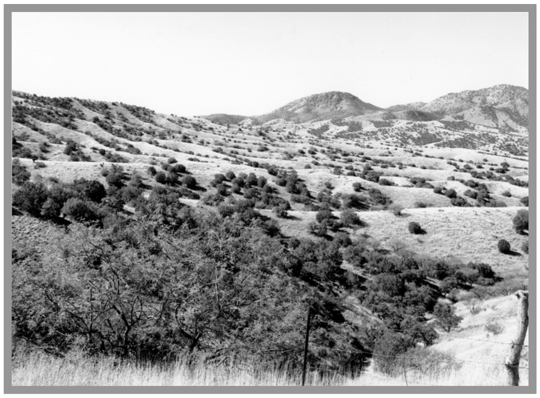
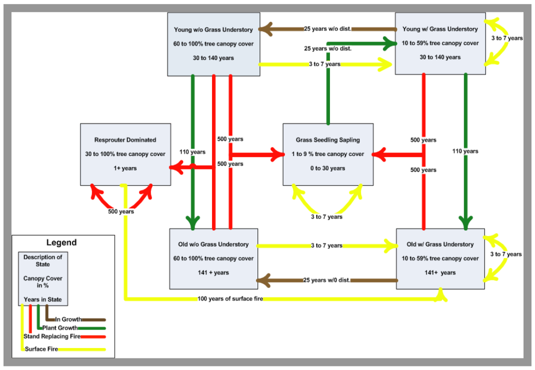
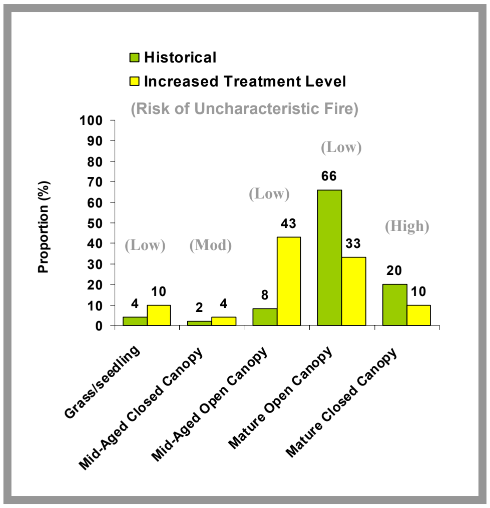
SOUTHWEST FOREST ASSESSMENT PROJECT

HISTORICAL RANGE OF VARIATION AND STATE AND TRANSITION MODELING
OF
HISTORICAL AND CURRENT LANDSCAPE CONDITIONS FOR POTENTIAL NATURAL VEGETATION TYPES OF THE SOUTHWESTERN U.S.







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Top Left:	**1987 photograph of Barrel Canyon near the Empire Mountains in southeastern Arizona. Courtesy of Unites States Geological Survey.**
Top Right:	**Avalanche path and debris from March 2005 slide in Abineau Canyon, San Francisco Peaks north of Flagstaff, AZ. Photo by Edward Smith of The Nature Conservancy.**

Historical Range of Variation
and
State and Transition Modeling of Historic and Current Landscape Conditions for Potential Natural Vegetation Types of the Southwest

The Nature Conservancy
SAVING THE LAST GREAT PLACES ON EARTH

Southwest Forest Assessment Project
Heather Schussman and Edward Smith
2006

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1.1 Introduction

Definition of HRV-

The Historical Range of Variation or Variability (HRV) is a description of the change over time and space in the ecological condition of potential natural vegetation types and the ecological processes that shape those types. Potential natural vegetation types (PNVT) represent the vegetation type and characteristics that would occur when natural disturbance regimes and biological processes prevail (Table 1 – 1). We base HRV descriptions on the best available empirical information that has been documented, peer-reviewed, and published in journals, reports and books (more in Methods, 1.2). For the purposes of this document, HRV descriptions focus on characteristics important for managing PNVTs found on National Forests in Arizona and New Mexico, including: vegetation composition and structure and how this attribute varies across the region within a PNVT; patch or stand characteristics such as size and spatial distribution; patch dynamics such as succession; the dominant disturbance processes and frequency of disturbance that shape ecological conditions within a PNVT over time; anthropogenic disturbances or exclusion of natural disturbance regimes; and the effects of climatic fluctuations.

Table 1-1. List of potential natural vegetation types that exist on Region III forests, for which historical range of variation is investigated. Potential Natural Vegetation Types are coarse scale groupings of ecosystem types that share similar geography, vegetation, and historic disturbance processes such as fire, drought, and native herbivory.

Alpine Tundra	Mixed Conifer forest
Aspen forest and woodland	Montane grassland
Cottonwood willow riparian forest	Montane willow riparian forest
Deserts	Pinyon Juniper woodland
Gallery coniferous riparian forest	Plains grassland
Great Basin grassland	Ponderosa Pine forest
Great Plains Grassland	Sagebrush shrubland
Interior chaparral	Semi-desert grassland
Juniper woodland	Shinnery Oak
Madrean encinal	Spruce-fir forest
Madrean pine oak woodland	Sub-alpine grassland
Mixed broadleaf deciduous riparian forest	Wetlands/cienega

Descriptions of HRV also focus on quantifying the rate of change in PNVT characteristics and the influence of humans on changes in PNVT characteristics. Several authors have noted that contemporary patterns of vegetation and their dynamic processes developed in the Southwest during the early Holocene, around 11,000 to 8,000 years ago (Allen 2002, Anderson 1993, Weng and Jackson 1999). However, due to limitations on the availability of recorded data from tree rings, pollen, and charcoal discussed in the

Methods section (1.2), unless otherwise noted, the time period that we consider to frame the "**Pre-settlement**" portion of the HRV descriptions is between the years 1000 to 1880. Large-scale expansion and westward movement and settlement by United States citizens and European (and other ethnic) immigrants following the Civil War mark the onset of major anthropogenic disturbances in the Southwest: extensive, commercial livestock grazing, river damming and canal construction, railroad logging, and widespread fire regime alteration, all of which have had significant impacts on vegetation and ecological processes (Carlson 1969, deBuys 1985, Allen 1989, Covington and Moore 1994, Touchan and others 1996). Thus we refer to that portion of the HRV that resulted from conditions after 1880 as the "**Post-settlement**" or anthropogenic disturbance period. There is ample evidence to suggest that while aboriginal or Native American influences on the landscape prior to 1800 were detectable in some locations, the magnitude of anthropogenic disturbance after 1880 was much greater (Allen 2002).

