front Range. Pages 19-27 in R.K. Vance, W.W. Covington, and C.D. Edminster (tech. coords), Ponderosa pine ecosystems restoration and conservation: steps toward stewardship. U.S.D.A. Forest Service Proceedings RMRS-P-22.

Huenneke, L. 1988. SCOPE program on biological invasions: a status report. Conservation Biology 2:8-10.

Huggard, D.J. and A. Arsenault. 1999. Comment—Reverse cumulative standing age distributions in fire-frequency analysis. Canadian Journal for Forest Research 2:1449-1456.

Hughes, J.D. 1987. American Indians in Colorado. Pruett Publishing Company, Boulder, Colorado.

Hughes, M.K. and G. Funkhouser. 1998. Extremes of moisture availability reconstructed from tree rings for recent millenia in the Great Basin of Western North America. Pages 99-108 *in* M. Beniston and J.L. Innes, eds. The Impacts of Climate Variability on Forests. Springer, Berlin.

Hutto, R.L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountain (U.S.A.) conifer forests. Conservation Biology 9: 1041-1058.

Illg, C. and G. Illg. 1997. Forest fire, water quality, and the incident at Buffalo Creek. American Forests (Winter/Spring):33-35.

Ingwall, H. 1923. History: Pike National Forest. USDA Forest Service, Historic Report of the Recreation Assistant, Pueblo, Colorado.

Intergovernmental Panel on Climate Change. 2000. http://www.ipcc.ch

Ives, J.D., A.I. Mears, P.E. Carrara and M.J. Bovis. 1976. Natural hazards in mountain Colorado. Annals of the Association of American Geographers 66:129-144.

Jack, J.G. 1900. Pikes Peak, Plum Creek, and South Platte Reserves. Twentieth annual report of the United States Geological Survey to the Secretary of the Interior, 1898-1899. United States Government Printing Office, Washington, D.C.

James, R.L., and D.J. Goheen. 1980. Distribution and characteristics of conifer root diseases on the San Isabel, Rio grande, San Juan, and Grand Mesa National Forests in Colorado. USDA Forest Service Region 2, Biological Evaluation R2-80-4.

Jarvis, P.J. 1979. The ecology of plant and animal introductions. Progress in Physical Geography 6:187-214.

Johnsen, T.N. 1962. One-seed juniper invasion of northern Arizona grasslands. Ecological Monographs 32: 187-207.

Johnson, E. A. and S. L. Gutsell. 1994. Fire frequency models, methods and interpretations. Advances in Ecological Research 25:239-287.

Johnson, D.W. 1999. Service trip report LSC-99-07, white pine blister rust survey. Letter to Forest Supervisor, Arapahoe and Roosevelt National Forests, July 21, 1999.

Johnson, S.J. (editor) 1997. Forest insect and disease conditions in the Rocky Mountain Region 1996. USDA Forest Service, Rocky Mountain Region, Renewable Resources, Forest Health Management, Lakewood, Colorado.

Jones, J. R. and N. V. DeByle (1985). Soils. Aspen ecology and management in the western United States. Pages 19-24 in: N.V. DeByle and R.P. Winokur, editors. Fort Collins, CO, USDA Forest Service Rocky Mountain Forest and Range Experimental Station. GTR RM-119.

Jones, J.R. (1985). Distribution. Aspen ecology and management in the western United States. Pages 9-10 in: N.V. DeByle and R.P. Winokur, editors. Fort Collins, CO, USDA Forest Service Rocky Mountain Forest and Range experimental Station. GTR RM-119.

Kashian, D., W. Romme, and C. Regan. 2004. Variable trajectories of aspen succession and decline in the northern Front Range. Abstract, Annual Meeting of the Ecological Society of America, Portland, Oregon.

Kaufmann, M.R., P.J. Fornwalt, L.S., Huckaby, and J.M. Stoker. 2001. Cheesman Lake – a historical ponderosa pine landscape guiding restoration in the South Platte watershed of the Colorado Front Range. Pages 9-18 in R.K. Vance, W.W. Covington, and C.D. Edminster (tech. coords), Ponderosa pine ecosystems restoration and conservation: steps toward stewardship. U.S.D.A. Forest Service Proceedings RMRS-P-22

Kaufmann, M.R., R.T. Graham, D.A. Boyce, Jr., W.H. Moir, L. Perry, R.T. Reynolds, R.L. Bassett, P. Mehlhop, C.B. Edminster, W.M. Block and P.S. Corn. 1994. An ecological basis for ecosystem management. USDA Forest Service, General Technical Report RM-246.

Kaufmann, M.R., L.S. Huckaby, C.M. Regan, and J. Popp. 1998. Forest reference conditions for ecosystem management in the Sacramento Mountains, New Mexico. USDA Forest Service, General Technical Report RMRS-19.

Kaufmann, M.R., C.M. Regan, and P.M. Brown. 2000. Heterogeneity in ponderosa pine/Douglas-fir forests: Age and size structure in unlogged and logged landscapes of central Colorado. Canadian Journal of Forest Research. 30:698-711.

Kay, C.E. 1997. Is aspen doomed? Journal of Forestry 95:4-11.

Kaye, M. W. 2002. Forest ecology of quaking aspen in Rocky Mountain National Park: a landscape survey, a reconstruction, and a predictive model. Ph.D. dissertation. Colorado State University. Fort Collins, CO.

Kemp, D.C. 1960. Silver, gold and black iron: A story of the Grand Island Mining District of Boulder County, Colorado. Sage Swallow, Denver.

Kerr, R.A. 2000. Dueling models: future U.S. climate uncertain. Science 288: 2113.

Kipfmueller, K.F. 1997. A fire history of a subalpine forest in southeastern Wyoming. Masters Thesis, University of Wyoming, Laramie, Wyoming.

Kipfmueller, K.F. and W.L. Baker. 1998a. A comparison of three techniques to date stand-replacing fires in lodgepole pine forests. Forest Ecology and Management 104: 171-177.

Kipfmueller, K.F. and W.L. Baker. 1998b. Fires and dwarf mistletoe in a Rocky Mountain lodgepole pine ecosystem. Forest Ecology and Management. 108: 77-84.

Kipfmueller, K.F. and W.L. Baker. 2000. A fire history of a subalpine forest in southeastern Wyoming, USA. Journal of Biogeography 27:71-85.

Kittel, T. G. F., J.A. Royle, C. Daly, N.A. Rosenbloom, W.P. Gibson, H.H. Fisher, D.S. Schimel, L.M. Berliner, and VEMAP2 Participants. A gridded historical (1895-1993) bioclimate dataset for the conterminous United States. Page 219-222 *in* Proceedings of the 10th Conference on Applied Climatology; 1997 Oct 20-1997 Oct 24; Reno, Nevada. Boston: American Meteorological Society.

Knight, D.H. 1987. Parasites, lightning, and the vegetation mosaic in wilderness landscapes. Pages 59-83 *in* M.G. Turner, editor, Landscape heterogeneity and disturbance. Springer-Verlag, New York.

Knight, D.H. 1994. Mountains and plains: The ecology of Wyoming landscapes. Yale University Press, New Haven, Connecticut.

Knight, D.H. and W.A. Reiners. 2000. Natural patterns in Southern Rocky Mountain landscapes and their relevance to forest management. Pages 15-30 *in* R.L. Knight, F.W. Smith, S.W. Buskirk, W.H. Romme and W.L. Baker, editors, Forest fragmentation in the Southern Rocky Mountains. Colorado University Press, Boulder, Colorado.

Knight, R.L., F.W. Smith, S.W. Buskirk, W.H. Romme and W.L. Baker, editors. 2000. Forest fragmentation in the Southern Rocky Mountains. Colorado University Press, Boulder, Colorado.