We include post-settlement or anthropogenic disturbances as an important part of the HRV for PNVTs because in many cases the pre-settlement vegetation patterns and processes have been significantly altered by humans, not only in magnitude but also in rates of change. When empirical data are available, we document the processes, such as altered herbivory, silvicultural activities, habitat fragmentation, altered hydrology, mining, fire management, and introduction of exotic species of plants and animals. We then describe the effects of these processes on the characteristics, natural processes, and vegetation dynamics observed for PNVTs.

HRV's Application in Land Management Decision-Making – Understanding the response of PNVTs to disturbance processes (or the absence of disturbance processes) and the characteristics of PNVTs over time enables land managers to better characterize components of ecosystem diversity. In the context of land management planning, HRV enables managers to identify desired future conditions and the need for change by comparing current conditions with the range of historical conditions. HRV also describes the evolutionary context for PNVTs present today by identifying the disturbance processes (and variability) that serve as major determinants of PNVT characteristics (Morgan and others 1994). Understanding the relationship among disturbance processes, the responses of organisms to these processes, and current conditions enables managers to evaluate the potential for proposed management actions to meet ecological sustainability goals. Moreover, since the form and function of PNVTs are shaped by these processes, HRV characterizations can assist land managers in evaluating how and where appropriate disturbance regimes may be integrated into management actions.

HRVs characterize a range of *reference conditions* against which ecosystem change, anthropogenic or stochastic, can be measured (White and Walker 1997) and the landscape-scale effects of succession and disturbance on vegetation characteristics over time (Landres and others 1999). Identifying reference conditions and the range of variation is important for identifying land management goals and land-use allocations. Historical Range of Variation descriptions also enable land managers to better predict where management actions are likely to have the greatest effect on restoring some of the patterns and processes identified in the HRV. However, the current biophysical conditions under which land management is practiced are different from the evolutionary environment under which ecological systems developed. For example, climate continues to change, which affects vegetation mortality, reproduction, and disturbance processes. Anthropogenic effects of landscape fragmentation through road construction, exotic species introductions, and fire suppression also contribute to what has been called the "no analogue" condition: the current evolutionary environment may be different from the historic evolutionary environment, and some historical conditions may be neither attainable nor desirable as management goals (Swetnam and others 1999).

The Historic Range of Variation identifies the scope, magnitude, variability and probability of occurrence for processes that govern the form and function of PNVTs. Complete understanding of PNVTs is unattainable, but cataloguing and organizing what is known about systems can give managers easy access to that information and facilitate its incorporation into planning processes and documents. Some aspects of HRV have not been documented in the literature, and some pre-settlement patterns that are documented may not be desirable or attainable given the dynamic nature of climate and ecological systems. However, management actions can be adapted as information gaps are filled, and well designed land management hypotheses can be tested with rigor. HRV does not absolutely define an acceptable range of conditions, but can help with setting meaningful, empirically based boundaries. If the explicit goals of management actions aspire toward conditions that are outside of the HRV (departure), then the rationale used in developing such goals can be evaluated, assumptions documented, and results of pertinent management actions can be monitored closely (Morgan and others 1994). The vegetation characteristics and process probabilities described in an HRV can form the basis for quantitative models of vegetative change by providing the variables that populate the models. Several models have been developed to incorporate a combination of deterministic, stochastic, and probabilistic events into predictive models of ecosystem change (Morgan and others 1994). Models can be used to test the effects of various management scenarios on ecological systems.

In summary, a well researched and organized HRV description enables managers of that system to:

- Understand reference conditions and reference variability for ecological systems;
- Understand the effects of natural disturbance processes in the absence of anthropogenic activities;
- Understand likely direction of ecological systems under various management scenarios and thus help identify and understand the need for change;
- Evaluate and predict management outcomes;
- Understand the relationship between natural disturbance processes and anthropogenic activities in the development of short- and long-term management goals.

Influence of Temporal and Spatial Scale on Reported Values - The effect of scale, both spatial and temporal is well recognized for its importance in HRV descriptions (Morgan and others 1994). Reported values of ecosystem characteristics and processes are dependent upon the scale at which they are measured, and the amount of variability of measured values also varies at different scales (Wiens 1985, Turner and Gardner 1991). For example, species richness (total number of species) increases in many ecosystem types with increasing plot size (Darlington 1957), a tenet that is basic to biogeography. Similarly, the reported values of ecological processes such as fire are dependent upon the temporal and spatial scales at which they are measured, due to differences in topography and aspect (spatial) and climatic changes (temporal). However, spatial variability of topography and aspect can be viewed at multiple scales, from microsite differences operating at the smallest scale of a few feet to the landscape scale of millions of acres. Similarly, climatic differences can operate at multiple scales from short-term drought of a few years, to decadal to century scale trends of long-term drought. Also, size of the sampling area (spatial), and length of the sampling period (temporal) both affect the reported values for ecological processes, resulting in variation in the estimated parameter due to sampling. The selection of the appropriate scales of time and space for HRVs should be based upon the analytical objectives (Bourgeron and Jensen 1993). For this project, the focus of the analysis is in understanding vegetation dynamics for a variety of PNVTs in the Southwest Region of the United States. For this reason, we have chosen to report values for the full extent of each PNVT across the two-state Region III of the United States Forest Service. The spatial scale thus falls into the range of hundreds of thousands to millions of acres, depending on the PNVT, and with the exception of Alpine/Tundra, Gallery Coniferous Riparian Forest, Montane Grassland, and Wetland/Cienega (Table 1-2). Similarly, since the time period of inquiry for establishing HRV focuses on pre- and post-settlement times for these PNVTs, and time scale should encompass multiple generations of vegetation (Morgan and other 1994), the time scale of inquiry is over hundreds of years, from approximately 1000 until the present. Ultimately, we have allowed the availability of published empirical data to be our guide in determining and reporting relevant information regarding the magnitude and variability of ecosystem characteristics and processes for these HRVs.

Table 1-2. Approximate area (in acres) of potential natural vegetation types (PNVTs) in Arizona and New Mexico across major landowners. The Other landowner category in this table includes: Bureau of Reclamation, non-federal parks, Valles Caldera National Preserve, county lands, Department of Energy, USDA Research, State Game and Fish, and unnamed lands. USFS Region 3 National Grasslands in New Mexico, Oklahoma and Texas were not included in this analysis. Data used to generate this table came from The Southwest Regional Gap Analysis Program (SWReGAP) and the landownership GIS-based layer. Note that accuracy testing has not been conducted for SWReGAP data. Total acres in bold indicate the scale for which HRVs were developed.