Kufeld, R. C. 1983. Response of elk, mule deer, cattle, and vegetation to burning, spraying, and chaining of Gambel oak rangeland. Technical publication No. 34. Colorado Division of Wildlife. Denver, CO.

Kulakowski, D. and T.T. Veblen. 2002. Influences of fire history and topography on the pattern of a severe wind blowdown in a Colorado subalpine forest. Journal of Ecology 90:806-819.

Kulakowski, D., and T.T. Veblen. 2003. Subalpine forest development following a blowdown in the Mount Zirkel Wilderness, Colorado, USA. Journal of Vegetation Science 14: 653-660.

Kulakowski, D., T.T. Veblen, and P. Bebi. 2003. Effects of fire and spruce beetle outbreak legacies on the disturbance regime of a subalpine forest in Colorado. Journal of Biogeography 30: 1445-1456.

Kulakowski, D., T.T. Veblen and S. Drinkwater. 2004. The persistence of quaking aspen (*Populus tremuloides*) in the Grand Mesa area, Colorado. Ecological Applications. In press.

Kulakowski, D., T.T. Veblen, and B. Kurzel. Unpublished manuscript. Influences of infrequent fire, topography, and vegetation legacies on the persistence of quaking aspen in the Flat Tops area, Colorado.

Kurzel, B.P. 2004. Regeneration dynamics of persistent quaking aspen (Populus tremuloides) in western Colorado. Masters Thesis, University of Colorado, Boulder, CO.

Landres, P., P. Morgan, and F. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179-1188.

Langenheim, J. H. 1962. Vegetation and environmental patterns in the Crested Butte Area, Gunnison County, Colorado. Ecological Monographs. 32: 249-285.

Larkin, R.P., P.K. Grogger and G.L. Peters. 1980. The Southern Rocky Mountains. Kendall/Hunt Publishing Company, Dubuque, Iowa.

Larsson, S., R. Oren, R.H. Waring, and J.W. Barrett. 1983. Attacks of mountain pine beetle as related to tree vigor of ponderosa pine. Forest Science 29:395-402.

Lauenroth, W.K., and D.G. Milchunas. 1989. The shortgrass steppe. *in* R.T. Coupland, editor, Natural grasslands. Ecosystems of the World series, Elsevier Press, New York.

Laven, R.D., P.N. Omi, J.G. Wyant and A.S. Pinkerton. 1980. Interpretation of fire scar data from a ponderosa pine ecosystem in the Central Rocky Mountains, Colorado. USDA Forest Service, General Technical Report RM-81.

League, K. 2004. The role of recent climatic variability on episodic Pinus ponderosa recruitment patterns along the forest-grassland ecotone of northern Colorado. Masters Thesis, University of Colorado, Boulder, CO.

Lertzman, K., J. Fall and B. Dorner. 1998. Three kinds of heterogeneity in fire regimes: At the crossroads of fire history and landscape ecology. Northwest Science 72:4-23.

Lessard, G., and B. Raimo. 1986. Douglas-fir tussock moth, *Orgyia pseudotsugata* (McDunnough), Pike National Forest. USDA Forest Service Region 2, Biological Evaluation R2-86-3.

Logan, J.A. and B.J. Bentz. 1999. Model analysis of mountain pine beetle (Coleoptera: Scolytidae) seasonality. Environmental Entomology 28: 924-934.

Logan, J.A., P.V. Bolstad, B.J. Bentz, and D.L. Perkins. 1995. Assessing the effects of changing climate on mountain pine beetle dynamics. Pages 92-105 *in* R.W. Tinus, editor, Interior west global change workshop. USDA Forest Service, General Technical Report RM-262.

Mack, R.N. 1989. Temperate grasslands vulnerable to plant invasions: characteristics and consequences. Pages 155-179 *in* J.A. Drake, H.A. Mooney, F. di Castri, R.H. Groves, F.H. Kruger, M. Rejmanek, and M. Williamson, editors, Biological invasions: a global perspective. John Wiley Publishers, Chichester.

Madany, M.H. and N.E. West. 1983. Livestock grazing-fire regime interactions within montane forests of Zion National park, Utah. Ecology 64: 661-667.

Madole, R.F. 1997. Fire frequency during the Holocene in a small, forested drainage basin in the Southern Rocky Mountains. *in* Abstracts of the All Scientists Conference, Rocky Mountain National Park, June 9-10, 1997.

Malakoff, D. 2002. Arizona ecologist puts stamp on forest restoration debate. Science 297: 2194-2196.

Malanson, G.P., and D.R. Butler. 1990. Woody debris, sediment, and riparian vegetation of a sublalpine river, Montana, U.S.A. Arctic and Alpine Research 22:183-194.

Marr, J.W. 1961. Ecosystems of the east slope of the Front Range in Colorado. University of Colorado Studies Series in Biology 8.

Martin, P.S., and H.E. Wright, Jr. 1967. Pleistocene extinctions: the search for a cause. Yale University Press, New Haven, Connecticut.

Mason, E. 1963. The early history of Manitou Park, Colorado. Unpublished manuscript, Special Collections and Archives, Tutt Library, The Colorado College, Colorado Springs, Colorado.

Massey, C.L. and N.D. Wygant. 1954. Biology and control of the Engelmann spruce beetle in Colorado. USDA, Circular No. 944.

Mast, J.N., T.T. Veblen and Y.B. Linhart. 1998. Disturbance and climatic influences on age structure of ponderosa pine at the pine/grassland ecotone, Colorado Front Range. Journal of Biogeography 25:743-767.

McBride, J.R. 1983. Analysis of tree rings and fire scars to establish fire history. Tree Ring Bulletin 43:51-67.

McCabe, G.J., M.A. Palecki, and J.L. Betancourt. 2004. Pacific and Atlantic Ocean influences on multidecadal drought frequency in the United States. Proceedings National Academy of Science 101: 4136-41.

McConnell, V. 1966. Bayou Salado: The Story of South Park. Sage Books, Denver, Colorado.

McCune, B. 1983. Fire frequency reduced two orders of magnitude in the Bitteroot Canyons, Montana. Canadian Journal of Forest Research 13:212-218.

McIntosh, R.P. 1999. The succession of succession: A lexical chronology. Bulletin of the Ecological Society of America 80:256-265.

McKenzie, C. (2001). Establishment patterns and hight growth rates of aspen and conifer species growing on the Gunnison National Forest in western Colorado. Master thesis. Department of Forest Sciences. Fort Collins, CO, Colorado State University.

McPherson, G.R. 1997. Ecology and management of North American savannas. University of Arizona Press, Tucson, Arizona.

McTighe, J. 1984. Roadside History of Colorado. Johnson Books, Boulder, Colorado.

Meagher, G. S. 1943. Reaction of pinyon and juniper seedlings to artificial shade and watering. J. Forestry. 41: 480-482.

Mitton, J.B. and M.C. Grant. 1996. Genetic variation and the natural history of quaking aspen. Bioscience 46:25-31.

Mladenoff, D.J., M.A. White, J. Pastor, and T.R. Crow. 1993. Comparing spatial pattern in unaltered old-growth and disturbed forest landscapes. Ecological Applications 3:294-306.

Michaelsen, J. and L.G. Thompson. 1992. A comparison of proxy records of El Niño/Southern Oscillation. Pages 323-346 *in* H.F. Diaz and V. Markgraf, editors, El Niño: historical and paleoclimatic aspects of the Southern Oscillation. Cambridge University Press, Cambridge.

Miller, J.R., L.A. Joyce, R.L. Knight, and R.M King. 1996. Forest roads and landscape structure in the southern Rocky Mountains. Landscape Ecology 11:115-127.

Moir, W.H. 1969. The lodgepole pine zone in Colorado. American Midland Naturalist 81:87-98.