Potential Natural Vegetation Type	**US Forest Service**	**Bureau of Land Management**	**Department of Defense**	**National Park Service**	**Private**	**State Trust**	**Tribal**	**US Fish and Wildlife Service**	**Other**	**Total**
Alpine/Tundra	1,600	0	0	0	6,100	0	0	0	0	**7,700**
Aspen Forest and Woodland	335,900	500	0	3,400	93,200	2,200	75,900	0	11,600	**522,700**
Barren	0	26,900	13,000	100	35,900	14,900	196,400	2,100	300	**289,600**
Cottonwood Willow Riparian Forest	19,500	74,800	14,900	7,100	219,500	55,600	389,000	28,500	11,000	**819,900**
Deserts	1,018,300	8,593,300	3,537,800	1,321,000	3,418,000	3,340,700	3,429,500	1,583,200	252,800	**26,494,600**
Disturbed/Altered	83,300	9,200	600	6,000	218,200	37,200	47,800	5,600	400	**408,300**
Gallery Coniferous Riparian Forest	100	0	0	0	1,100	0	100	0	0	**1,300**
Great Basin/ Colorado Plateau Grassland and Steppe	684,400	2,853,400	23,000	572,300	5,695,500	2,599,300	12,175,500	43,200	18,500	**24,665,100**
Great Plains Grassland	316,800	1,270,300	29,000	10,000	16,055,000	3,158,400	181,000	14,100	11,400	**21,046,000**
Interior Chaparral	1,345,900	414,600	33,800	31,300	590,500	350,800	333,100	6,400	11,000	**3,117,400**
Madrean Encinal Woodland	2,736,200	518,800	151,400	34,400	1,259,800	609,300	1,165,200	14,800	2,200	**6,492,100**
Madrean Pine-Oak Woodland	831,900	20,200	1,700	5,000	89,200	30,100	438,400	100	200	**1,416,800**
Mixed Broadleaf Deciduous Riparian Forest	42,600	36,200	5,000	4,200	115,800	17,300	65,500	7,900	4,300	**298,800**
Mixed Conifer Forest	1,216,300	33,900	2,700	43,500	225,900	13,800	191,000	1,000	52,000	**1,780,100**
Montane Grassland	17,200	0	0	0	16,900	0	2,300	0	0	**36,400**
Montane Willow	17,300	14,400	800	600	42,800	11,500	12,100	100	4,100	**103,700**

Potential Natural Vegetation Type	**US Forest Service**	**Bureau of Land Management**	**Department of Defense**	**National Park Service**	**Private**	**State Trust**	**Tribal**	**US Fish and Wildlife Service**	**Other**	**Total**
Riparian Forest										
Pinyon-Juniper Woodland	3,375,200	2,872,700	22,300	556,700	4,442,500	1,505,300	5,647,800	19,000	51,600	**18,493,100**
Ponderosa Pine Forest	5,835,300	112,500	16,400	94,200	1,408,400	147,000	1,588,900	900	44,100	**9,247,700**
Sagebrush Shrubland	134,500	685,200	1,600	66,300	642,100	184,700	977,200	21,200	11,700	**2,724,500**
Semi-desert Grassland	1,642,300	8,013,000	1,463,300	99,000	7,996,600	5,914,600	951,900	321,000	185,000	**26,586,700**
Spruce-fir Forest	355,200	35,000	1,000	7,000	128,200	2,300	72,000	300	10,000	**611,000**
Sub-alpine Grasslands	311,700	13,900	200	2,500	183,400	10,700	55,700	0	27,000	**605,100**
Urban/Agriculture	20,800	35,100	49,200	2,300	4,119,500	219,000	334,900	5,600	23,900	**4,810,300**
Water	25,300	25,000	2,300	79,100	122,000	900	38,100	15,600	55,500	**363,800**
Wetland/Cienega	8,900	9,500	200	400	35,000	7,100	6,800	2,900	1,100	**71,900**

Urgency, Limitations, Assumptions, and Misuse of HRV – As time passes, fewer records of HRV are available to help fill in gaps in our knowledge; old trees, snags, stumps and logs burn or decay, and records from professionals who have witnessed change are lost or not archived making it difficult to assess some important sources of information before they are gone. It is important to prioritize data gaps and to encourage efforts to fill gaps, although in many cases, pre-settlement information may never be available. Historical data must be interpreted with caution, as it is not always possible to assign causation to observed phenomena, as confounding factors may not always be discernible, and their relative contribution to observed records may not be accountable (Morgan and others 1994).

Use of Reference Sites - When historical data are lacking, especially for pre-settlement conditions, it has been suggested that areas with relatively unaltered disturbance regimes can be used to assess and describe the HRV for an area of similar biophysical setting (Morgan and others 1994). Hence, wilderness areas with intact fire regimes, or research natural areas where livestock grazing has been excluded, and riverine systems with intact flow regimes for example may provide valuable information on ecosystems where these disturbance regimes have been altered in a majority of sites or areas. However, the degree to which even large wildernesses have been affected by humans, and the lack of breadth of biophysical settings represented by preserved areas limit the availability of reference sites. Within each PNVT description, we have identified reference sites that were used for developing its HRV.

1.2 Methods Used in Determining HRV

Introduction - We utilized extensive library searches of Northern Arizona University, University of Arizona, and University of New Mexico, and published reports from Rocky Mountain Research Station. We used published, peer-reviewed journal articles, as well as published conference proceedings, reports, theses and dissertations, and book chapters as sources of information. We limited our search to relevant literature that came from studies of Southwest ecosystems, with a geographical emphasis on Arizona, New Mexico, and northern Mexico to ensure compatibility and relevance to Southwest ecosystems. Sometimes, results from studies in Utah, Colorado, California and other states were reported to show similarities or differences among geographic areas.

Dendroecology - Annual growth rings left by trees in living tissue, stumps, snags, logs, and even archeological artifacts such as vigas and latillas of pueblo construction have been analyzed to estimate past and present age classes, seral stages, or community composition (Morgan and others 1994, Cooper 1960, White 1985). Growth rings that have been scarred by fire (fire rings) along with analysis of existing or past age structure have been used to estimate past patterns and processes of several vegetation types (e.g., Romme 1982, Arno and others 1993, Morgan and others 1994). Forest tree rings can also be analyzed to discern climatic variation, forest structure, insect outbreaks, patch dynamics or successional pathways, frequency and severity of fire regimes, and other processes (e.g., Fritts and Swetnam 1989). In most cases, the size of plots used in Southwest studies we cite ranged in size from 25 to 250 acres. In some cases, it may be difficult to parse out and differentiate between confounding factors such as climatic fluctuation, competition, and insect outbreak. Every year, fire, silvicultural practices, and decomposition remove more of the available record.

Paleoecology - Deposits of plant pollen and charcoal in wetland soils and stream sediments, and in packrat middens can be analyzed to estimate even longer records of vegetation presence on the landscape (e.g., Anderson 1993, Allen 2002).

Narrative Descriptions - Several early explorers and historical writers left narrative descriptions of the ecological condition of the landscape as they found it. We chose not to incorporate this information into our HRVs except on rare occasion when general trends were observed by multiple observers and reported in the literature (e.g., Muldavin and others 2002).

Historic Photographs - We conducted an exhaustive search of available historic photographs in order to create the SWFAP photographic database. The goal of compiling this database was to identify photographs that would be useful for describing the HRV of vegetative characteristics and VDDT model states for each PNVT. The details regarding the creation of this database are outlined below.

In order to compile the SWFAP photographic database, archives that stored historical and present day landscape scale photographs of the Southwest were researched (Table 1-3).

Table 1-3. Photographic archive, location of archive, persons contacted, identification of the types of photographs (potential natural vegetation types = PNVTs) obtained from each archive, and additional information regarding the photographs collected. Note that not all photographs researched and collected were incorporated into the final SWFAP photographic database.

Photographic Archive	Location of Archive	Contact Person	Repeat Photographs Collected	PNVTs for which photographs were obtained for	Additional Comments
Apache-Sitgreaves National Forest	Springerville, AZ	Bob Dyson	No	aspen, interior chaparral, mixed conifer, montane grasslands, pinyon-juniper, riparian, spruce-fir	The photographs came from the A-S historic archives, and were sent on a CD. The CD included about 500 photographs, although none of the photographs have information regarding dates taken or the specific locations of the photographs.
Carson National Forest	Taos, NM	Bill Westbury and Dave Johnson	No	aspen, mixed conifer, montane grassland, riparian, spruce-fir	
Coronado National Forest	Tucson, AZ	Bill Gillespie and Geoff Soroka	No	aspen, interior chaparral, Madrean encinal, Madrean pin-oak, mixed conifer, pinyon-juniper, semi-desert grasslands	Two sources were used. One was from Bill Gillespie, and included only historical photos. The other source was from Geoff Soroka, where most photos were taken in part to ground-truth the mid-scale vegetation mapping effort.

Ecological Restoration Institute	Northern Arizona University	Dennis Lund	No	aspen, mixed conifer, pinyon-juniper, ponderosa pine	photos from Dennis's collection from national and local USFS archives
Gila National Forest	Silver City, NM	Reese Lolly	No	interior chaparral, mixed conifer, pinyon-juniper, ponderosa pine	
‘*Historic increases in woody vegetation in Lincoln County, New Mexico*’ by E. Hollis Fuchs	n/a	E. Hollis Fuchs	Yes	mixed conifer, montane grasslands, ponderosa pine, pinyon-juniper, riparian, semi-desert grasslands	Photographs taken directly from Hollis’ book.
Jornada Experimental Range	Las Cruces, NM	n/a	Yes	semi-desert grasslands	photos from on-line archive
Rocky Mountain Research Station	Flagstaff, AZ	Susan Olberding	No	interior chaparral (on-line resource only), ponderosa pine, riparian	includes mostly photographs from the Ft. Valley Research Station archive, but also from the RMRS on-line photographs
Saguaro National Park	Tucson, AZ	James Leckie	No	Madrean encinal, Madrean pine-oak	Photographs from several field season that investigated the effects of fire over several years
Santa Fe National Forest	Santa Fe, NM	Mike Bremer	No	mixed conifer, pinyon-juniper, riparian, spruce-fir	
Santa Rita Experimental Range	southeastern AZ	n/a	Yes	semi-desert grasslands	photos from on-line archive
Sharlot Hall Museum	Prescott, AZ	Ryan Flahive	No	aspen, interior chaparral, mixed conifer, pine-oak, pinyon-juniper, riparian	
The changing mile revisited' by Turner, Webb, Bowers, and Hastings.	Tucson, AZ	Ray Turner and Diane Boyer	Yes	Madrean encinal, riparian, semi-desert grasslands	These photographs were taken directly from this book.
United States Geological Survey	Tucson, AZ	Diane Boyer and Ray Turner	Yes	Madrean encinal, riparian, semi-desert grasslands	From the Desert Laboratory Repeat Photography Collection
United States Geological Survey	Los Alamos, NM	Craig Allen	Yes	pinyon-juniper, ponderosa pine, mixed conifer, spruce-fir	Photographs taken from an unpublished paper by Hogan and Allen (2000).
US Forest Service Region 3	Albuquerque, NM	Sheila Poole	Some	alpine-tundra, aspen, interior chaparral, Madrean encinal, Madrean pine-oak, mixed conifer, montane grasslands, pinyon-juniper, riparian, semi-desert grasslands, spruce-fir	

US Forest Service unpublished report "*Wood plenty, grass good, water none*" by Harley Shaw	n/a	Harley Shaw	Yes	pinyon-juniper, semi-desert grasslands	Photographs taken from Harley's manuscript that will be published in the near future by the RMRS.

Many of these photographic archives included museums and federal agencies like the US Geological Survey, the National Park Service, individual National Forests, USFS Research Stations, and the USFS Regional Office. In addition to traditional photograph archives, other sources of photographs came from published books of repeat photography, unpublished manuscripts of repeat photography, and photographs taken in the field for vegetation mapping purposes or other reasons. Several historical societies and Arizona and New Mexico state agencies were contacted about potential photographs, however, none proved to have photographs that would meet the needs of this project. Our goal was to obtain photographs of each PNVT from a variety of locations, so that one area (or state) was not over-represented, showing a variety of conditions with an emphasis on repeat photography sequences.