Moore, R. 1992. Soil survey of Pike National Forest, Eastern Part, Colorado, parts of Douglas, El Paso, Jefferson, and Teller counties. USDA, Forest Service and Soil Conservation Service in cooperation with Colorado Agricultural Experiment Station, Washington, D.C.

Morgan, P., G.H. Aplet, J.B. Haufler, H.C. Humphries, M.M. Moore and W.D. Wilson. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. Journal of Sustainable Forestry 2:87-111.

Mueggler, W.F. 1985. Vegetation associations. Pages 45-56 *in* N.V. DeByle and R.P. Winokur, editors, Aspen: ecology and management in the western United States. USDA Forest Service, General Technical Report RM-119.

Muir, P.S. and J.E. Lotan. 1985. Disturbance history and serotiny of *Pinus contorta* in western Montana. Ecology 66: 1658-1668.

Murphy, P.H. 1982. The forest vegetation of the Lost Creek Area in the Southern Front Range, Colorado. Masters Thesis, University of Colorado, Boulder, Colorado.

Negron, J.F. 1998. Probability of infestation and extent of mortality associated with the Douglas-fir beetle in the Colorado Front Range. Forest Ecology and Management 107:71-85.

Neilson, R. P. and L. H. Wullstein. 1983. Biogeography of two southwest American oaks in relation to atmospheric dynamics. Journal of Biogeography. 10: 275-297.

Ogle, K. 2001. Drought-induced Pinyon Mortality: The Role of Environmental Stress, Tree Age, and Recent Growth. Workshop on Biocomplexity in Pinyon-Juniper Woodlands - January 29-31, Flagstaff, Arizona.

Oliver, C. D. and B.C. Larson. 1990. Forest stand dynamics. McGraw Hill, New York, New York.

Olsen, W.K., J.M Scmid, and S.A. Mata. 1996. Stand characteristics associated with mountain pine beetle infestations in ponderosa pine. Forest Science 42:310-327.

Paine, R.T. and S.A. Levin. 1981. Intertidal landscapes: disturbance and the dynamics of pattern. Ecological Monographs 51:145-178.

Paine, T.D., K.F. Raffa, and T.C. Harrington. 1997. Interactions among scolytid bark beetles and live host conifers. Annual Reviews Entomology 42: 179-206.

Parendes, L.A. and J.A. Jones. 2000. Role of light availability and dispersal in exotic plant invasion along roads and streams in the H.J. Andrews Experimental Forest, Oregon. Conservation Biology 14:64-75.

Parker, A.J. and K.C. Parker. 1994. Structural variability of mature lodgepole pine stands on gently sloping terrain in Taylor Park Basin, Colorado. Canadian Journal of Forest Research 24:2020-2029.

Pearson, G.A. 1934. Grass, pine seedlings and grazing. Journal of Forestry 32:545-555.

Pedersen, B.S. 1998. The role of stress in the mortality of midwestern oaks as indicated by growth prior to death. Ecology 78:79-93.

Peet, R.K. 1981. Forest vegetation of the Colorado Front Range. Vegetatio 43:131-140.

Peet, R.K. 2000. Forests of the Rocky Mountains. Pages 75-122 *in* M. G. Barbour and W.D. Billings, editors, North American terrestrial vegetation. 2nd edition. Cambridge University Press, New York, New York.

Pepin, N. 2000. Twentieth-century change in the climate record for the Front Range, Colorado, U.S.A. Arcitic, Antarctic, and Alpine Research 32:135-146.

Petersen, K.L. 1994. A warm and wet Little Climatic Optimum and a cold and dry Little Ice Age in the southern Rocky Mountains, U.S.A. Climatic Change 26:243-269.

Pickett, S.T.A., S.L. Collins and J.J. Armesto. 1987. Models, mechanisms and pathways of succession. Botanical Review 53:335-371.

Pickett, S.T.A, and P.S. White, (editors). 1985. The ecology of natural disturbance and patch dynamics. Academic Press, New York, New York.

Plummer, F.G. 1912. Forest fires: their causes, extent and effects, with a summary of recorded destruction and loss. USDA Forest Service, Bulletin 117.

Porter, C. and M.R. Porter. 1950. Ruxton of the Rockies. University of Oklahoma Press, Norman, OK.

Powell, D.C. 1987. Field guide to forest plants of South-Central Colorado. USDA Forest Service, Rocky Mountain Region, No. R2-ECOL-87-01 Pueblo, Colorado,.

Prentice, I.C. 1988. Records of vegetation in time and space: the principles of pollen analysis. Pages 17-42 *in* B. Huntley and T. Webb III, editors, Vegetation history. Junk, Dordrecht.

Pruess, J., E.E. Wohl, and R.D. Jarrett. 1998. Methodology and implications of maximum paleodischarge estimates for mountain channels, upper Animas River basin, Colorado. USA. Arctic and Alpine Research 30:40-50.

Pyne, S.J. 1982. Fire in America: A cultural history of wildland and rural fire. Princeton University Press, Princeton, New Jersey.

Reap, R.M. 1986. Evaluation of cloud-to-ground lightning data from the Western United States for the 1983-1984 summer seasons. Journal of Climate and Applied Meteorology 12:85-92.

Rebertus, A.J., B.R. Burns, and T.T. Veblen. 1991. Stand dynamics of *Pinus flexilis*-dominated subalpine forests in the Colorado Front Range. Journal of Vegetation Science 2:445-458.

Rebertus, A.J. T.T. Veblen, L.M. Roovers, and J.N. Mast. 1992. Structure and dynamics of old-growth Engelmann spruce-subalpine fir in Colorado. Pages 139-153 *in* Proceedings of the conference on old-growth forests in the Rocky Mountains and Southwest, Portal, Arizona, March 9-13, 1992. USDA Forest Service, General Technical Report RM-213.

Reed, R.A., J. Johnson-Barnard, and W.L. Baker. 1996. Contribution of roads to forest fragmentation in the Rocky Mountains. Conservation Biology 10:1098-1106.

Richmond, G.M. 1960. Glaciation of the east slope of Rocky Mountain National Park, Colorado. Geological Society of America Bulletin 71:1371-82.

Roe, A.L. and G.D. Amman. 1970. The mountain pine beetle in lodgepole pine forests. USDA Forest Service, Research Paper INT-71.

Rogers, G.F., H.E. Malde, and R.M. Turner. 1984. Bibliography of repeat photography for evaluating landscape change. University of Utah Press, Salt Lake City, Utah.

Romme, W.H. and D.G. Despain. 1989. Historical perspective on the Yellowstone fires of 1988. BioScience 39:695-699.

Romme, W.H. and D.H. Knight. 1981. Fire frequency and subalpine forest succession along a topographic gradient in Wyoming. Ecology 62:319-326.

Romme, W. H., L. Floyd-Hanna, et al. (2001). Aspen's ecological role in the West. Sustaining aspen in western landscapes: symposium proceedings RMRS-P-18. W. D. Shepperd, D. Binkley, D. L. Bartos, T. J. Stohlgren and L. G. ESkew. Ft.Collins, USDAForest Service Rocky Mountain Forest and Range Experimental Station.

Romme, W.H., M.G. Turner, R.H. Gardner, W.W. Hargrove, G.A. Tuskan, D.G. Despain, and R.A. Renkin. 1997. A rare episode of sexual reproduction in aspen (*Populus tremuloides* Michx.) following the 1988 Yellowstone fires. Natural Areas Journal 17: 17-25.

Romme, W.H., T.T. Veblen, M.R. Kaufmann, R. Sherriff and C.M. Regan. 2003a. Ecological effects of the Hayman Fire Part1: Historical (Pre-1860) and current (1860-2002) fire regimes. Pages 151-163 in: Hayman Fire Case Study Analysis. USDA Forest Service Tech. Rep. RMRS-GTR-114.

Romme, W.H., M.R. Kaufmann, T.T. Veblen, R. Sherriff and C.M. Regan. 2003b. Ecological effects of the Hayman Fire Part 2: Historical (Pre-1860) and current (1860-2002) forest and landscape structure. Pages 164-171 in: Hayman Fire Case Study Analysis. USDA Forest Service Tech. Rep. RMRS-GTR-114.

Rothermel, R.C. 1983. How to predict the spread and intensity of forest and range fires. USDA Forest Service, General Technical Report INT-143.

Rowdabaugh, K.M. 1978. The role of fire in the ponderosa pine-mixed conifer ecosystems. Ph.D. Dissertation, Colorado State University, Fort Collins, Colorado.

Rumbaitis-Del Rio, C. 2004. Compound disturbance in a managed landscape: Ecological effects of catastrophic blowdown, salvage-logging and wildfire in a subalpine forest. Ph.D. thesis, University of Colorado, Boulder, Colorado.

Rummel, R.S. 1951. Some effects of livestock grazing on ponderosa pine forest and range in central Washington. Ecology 32: 594-607.

Sackett, S.S. 1984. Observations on natural regeneration in ponderosa pine following a prescribed fire in Arizona. USDA Forest Service Research Note RM-435.

Savage, M., P.M. Brown and J. Feddema. 1996. The role of climate in a pine forest regeneration pulse in the southwestern United States. Ecoscience 3:310-318.

Schier, G. A., J. R. Jones, et al. (1985). Vegetative Regeneration. Aspen ecology and management in the western United States. e. N.V. DeByle and R.P. Winokur. Fort Collins, CO, USDA Forest Service Rocky Mountain Forest and Range experimental Station. GTR RM-119: 29-33.

Schmid, J.M. and G.D. Amman. 1992. *Dendroctonus* beetles and old-growth forests in the Rockies. P. 51-59 *in* M.R. Kaufmann, W.H. Moir, and R.L. Bassett, editors, Old-growth forests in the Southwest and Rocky Mountain regions. USDA Forest Service, General Technical Report RM-213.

Schmid, J.M. and R.H. Frye. 1977. Spruce beetle in the Rockies. USDA Forest Service, General Technical Report RM-49.

Schmid, J.M. and T.E. Hinds. 1974. Development of spruce-fir stands following spruce beetle outbreaks. USDA Forest Service Research Paper RM-131.

Schmid, J.M. and S.A. Mata. 1996. Natural variability of specific forest insect populations and their associated effects in Colorado. USDA Forest Service, General Technical Report RM-275.

Schoennagel, T.L., M.G. Turner, and W.H. Romme. 2003. The influence of fire interval and serotiny on postfire lodgepole pine density in Yellowstone National Park. Ecology 84: 2967-2978.

Schoennagel, T.L., T.T. Veblen and W.H. Romme. 2004. The interaction of fire, fuels and climate across Rocky Mountain forests. BioScience 54: 661-676.

Schubert, G.H. 1974. Silviculture of southwestern ponderosa pine pine: The state of our knowledge. U.S. Forest Service Research Paper RM-123.

Seastedt, T.R. 2002. Base camps of the Rockies: The intermountain grasslands. Pages 219-236 in: J.S. Baron, editor, Rocky Mountain futures: An ecological perspective. Island Press, Washington, D.C.

Selmants, P.C. and D.H. Knight. 2003. Understory plant species composition 30-50 years after clearcutting in southeastern Wyoming coniferous forests. Forest Ecology and Management 185:275-289.

Sherriff, R. 2000. Fire history at high elevation in the Colorado Front Range. Masters thesis, University of Colorado, Boulder, CO.

Sherriff, R., T.T. Veblen, and J.S. Sibold. 2001. Fire history in high elevation subalpine forests in the Colorado Front Range. Ecoscience 8:369-380.

Sherriff, R. 2004. The historic range of variability of ponderosa pine in the northern Colorado Front Range: Past fire types and fire effects. Ph.D. thesis, University of Colorado, Boulder, Colorado.

Sherriff, R. and T.T. Veblen. Unpublished manuscript. Estimation of areas of different fire regime types in ponderosa pine forests of the northern Colorado Front Range.

Shimek, S.W. 1996. A dendrochronological history of western spruce budworm (*Choristoneura occidentalis* Freeman) infestations and outbreaks in the northern Colorado Front Range. Masters Thesis, University of Colorado, Boulder, Colorado.

Skinner, T.V. and R.D. Laven. 1983. A fire history of the Longs Peak region of Rocky Mountain National Park. Pages 71-74 *in* Proceedings of the seventh conference on fire and forest meteorology, April 25-28. American Meteorological Society, Boston, Massachusetts.

Sibold, J. 2001. The forest fire regime of an upper montane and subalpine forest, Wild Basin, Rocky Mountain National Park. Masters Thesis, University of Colorado, Boulder, CO.

Sibold, J.S. and T.T. Veblen. Unpublished manuscript. Relationships of fire to climatic variability in the subalpine zone of the northern Front Range, Colorado, USA.

Sibold, J.S., T.T. Veblen, and M.E. Gonzalez. Unpublished manuscript. Spatial and Temporal Variation in Historic Fire Regimes in Subalpine Forests across the Colorado Front Range.

Simmons, V.M. 2000. The Ute Indians of Utah, Colorado, and New Mexico. University Press of Colorado, Boulder, CO.

Smith, P. 1981. A look at Boulder from settlement to city. Pruett, Boulder.

Speer, J.H., T.W. Swetnam, B.E. Wickman, and A. Youngblood. 2001. Changes in pandora moth outbreak dynamics during the past 622 years. Ecology 82:679-697.

Steadman, D.W., and J.I. Mead (editors). 1995. Late quaternary environments and deep history: a tribute to Paul S. Martin. The Mammoth Site of Hot Springs, South Dakota, Inc. Scientific Papers Volume 3.

Stewart, O.C. 1942. Culture element distributions: XVIII Ute-Southern Paiute. University of California Anthropological Records 6:231-356.

Stewart, O.C. 1956. Fire as the first great force employed by man. Pages 115-133 *in* W.L.J. Thomas, editor, Man's role in changing the face of the earth. University of Chicago Press, Chicago, Illinois.

Stewart, O.C. 2002. Forgotten fires: Native Americans and the transient wilderness. University of Oklahoma Press, Norman, Oklahoma.

Stockton, C.W. 1976. Long-term streamflow reconstruction in the upper Colorado River basin using tree rings. Pages 401-441 *in* C.G. Clyde, D.H. Falkenborg, and J.P. Riley, editors, Colorado River basin modeling studies. Utah State University, Logan, Utah.

Stohlgren, T.J., T.N. Chase, R.A. Pielke, Sr, T.F.G. Kittel, and J.S. Baron. 1998. Evidence that local land use practices influence regional climate, vegetation, and stream flow patterns in adjacent natural areas. Global Change Biology 4:495-504.

Sudworth, G.B. 1900. White River Plateau Timber Land Reserve. Pages 117-179 *in* Twentieth Annual Report of the United States Geological Survey, Part V. Forest Reserves. United States Government Printing Office, Washington, D.C.

Suzuki, K., H. Suzuki, D. Binkley, and T.J. Stohlgren. 1999. Aspen regeneration in the Colorado Front Range: differences at local and landscape scales. Landscape Ecology 14:231-237.

Swanson, F.J., J.A. Jones, D.O. Wallin, and J.H. Cissel. 1993. Natural variability - implications for ecosystem management. Pages 89-103 *in* M.E. Jensen and P.S. Bourgeron, editors, Ecosystem management: principles and applications, Volume II, Eastside Forest Health Assessment. USDA Forest Service, General Technical Report PNW-318.