When viewing photographic archives, or photographs from the field, we viewed all of the photographs available, and then selected those photographs that we deemed potentially appropriate photographs for this project. The criteria used to make the initial selection of photographs from the archives are outlined below:

- We discarded all photographs where buildings and/or people were the main subject, and one could not see the vegetation well
- We discarded all photographs where the quality of the photo was poor
- We discarded photographs if they were repeating the same subject matter (i.e. two photographs taken at the same time of the same landscape, we would hold on to the 'best' one and discard the other)
- We discarded many photographs that repeated the same subject matter and model state (i.e. if there were 30 photographs of park-like ponderosa pine from roughly the same location and roughly the same dates, we kept approximately the 'top' 5)
- We retained any photographs that were repeats over time
- We retained any photographs of PNVTs that we had a limited number of, or that we had limited numbers for that location (i.e. if we had hundreds of ponderosa pine forest photographs in Arizona but few for New Mexico, we would select the best photographs for Arizona and keep all the ones that were taken in New Mexico)
- We retained any photographs of PNVTs that we thought were good examples of various model states within a PNVT (i.e., open canopy, closed canopy, early seral, late seral)
- We attempted to get as many historical photographs (vs. current day) as possible, although we were limited by availability

After the initial selection of photographs was made, Nature Conservancy ecologists evaluated all photographs for their inclusion into the final SWFAP Photographic Database. Any photograph incorporated into the HRV and state-and-transition model documents were incorporated into the final SWFAP Photographic Database.

The SWFAP Photographic Database uses Extensis Portfolio 7.0 software for Windows to organize and display the selected photographs. Information regarding each photo, including: file name, title, location, date, photographer, if it is linked to a model state in the state-and-transition documents, if it is a repeat of another photograph taken at the same location but different time, copyrights, and source of photograph are included in the database.

Climate Analysis - In Arizona and New Mexico, precipitation is primarily bimodal, highly variable from year to year and from location to location, and has a large impact on vegetation. Extended wet or dry periods can cause changes in vegetation at the life form (grass, shrub, or tree) and/or species composition level (McPherson and Weltzin 1998; Swetnam and Betancourt 1998; Turner and others 2003). The wet period of the late 1970's early 1980's in the southwest has been documented to coincide with the expansion of multiple tree species; wet winters in general tend to coincide with increases in shrub cover, while extended dry periods have coincided with grass, shrub, and tree mortality (Barton and others 2001; Crimmins and Comrie 2004; Grissino-Mayer and Swetnam 2000; Miller and Rose 1999; Savage 1991; Swetnam and Betancourt 1998).

While there is an understanding that climate and, precipitation in particular, play an important role in Southwest vegetation dynamics, little information regarding historical patterns of dry and wet events exists for the Southwest despite multiple regional climate reconstructions (Cook and others 1999; Ni and others 2002). Additionally, the focus of most long-term climate studies, at any scale, is to identify extreme conditions (Cook and others 1999; Cleaveland and Duvick; Laird and others 1996; Meko and others 1995; Ni and others 2002; Salzer and Kipfmueller 2005; Stahle and others 1985; Stahl and Cleaveland 1988). This focus yields little information regarding lower impact events and relies heavily on statistical thresholds, which makes identifying connections with ecological impacts difficult to assess.

Given that there is ecological data to support the idea that both extreme and lower impact (or non-extreme) events can effect Southwest vegetation; the goal of this analysis is to 1) describe historic year to year climate variability, 2) identify the range, frequency, and length of extreme and non-extreme climate events, 3) compare the occurrence of these events spatially throughout the Southwest and temporally across the last 1000 years.

Data - There are two publicly available climate reconstruction data sets that cover the Southwest region for the last 1000 years; a summer (June to August) Palmer Drought Severity Index (PDSI) reconstruction and a winter (November to April) precipitation reconstruction (Cook and others 1999; Ni and others 2002). Both reconstructions correlate tree ring information with climatic information (PDSI or winter precipitation) in order to model past climate values. The nation-wide summer PDSI information covers years 0 to 2003, and is available for 8 grid locations (4 in Arizona and 4 in New Mexico) across the Southwest (Figure 1-1a). We limited our use of this data set to years 1000 to 1988 in order to be able to make comparisons with the winter precipitation data set. The subset of the summer PDSI data utilizes between and 5 and 9 tree chronologies per grid location. The Southwest winter precipitation data covers from years 1000 to 1988, is available for 15 climate divisions (7 in Arizona and 8 in New Mexico) throughout the Southwest, and utilizes 19 tree chronologies (Figure 1-1b). While there are some differences in the two data sets, they both utilize many of the same tree chronologies and, since summer PDSI is partly a measure of the lack of precipitation in late winter/early spring, identify roughly the same climate feature – winter precipitation.

It is important to note some key caveats regarding the data sets. The percent of variation in the cool season precipitation record explained (R2 value) by Ni and others (2002) reconstruction varies for each climate division and should be considered when evaluating results (Table 1-4) (CLIMAS 2005 http://www.ispe.arizona.edu /climas/research / paleoclimate/product.html). Similarly, the Cook and others (1999) reconstructions are based on anywhere from 5 to 9 tree chronologies with less certainty in the reconstruction occurring with fewer chronologies (

Table 1-5). Additionally, information used to build both reconstruction models comes from upper elevation pine species which should be considered when extrapolating these data to lower elevation warm season dominated vegetation types or areas. Even with the above mentioned constraints, these climate data give an unprecedented regional look at historic climate conditions throughout the Southwest.

Table 1-4. Percent of variation in the known cool season precipitation record explained (R2 value) by Ni and others (2002) for all 15 climate divisions in Arizona and New Mexico (CLIMAS 2005 http://www.ispe.arizona.edu/climas/ research/paleoclimate/ product.html).

	Az1	**Az2**	**Az3**	**Az4**	**Az5**	**Az6**	**Az7**	**Nm1**	**Nm2**	**Nm3**	**Nm4**	**Nm5**	**Nm6**	**Nm7**	**Nm8**
R^2 (%)	49	62	48	50	42	51	44	65	59	44	44	41	40	42	36

Table 1-5. Number of tree chronologies used in climate reconstructions for each PDSI grid point location for the Southwest.