Swetnam, T.W., C.D. Allen, and J.L. Betancourt. 1999. Applied historical ecology: Using the past to manage for the future. Ecological Applications 9:1189-1206.

Swetnam, T.W. and C.H. Baisan. 1996. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 11-32 in C.D. Allen, editor. Fire effects in southwestern forests. Proceedings of the second La Mesa fire symposium. U.S. Department of Agriculture Forest Service, General Technical Report RM-286.

Swetnam, T.W. and J.L. Betancourt. 1992. Temporal patterns of El Niño/Southern Oscillation - wildfire teleconnections in the southwestern United States. Pages 259-270 *in* H.F. Diaz and V. Markgraf, editors, El Niño: historical and paleoclimatic aspects of the Southern Oscillation.

Swetnam, T.W. and J.L. Betancourt. 1998. Mesoscale disturbance and ecological response to decadal variability in the American Southwest. Journal of Climate 11:3128-3147.

Swetnam, T.W. and A.M. Lynch. 1989. A tree-ring reconstruction of western spruce budworm history in the southern Rocky Mountains. Forest Science 35:962-986.

Swetnam, T.W. and A.M. Lynch. 1993. Multicentury, regional-scale patterns of western spruce budworm outbreaks. Ecological Monographs 63:399-424.

Tausch, R. J. 1999. Historic pinyon and juniper woodland development. *In:* Monsen, S. B. and R. Stevens, comps. Proceedings: ecology and management of pinyon-juniper communities within the Interior West; 1997 September 15-18. Provo, UT. Proc.RMRS-P-9. Ogden, UT. USDA Forest Service. Rocky Mountain Research Station. Pp. 12-19.

Thornton, R. 1987. American Indian holocust and survival: a population history since 1492. University of Oklahoma Press, Norman, Oklahoma.

Tice, J.H. 1872. Over the plains and on the mountains. Industrial Age Printing, St. Louis.

Tinker, D.B., C.A.C. Resor, G.P. Beauvais, K.F. Kipfmueller, C.I. Fernandes, and W.L. Baker. 1998. Watershed analysis of forest fragmentation by clearcuts and roads in a Wyoming forest. Landscape Ecology 13:149-165.

Tomback, D.F., L.A. Hoffman, and S.K. Sund. 1990. Coevolution of whitebark pine and nutcrackers: implications for forest regeneration. USDA Forest Service, General Technical Report INT-270:118-129, Ogden, Utah.

Touchan, R., C.D. Allen and T.W. Swetnam. 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. Pages 33-46 *in* C.D. Allen, editor, Fire effects in southwestern forests. Proceedings of the second La Mesa fire symposium. USDA Forest Service General Technical Report RM-286, Los Alamos, New Mexico.

USDA Forest Service. 1920. Cumulative silivical report - Roosevelt National Forest. Rocky Mountain Range and Experiment Station, Fort Collins, Colorado.

USDA Forest Service. 1972. History of Routt National Forest. Basic preparation 1965; revised 1972. Unpublished report, Routt National Forest, Superintendent's Office, Laramie, Wyoming.

USDA Forest Service. 1995. Draft soil and ecological land unit survey: northern San Isabel and western Pike National Forests, Colorado. USDA Forest Service, Soil Conservation Service, and Foothill Engineering, Inc.

USDA Forest Service. 2002a. Final environmental impact statement to accompany the land and resource management plan. USDA Forest Service. White River National Forest. Glenwood Springs, CO.

USDA Forest Service. 2002b. Forest insect and disease conditions in the Rocky Mountain region. unpublished report. Compilers: J. L. Harris, R. Mask, and J. Witcosky. USDA Forest Service, Lakewood, CO.

USDA Forest Service. 2002c. A Collaborative Approach for Reducing Wildland Fire Risks to Communities and the Environment: 10-Year Comprehensive Strategy. (9 June 2004; <I>www.fireplan.gov/reports/7-19-en.pdf<I>)

US House of Representatives. 1880. Testimony in relation to the Ute Indian outbreak, taken by the Committee on Indian Affairs of the House of Representatives. 46th Congress, 2nd Session. Washington, DC. Mis. Doc. No. 38.

Vale, T.R. 1982. Plants and people: vegetation change in North America. Resource Publications in Geography, Association of American Geographers, Washington, D.C.

Van Devender, T.R., and F.G. Hawksworth. 1986. Fossil mistletoes in packrat middens of the southwestern United States. Madroño 33:85-99.

Van Wagner, C.E. 1977. Conditions for the start and spread of crown fire. Canadian Journal of Forest Research 7: 23-34.

Veblen, T.T. 1986a. Age and size structure of subalpine forests in the Colorado Front Range. Bulletin of the Torrey Botanical Club 113:225-240.

Veblen, T.T. 1986b. Treefalls and the coexistence of conifers in subalpine forests of the central Rockies. Ecology 67:644-49.

Veblen, T.T. 2000. Disturbance patterns in southern Rocky Mountain forests. Pages 31-54 *in* R.L. Knight, F.W. Smith, S.W. Buskirk, W.H. Romme and W.L. Baker, editors, Forest fragmentation in the Southern Rocky Mountains. Colorado University Press, Boulder, Colorado.

Veblen, T.T. 1992. Regeneration dynamics. Pages 152-187 *in* D.C. Glenn-Lewin, R.K. Peet and T.T. Veblen, editors, Plant succession: theory and prediction. Chapman and Hall, London.

Veblen, T.T. 2003a. An introduction to key issues in fire regime research for fuels management and ecological restoration. Pages 259-276 in: P. Omi and L. Joyce (technical eds). Fire, Fuel Treatments and Ecological Restoration: Conference proceedings; 2002 16-18 April; Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 475 p.

Veblen, T.T. 2003b. Historic range of variability of mountain forest ecosystems: concepts and applications. Forest Chronicle 79:223-226.

Veblen, T.T., K.S. Hadley, M.S. Reid, and A.J. Rebertus. 1989. Blowdown and stand development in a Colorado subalpine forest. Canadian Journal of Forest Research 19:1218-1225.

Veblen, T.T., K.S. Hadley, and M.S. Reid. 1991a. Disturbance and stand development of a Colorado subalpine forest. Journal of Biogeography 18:707-716.

Veblen, T.T., K.S. Hadley, M.S. Reid, and A.J. Rebertus. 1991b. Methods of detecting past spruce beetle outbreaks in Rocky Mountain subalpine forest. Canadian Journal of Forest Research 21: 242-254.

Veblen, T.T., K.S. Hadley, M.S. Reid, and A.J. Rebertus. 1991c. Stand response to spruce beetle outbreak in Colorado subalpine forests. Ecology 72: 213-231.

Veblen, T.T., K.S. Hadley, E.M. Nel, T. Kitzberger, M.S. Reid, and R. Villalba. 1994. Disturbance regime and disturbance interactions in a Rocky Mountain subalpine forest. Journal of Ecology 82:125-135.

Veblen, T.T. and T. Kitzberger. 2002. Inter-hemispheric comparison of fire history: The Colorado Front Range, U.S.A. and the Northern Patagonian Andes, Argentina. Plant Ecology 163: 187-207.

Veblen, T.T., T. Kitzberger and J. Donnegan. 2000. Climatic and human influences on fire regimes in ponderosa pine forests in the Colorado Front Range. Ecological Applications 10:1178-1195.

Veblen, T.T., T. Kitzberger and J. Donnegan. 1996. Fire ecology in the wildland/urban interface of Boulder County. Unpublished report to the City of Boulder Open Space Department, Boulder, Colorado.