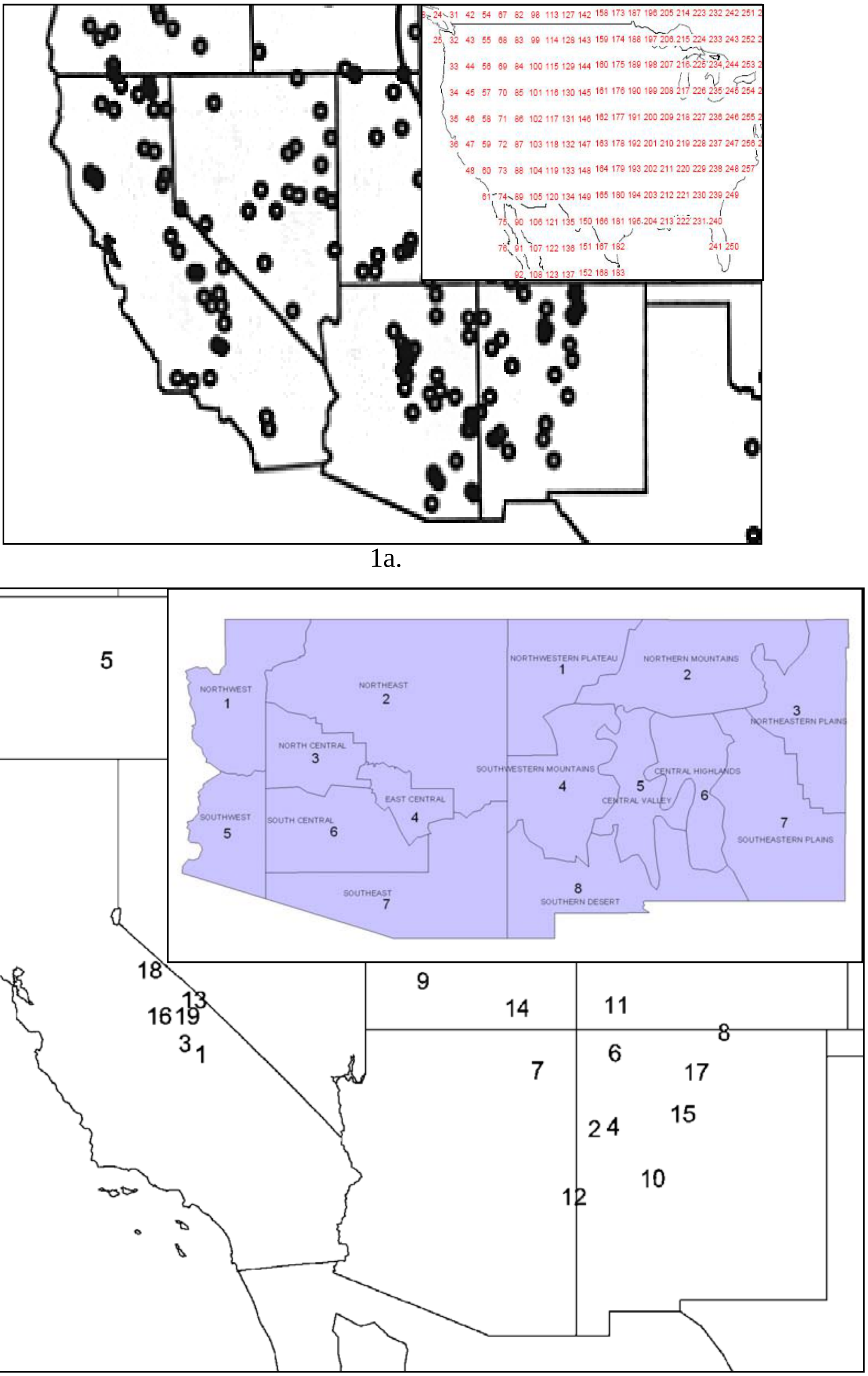
	88	**89**	**104**	**105**	**119**	**120**	**133**	**134**
# of Tree Chronologies	8-9	5-9	8-9	5-9	9	6-9	8-9	5-9

Methods- For a detailed discussion of the methodology used to identify 1) year to year variability, 2) range, frequency, and length of extreme and non-extreme events, and 3) spatial and temporal comparison, see Schussman 2006 (Assessing Low, Moderate, and High Severity Dry and Wet Events Across the Southwestern United States from Year 1000 to 1988).

Results - A comparison of the percent of dry and wet winter precipitation years, for the 15 climate divisions that span Arizona and New Mexico, showed a pattern of 19% of the years, between year 1000 and 1988, classified as severe drought or extremely wet years, 11% classified as drought years, 8% classified as wet years, and 43% classified as normal years (Figure 1-2 and Appendix 1- Table 1.1 and Figures 1.1 to 1.15). The long-term winter precipitation averages for each climate division range from 2.4 to 9.8 inches/yr. Comparisons of the 8 summer PDSI locations showed the pattern of 11% of the years classified as severe and extreme drought, 27 % classified as moderate and mild drought, 38% classified as near normal and incipient wet and dry spells, 20% classified as slightly or moderately wet, and 5% classified as very and extremely wet years (

Table 1-5, Figure 1-3, and Appendix 1 - Table 1.2 and Figures 1.16 to 1.23). Overall there is little regional variability in the percent of dry and wet years for either the winter precipitation or summer PDSI data sets. Of the regional variability that is present, the majority of the variation occurs within the winter precipitation data set between severe drought and drought years. For example, New Mexico climate divisions 2, 3, and 6 had fewer severe drought years than the average, but had higher drought years.

There is also little regional variability in the total number of drought, normal, and wet events that occurred in either the winter precipitation of summer PDSI data sets (Figure 1-4, Figure 1-5, Appendix 2 - Tables 2.1 and 2.2 and Figures 2.1 to 2.23). Specifically, there were on average 52 drought events, 41 wet events, and 85 normal events identified for the winter precipitation data and 71 drought events, 54 wet events, and 104 normal events identified for the summer PDSI data set. In contrast, the range of the length of events does exhibit some regional variability with winter precipitation events ranging between 9 and 26 years for the longest drought events, between 14 and 23 years for the longest wet events, and between 19 and 40 years for the longest normal events. This level of variability is also seen in the summer PDSI data set with between 19 and 25 years for the longest drought event, between 8 and 17 years for the longest wet events, and between 14 and 23 years for the longest normal events (Appendix 2 - Table 2.1 and Figures 2.1 – 2.23). The timing of the events identified is fairly consistent across the entire Southwest (ie all climate divisions and PDSI grid point locations document drought and wet events occurring in roughly the same years even though the magnitude of those events varies regionally).



1a.

1b.

Figure 1-1. Identification of tree chronology locations for both the PDSI (1a taken from Cook and others 1999) and winter precipitation (1b taken from Ni and others 2002) data sets, as well as PDSI grid point locations and climate division boundaries.

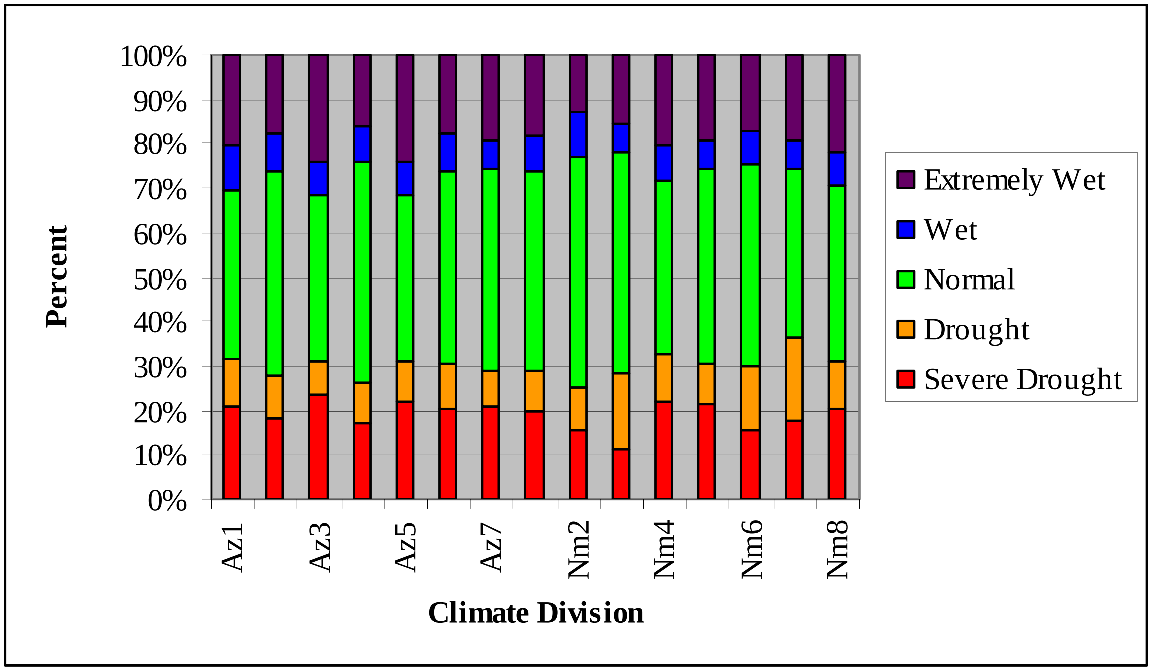


Figure 1-2. Comparison of the percent of years in all year types for all climate divisions in the Southwest.

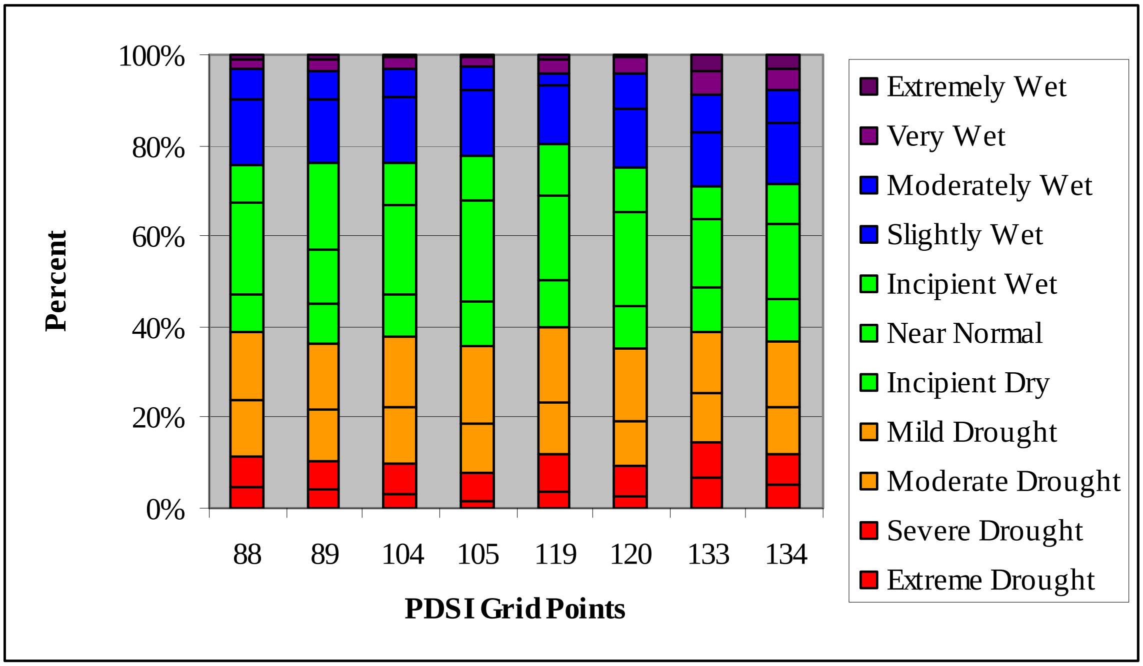


Figure 1-3. Comparison of the percent of years in all year types for all PDSI grid locations in the Southwest.

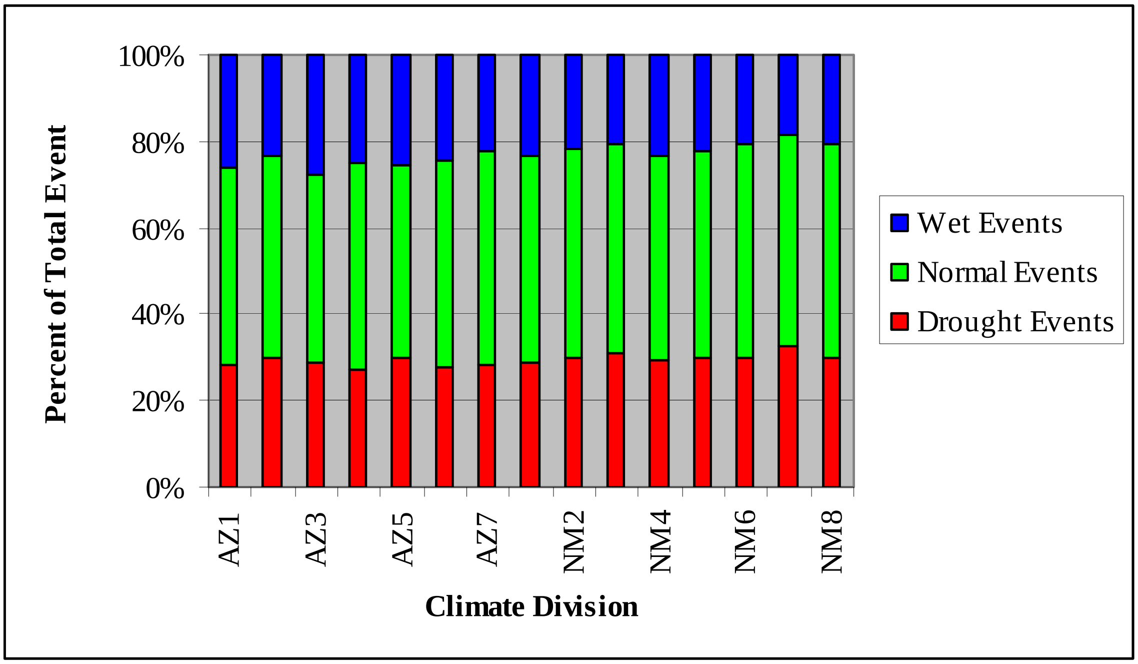


Figure 1-4. Comparison of the percent of events classified as drought, normal, and wet events for all climate divisions in the Southwest.