Veblen, T.T. and D.C. Lorenz. 1986. Anthropogenic disturbance and recovery patterns in montane forests, Colorado Front Range. Physical Geography 7:1-24.

Veblen, T.T. and D.C. Lorenz. 1991. The Colorado Front Range: a century of ecological change. University of Utah Press, Salt Lake City, Utah.

Villaba, R. and T.T. Veblen. 1998. Influences of large-scale climatic variability on episodic tree mortality at the forest-steppe ecotone in northern Patagonia. Ecology 79:2624-2640.

Walsh, S.J, D.R. Butler, T.R. Allen, and G.P. Malanson. 1994. Influence of snow patterns and snow avalanches on the alpine treeline ecotone. Journal of Vegetation Science 5:657-672.

Wagner, M.R., W.M. Block, B. Geils, and K.F. Wenger. 2000. Restoration ecology: A new forest management paradigm, or another merit badge for foresters? Journal of Forestry :22-27.

Walters, J.W., T.E. Hinds, D.W. Johnson, and J. Beatty. 1982. Effects of partial cutting on diseases, mortality, and regeneration of Rocky Mountain aspen stand. USDA Forest Service, Research Paper RM-240.

Weaver, T., D. Gustafson, and JH. Lichthardt. 2001. Fire effects on nitrogen pools and dynamics in terrestrial ecosystems: a meta-analysis. Ecological Applications 11: 1349-1365.

Weber, W.A. 1990. Colorado flora: Eastern Slope. University Press of Colorado, Niwot, Colorado.

West, E. 1997. The Way to the West: Essays on the Central Plains. University of New Mexico Press, Albuquerque.

Williams, M.W., M. Losleben, N. Caine, and D. Greenland. 1996. Changes in climate and hydrochemical responses in a high-elevation catchment in the Rocky Mountains, U.S.A. Limnology and Oceanography 41: 939-946.

Whipple, S.A., and R.L. Dix. 1979. Age structure and successional dynamics of a Colorado subalpine forest. American Midland Naturalist 101:142-158.

White, P.S. and S.T.A. Pickett. 1985. Natural disturbance and patch dynamics: an introduction. Pages 3-13 *in* S.T.A. Pickett and P.S. White, editors, The ecology of natural disturbance and patch dynamics. Academic Press, Inc., Orlando, Florida.

White, P.S. and J.L. Walker. 1997. Approximating nature's variation: Selecting and using reference information in restoration ecology. Restoration Ecology 5: 338-349.

White House. 2002. Healthy forests: An initiative for wildfire prevention and stronger communities. (10 June 2004; www.whitehouse.gov/infocus/healthyforests/Healthy_Forests_v2.pdf)

Whitney, S. 1992. Western Forests. Alfred A. Knopf, Inc., New York, New York.

Wier, J. 1987. Ute-White relationships in Middle Park: Governmental and semi-official. Grand County Historical Association Journal 7:57-75.

Wohl, E.E. 1992. Prehistoric floods in the Poudre Canyon. Progress Report to City of Fort Collins, Colorado.

Wohl, E. E. 2001. Virtual rivers: lessons from the mountain rivers of the Colorado Front Range. Yale University Press. New Haven, CT.

Wolle, M.S. 1949. Stampede to timberline. Sage, Denver.

Wood, R.E. 1983. Mortality caused by root diseases and associated pests on six National Forests in Arizona and New Mexico. USDA Forest Service, Southwestern Region, Forest Pest Management Report R3-83-13.

Woodhouse, C.A. 1993. Tree-growth response to ENSO events in the central Colorado Front Range. Physical Geography 14:417-435.

Woodhouse, C.A. 2001. A tree-ring reconstruction of streamflow for the Colorado Front Range. Journal of the Water Resources Association 37:561-570.

Woodhouse, C.A., and P.M. Brown. 2001. Tree-ring evidence for Great Plains drought. Tree-Ring Research 57:89-103.

Woodhouse, C.A., J.J. Lukas, and P.M. Brown. 2002. Drought in the western Great Plaines, 1845-56. Bulletin of the Am. Meteor. Soc. 83: 1485-1493.

Woodhouse, C.A. and J.T. Overpeck. 1998. 2000 years of drought variability in the central United States. Bulletin of the American Meteorological Society 79:2693-2714.

Woodward, A., E.G. Schreiner and D.G. Silsbee. 1995. Climate, geography, and tree establishment in subalpine meadows of the Olympic Mountains, Washington, U.S.A. Arctic and Alpine Research 27:217-225.

Wright, H. A. and A. W. Bailey. 1982. Fire ecology: United States and southern Canada. John Wiley and Sons, New York.

Wyckoff, W. 1999. Creating Colorado: The making of a western American landscape, 1860-1940. Yale University Press, New Haven, Connecticut.

Zimmerman, G.T. and R.D. Laven. 1984. Ecological implications of dwarf mistletoe and fire in lodgepole pine forests. Pages 123-131 *in* G.G. Hawksworth and R.F. Scharpf, editors, Proceedings of the symposium on the biology of dwarf mistletoes. USDA Forest Service, General Technical Report RM-111.

Zimmerman, G.T. and R.D. Laven. 1987. Effects of forest fuels smoke on dwarf mistletoe seed germination. Great Basin Naturalist 47:652-659.

Table 3.1 Details of the instrumental climate stations used for describing temperature and precipitation trends in the northern Front Range (Fig. 4.2). The period of the record refers to the time period of each record used in the composite records to describe regional trends (Fig. 4.2). Data from: the Colorado Climate Center of Colorado State University (http://ulysses.atmos.colostate.edu/) and the Institute for Alpine and Arctic Research of the University of Colorado at Boulder.

Station	Elevation	Period of the Record	
	(m)	Temperature	Precipitation
High Elevation East of the Continental Divide			
CU Mt. Research Station D-1	3743	1951-1997	1951-1997
Silver Lake	3111		1895-1950
CU Mt. Research Station C-1	3018	1951-1997	1951-1997
Longs Peak	2745	1895-1944	1895-1944
Allenspark	2593		1945-1993
High Elevation West of the Continental Divide			
Dillon	2715	1910-1997	1909-1997
Fraser	2611		1910-1974
Grand Lake SW	2556	1949-1997	1949-1997
Mid-Elevation on the Eastern Slope			
Estes Park	2364	1916-1994	1909-1994
Idaho Springs	2306	1905-1974	
Piedmont of the Eastern Slope			
Boulder	1647	1893-1997	1893-1997
Denver	1613	1873-1997	1872-1997
Fort Collins	1525		1893-1997

Table 3.2 Instrumental climate station locations, elevations, and periods of record used for selected stations in and near the PSI N.F.

Station Name	Latitude	Longitude	Elevation (m)	Period	
				Temp.	Precip.
Buena Vista	38°50'	106°08'	2432	1905-1998	1899-1998
Cañon City	38°26'	105°16'	1628	1897-1998	1897-1998
Cheesman Res.	39°13'	105°17'	2100	1905-1998	1905-1998
Denver	39°45'	105°00'	1591	1872-1974	1872-1974
Leadville	39°15'	106°18'	3103	1897-1982	1897-1982
Salida	38°32'	106°00'	2149	1897-1998	1897-1998
Westcliffe	38°08'	105°29'	2936	1931-1998	1932-1998

Table 3.3 Cover types of the Arapaho and Roosevelt National Forests (from USDA Forest Service RIS database 1999). Percentages refer to the sum of the cover types in the table and exclude land surfaces of lakes, rocks, and alpine tundra.

Forest Type	Total Area (ha)	Percent
Lodgepole Pine	207,635	44.6
Spruce - Fir	100,708	21.6
Ponderosa Pine	57,089	12.3
Natural grasslands	25,536	5.5
Douglas - Fir / Mixed Conifer	24,456	5.3
Shrublands	19,189	4.1
Aspen	18,297	3.9
Krummholz	8,215	1.8
Limber Pine	2,451	0.5
Bristlecone Pine	912	0.2
Cottonwood - Willow	384	0.1
Wetlands	362	0.1
Total	465,234	100

Table 3.4 Cover types of Pike and San Isabel National Forests (from USDA Forest Service, RIS database 1999).

	Pike National Forest		**San Isabel National Forest**	
	Ha	%	Ha	%
Aspen	32370	7.3	40742	9.2
Bristlecone Pine	12266	2.8	4904	1.1
Blue Spruce	1298	0.3	36	0.0
Douglas-Fir	103206	23.4	74559	16.8
Limber Pine	11052	2.5	2084	0.5
Lodgepole Pine	35718	8.1	46475	10.4
Ponderosa Pine	103533	23.5	33943	7.6
Pinyon-Juniper	488	0.1	24078	5.4
Spruce-Fir	70538	16.0	88137	19.8
Cottonwood	5	0.0	68	0.0
Water/Lakes	256	0.1	2126	0.5
Shrubland	14672	3.3	31035	7.0
Grassland	44931	10.2	43574	9.8
Barren/Rock	10954	2.5	53337	12.0
Total	441282		445098	

Table 5.1. Three general types of historic fire regimes recognized for the forested portion of the northern Colorado Front Range. See text for explanations of possible differences from the southern Front Range. Modified from Romme et al. 2003a.

Type of regime	Major characteristics	Major drivers	Distribution
Frequent, low-severity	Relatively short fire return intervals (5 to 30 years) to the same stand of c. 100 ha burning mainly on the ground surface; non-lethal to mature trees.	Dependent on sufficient accumulation of fine fuels following the previous fire. Favored by short droughts and often lag by several years of above average moisture favoring growth of fine fuels.	In the Front Range, limited to the lower elevations of the ponderosa pine cover type.

Infrequent, high-severity	Long fire return intervals, typically > 100 years and often > 400 years; high-severity burns, typically killing all canopy trees over areas of 100s or 1000s of ha.	Frequent lightning ignitions fail to spread in most years due to relatively moist fuels. Spread is highly dependent on extreme droughts during which fuel configurations have only a minor influence on fire behavior.	In the Front Range, typical of the subalpine zone of spruce-fir and lodgepole pine forests. Also typical of aspen, although large areas of aspen my impede fire spread in years of less extreme drought.
Mixed severity	Complex pattern of both high-severity (stand-replacing or partially stand-replacing) and low-severity (non-lethal, surface) fires occurring at highly variable intervals. Mean return intervals for the stand-replacing component range from c. 40 to 100 years at stand scales of 100 ha. Most fire events include both stand-replacing and low-severity components but their proportions are highly variable across the landscape.	Years of widespread fire are driven both by drought in the same year, and lagged by several years from periods of above average moisture availability that favor the production of fine fuels. Local site factors appear highly important in influencing fire spread in many fire events, but under extreme weather fuel configurations become less important.	In the Front Range, in the mid and upper montane zone of ponderosa pine and Douglas-fir forests.

Table 5.2. Spatial associations of fires with biotic and abiotic factors in Rocky Mountain National Park (RMNP), Arapaho and Roosevelt National Forests (AR), and Pike and San Isabel National Forests (PSI). The number of fires analyzed are 254 for the period 1915-1995 for RMNP, 1092 for the period 1970 - 1995 for AR, and 1906 for the period 1961-1998 for PSI. High elevation is >

3000 m, and low elevation is < 2750 m. Departures from observed and expected frequencies of spatial associations were tested with a chi-square analysis of fire frequency associated with a variable compared to frequency not associated with the variable. Higher indicates a statistically significant greater than expected co-occurrence. Significance levels are: *= p < 0.05; **= p < 0.01; and ***= p < .001. Lack of significance is indicated by n.s., and lack of data by n.a. Data sources are: RMNP GIS, PSI GIS, and AR GIS.

Variable	Rocky Mountain N. P.		Arapaho and Roosevelt N.F.		Pike and San Isabel N.F.	
	All fires (n = 254)	Lightning fires (n = 78)	All fires (n = 1092)	Lightning fires (n = 504)	All fires (n=1906)	Lightning fires (n=1023)
Cover type						
Spruce-fir	Lower***	Lower***	Lower***	Lower***	Lower***	Lower***
Lodgepole pine	n.s.	n.s.	Lower***	Lower***	Lower*	Lower***
Aspen	n.s.	n.s.	n.s.	n.s.	Lower***	Lower***
Limber pine	n.s.	n.s.	n.s.	n.s.	Lower*	n.s.
Bristlecone pine	n.a.	n.a.	n.a.	n.a.	Lower***	Lower***
Douglas-fir	Higher*	n.s.	Higher***	Higher***	Higher***	Higher***
Ponderosa pine	Higher***	Higher**	Higher***	Higher***	Higher***	Higher***
Pinyon-juniper	n.a.	n.a.	n.a.	n.a.	Lower***	Lower*
Abiotic factors						
High elevation	Lower***	Lower***	Lower***	Lower***	Lower***	Lower***
Low elevation	Higher***	Higher***	Higher***	Higher***	Higher***	Higher***

North-facing	n.s.	n.s.	n.s.	n.s.	Lower***	Lower***
South-facing	n.s.	n.s.	n.s.	n.s.	Higher*	Higher***

Table 6.1. Areas of forest cover types in Arapaho and Roosevelt National Forest affected by management activities. The three management classes listed are mutually exclusive, but minor use classifications (e.g. wood removal following a natural disturbance) are not included. Tie hack refers to early logging for railroad ties. Silvicuturally managed stands include stands which have been cut using a whole range of harvesting systems from individual tree selection to clear cuts. Source: USDA Forest Service RIS database 1999.

Forest Type	Total Area (ha)	Not managed (%)	Tie hacked (%)	Silviculturally managed (%)
Aspen	18,297	91.0	0.9	6.4
Spruce - Fir	100,708	87.0	3.0	8.2
Douglas - Fir / Mixed Conifer	24,456	88.0	0.3	6.7
Ponderosa Pine	57,089	85.0	0.9	7.1
Limber Pine	2,451	98.0	1.3	0.5
Lodgepole Pine	207,635	85.0	3.5	10.1
Bristlecone Pine	912	100.0	0.0	0.0
Cottonwood - Willow	384	100.0	0.0	0.0
Krummholz	8,215	100.0	0.0	0.0
Total	431,121	86.0	2.6	8.4

Table 7.1. Summary comparison of the current forested landscape and the range of landscape variation expected for c. 1500 to 1850 A.D. for the northern Front Range. See text for discussion of differences from the southern Front Range. Possible changes in fire regimes are stressed in the table. As explained in the text, the historical and current landscapes would have been characterized by similar levels of bark beetle and budworm outbreaks.

Zone and Main Cover Types	Historical Landscapes	Current Landscapes
Lower Montane Ponderosa pine	Pattern: Open woodlands of ponderosa pine, extensive grasslands, riparian forests, small dense patches of ponderosa, shrublands.	Pattern: More continuous forest cover and generally denser pine stands than occurred historically. Extreme conversion and fragmentation of natural landscape.
	Mechanisms: Moderately, frequent low-severity fires maintained open pine woodlands; patches of higher-severity fires resulted in openings or dense regeneration of pines or shrubs	Mechanisms: 20th century fire exclusion, late 19th and early 20th century grazing and logging conducive to ponderosa pine establishment. Widespread exurban development.

Mid and Upper Montane Ponderosa pine Douglas-fir / mixed conifer	Pattern: Heterogeneous landscape mainly of ponderosa pine-dominated patches of variable sizes and ages, Douglas-fir on more mesic sites, openings consisting of grasslands and severely burned former forest sites. Mechanisms: A mixed severity fire regime in which forest structure was shaped mainly by severe fires; low-intensity fires were less significant in forested areas but probably important in meadows.	Pattern: Still highly heterogeneous landscape in relation to site conditions influencing stand densities and composition, but much less heterogeneous forest stand ages mostly dating from c. 1850 to 1920. Meadows persist but show limited tree encroachment. Relative dominance of ponderosa pine and Douglas-fir not significantly changed from the historic landscape. Mechanisms: Major influence of severe, widespread fires of the 2nd half of the 19th century reflected in even-aged post-fire stands; relatively young stands also triggered by logging and other anthropogenic disturbances. Substantial exurban development.

Subalpine Lodgepole pine Aspen Spruce-fir	Pattern: Very large patches of even-aged forests varying in composition from pure lodgepole pine or aspen to spruce-fir. Mechanisms: Infrequent, high-severity fires followed both by successional replacement of species or recovery to the same dominant tree species according site conditions and seed/sprout availability.	Pattern: Relatively unchanged from the historical patterns except where logging or exurban development has affected limited areas. Mechanisms: Fire regimes have not changed significantly from the historic fire regime of large fires occurring at highly variable intervals.

APPENDIX 1: NOTABLE EARLY FIRES IN THE FRONT RANGE AS RECORDED IN HISTORICAL TEXTS

Year	**Location**	**Notes**	**Source**
c. 1848	Pikes Peak area	Reportedly set by Native Americans	Jack 1900, p. 69

1859	Chicago Creek just north of Pike N.F.	Three men were killed in the fire	Ingwall 1923, p. 41
1860	Gold Hill, Boulder County	Town destroyed by forest fire	Wolle 1949
1863	Missouri Gulch	Large forest fire	Ingwall 1923, p. 40
1866	Lower Falls, central Clear Creek County	One of the largest fires reported up to this date	Ingwall 1923, p. 40
1869	Clear Lake, Mount Evans		Ingwall 1923, p. 41
1868 or 1869	Tarryall Mountains	Fire burned 20 miles in length and up to 6 or 8 miles in width	Jack 1900 p. 97; Ingwall 1923, p. 41
1870	Fairplay	Burned the timber across the river from Fairplay	Ingwall 1923, p. 43
1870	Lost Park	It was so smoky at times one could see only a short distance	Ingwall 1923, p. 41
1871	Fall River, Clear Creek County	A very large fire burned the headwaters of Fall River	Ingwall 1923, p. 40
c. 1871	Mammoth Basin, Gilpin County	The greatest conflagration up to that time in Gilpin County	Ingwall 1923, p. 40
1873	Caribou, Boulder County	Town destroyed by fire	Smith 1981
1879	Twelve Mile Creek	Burned a large area of timber in north Clear Creek	Ingwall 1923, p. 40

June 1879	Lost Creek	Three large forest fires burned for about a month. Observed by the U.S. Army Signal Corps from Pikes Peak	Murphy 1982, p. 102
1879	Horseshoe		Ingwall 1923, p. 43
1879	Caribou, Boulder County	Town destroyed by fire again	Smith 1981
1879	South Park		Ingwall 1923, p. 43
c. 1880	Lost Creek	A great fire according to local legend	Murphy 1982, p. 35
c. 1880	North of Fountain Creek	Large burn	Jack 1900, p. 69
1880	Victor, Seven Lakes and the Middle Beaver Creek	Fire set by settlers south of Victor, spread northeast over Bull Mountain, by the head of Bison Creek across the Seven Lakes to the west slopes of Middle Beaver Creek and burned several thousand acres	Ingwall 1923, p. 41
1881	Black Mountain		Ingwall 1923, p. 43
1881	South Park		Ingwall 1923, p. 43
1891 or 1892	Marshall Pass		Agee and Cuenin 1924, p. 18
1891 or 1892	Vance Creek	Started by campers. Burned several hundred acres for several months.	Ingwall 1923, p. 43

1893	Head of the Saguache Creek	Set by miners leaving Creed during the great panic	Agee and Cuenin 1924, p. 18
1894	Upper Fourmile Canyon and Left Hand Canyon	Large fire nearly destroys Gold Hill again	Smith 1981
1898	Breckenridge Pass	Fire burned to treeline at 11,500 feet. Started by sparks from the Colorado and Southern Railway (South Park Line)	Jack 1900, p. 98
1898	Near Kenosha Pass	Grass fires and incipient timber fires started from sparks thrown by locomotives of the Colorado and Southern Railway	Jack 1900, p. 98
c. 1900	Eldora area, southwestern Boulder County	Fire burned 29,000 acres	Kemp 1960
1901	Currant Creek south of Thirty-nine Mile Mountain	Burned over 600 acres. Started in slash by a cook at the paper company camp. The timber cutters started a second fire about a half a mile away to make it appear the first fire had been set by campers, but due to a wind change the two fires did not unite.	Ingwall 1923, p. 42

1903	Tyler Gulch, near Bailey	3000 acres burned. Started intentionally by wood choppers.
1908	Mount Evans	Six square miles burned.

APPENDIX 2: Historical and Matched Landscape Photographs of Sites in Arapaho- Roosevelt National Forest and Rocky Mountain National Park

The following Plates consist of historical and modern landscape photographs published in Veblen and Lorenz (1991). Sources of specific photographs, precise locations, and other details are given in Veblen and Lorenz (1991).

Plate 1 (Plate 17 in Veblen and Lorenz 1991). Sunset, Boulder County.

The slope in the upper right had been recently burned by a stand-replacing fire at the time of the 1898 photograph. In the mid-ground and foreground, the density of the ponderosa pine woodland has increased.

Plate 2 (Plate 11 in Veblen and Lorenz 1991). Red Hill Valley, Boulder County.

This valley generally experienced an increase in the density of the ponderosa pine woodland. The distant slope at the head of the valley has converted from open woodland to dense forest of ponderosa pine. In the mid-ground and foreground, the lack of stumps or logs in the old photograph indicates that this area was not forested during the late 19th century.

Plate 3 (Plate 13 in Veblen and Lorenz 1991). McAllister Sawmill, Boulder Canyon.

This stand of mainly Douglas-fir plus some ponderosa pine was logged during the 1870s. Today, it is a dense even-aged stand of young Douglas-fir.

Plate 4 (Plate 14 in Veblen and Lorenz 1991). Sunshine, Boulder County.

The mining town of Sunshine had a population of 1500 people in 1876. Slopes which originally were mostly grassland and lacked stumps or snags, have converted to denser stands of ponderosa pine.

Plate 5 (Plate 19 in Veblen and Lorenz 1991). Frankeberger Point, Fourmile Canyon, Boulder County.

The slopes of Douglas-fir and ponderosa pine in the background were dense stands that had been logged and/or burned in the late 19th century. In the modern photograph they are dense, even-aged stands dominated by Douglas-fir. In the right foreground, a montane meadow has been invaded by ponderosa pine.

Plate 6 (Plate 30 in Veblen and Lorenz 1991). Northwest slope of Sugarloaf Mountain, Boulder County.

This slope of dense Douglas-fir and ponderosa pine was cut and burned by stand-replacing fire in the mid-19th century. Visible in the old photograph is an area of greater tree survival on a rocky site where the original stand density was less.

Plate 7 (Plate 51 in Veblen and Lorenz 1991). Deer Mountain from Prospect Mountain, RMNP.

This upper montane site shows that the 19th century landscape structure included some dense patches of ponderosa pine and Douglas-fir, open woodlands and also treeless meadows. The mid- and foreground show a moderate increase in the density of ponderosa pine woodlands.