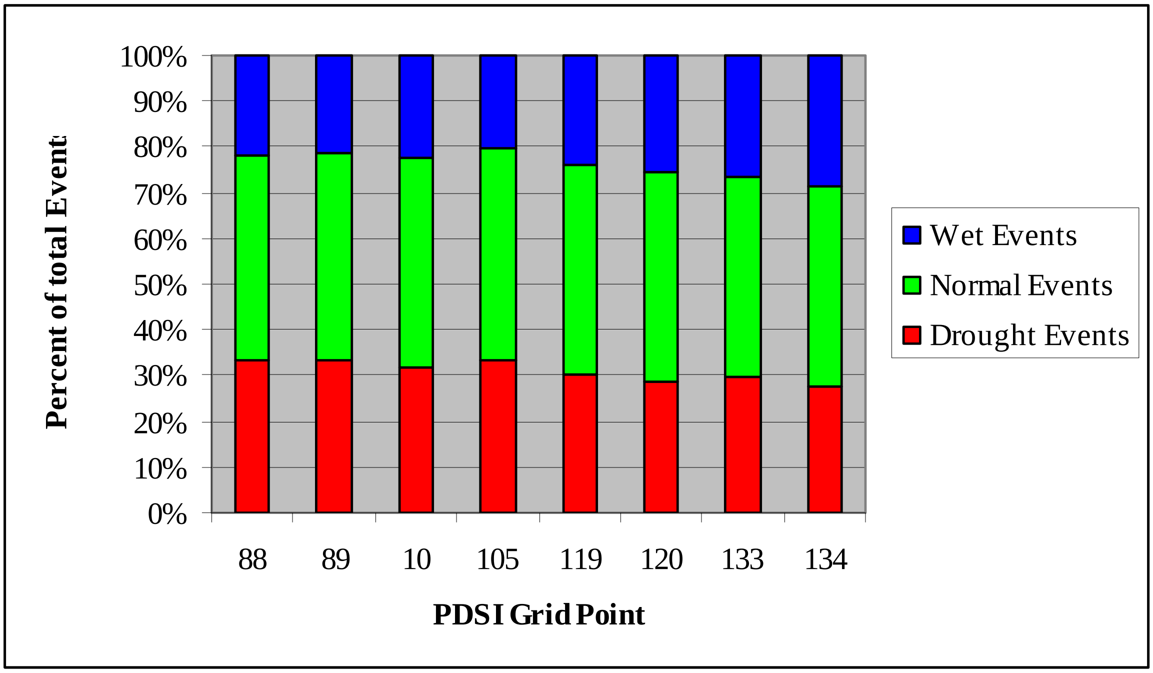


Figure 1-5. Comparison of the percent of events classified as drought, normal, and wet events for all PDSI grid locations in the Southwest.

The high end of the relative drought and wet magnitude ranges varies somewhat throughout the region (Appendix 2 - Table 2.1). Most strikingly, 5 climate divisions (AZ3, AZ6, AZ7, NM7, and NM8) and all PDSI grid points experienced droughts of greater magnitude than the regional 1950's range while 11 climate divisions (AZ2, AZ3, AZ4, AZ6, AZ7, NM3, NM4, NM5, NM6, NM7, and NM8) and all PDSI grid points experienced wet events of greater magnitude than the regional 1980's wet period. Relative drought magnitudes for the winter precipitation data set ranged between -866.5 and -25.4%, wet magnitudes ranged between 1,397.4 and -6.7%, and normal magnitudes ranged between 198.5 and -283.0% of **cumulative deviation from average** with the regional range of the 1950's drought and 1980's wet period having relative magnitudes between -629.0 and -102% and 139 and 634% respectively for all climate divisions. Ranges for summer PDSI relative magnitudes (**cumulative PDSI value**) ranged between -55.7 and -1.9 for drought events, between 28.9 and 2.1 for wet events, and between 10.0 and 6.2 for normal events with the regional range of the 1950's drought and 1980's wet period having relative magnitudes between -34.5 and -9.1 and 6.3 and 11.7 respectively. The amount of variability in the relative magnitude of events throughout the region was quite impressive. For example, for climate division AZ3, the 1950's drought was a fairly low intensity (-102) event for which 29 other drought events were of greater magnitude. However, for climate division NM3, the 1950's drought was the most severe event (-629%) recorded for the last 989 years.

Evaluation of the average years between drought and wet events of all severity levels (high, moderate, and low) showed a consistent pattern of lower severity events occurring more frequently than higher severity events (Appendix 2 - Table 2.2). Specifically, for the winter precipitation data set, low severity drought events occurred on average every 23 to 51 years, moderate events occurred every 18 to 69 years, and high severity events occurred greater than every 100 years (Appendix 2 - Table 2.2). Similarly, the summer PDSI data set showed low severity droughts events occurring every 18 to 26 years, moderate events every 19 to 37 years, and high severity events every 74 to 296 years. For wet events identified in the winter precipitation data low severity events occurred every 26 to 58 years, moderate events occurred every 34 to 65 years, and high severity events occurred every 220 to 838 years. Again summer PDSI events were similar with low severity events occurring every 24 to 47 years, moderate events occurring every 26 to 79 years, and high severity events occurring every 68 to 273 years. In contrast to this pattern, low and high severity normal events occurred less frequently than moderate events with low severity events occurring every 44 to 153 years, high severity events occurring every 50 to 149 years, and moderate events occurring every 7 to 12 years.

Discussion - For both Arizona and New Mexico, most areas have experienced drought and wet events of greater magnitude than the regional range of magnitudes experienced in the 1950's and 1980's. The magnitude and pattern of events in this analysis are in agreement with other climate assessments for the Southwest (Cook and others 1999. Ni and others 2002; Meko and others 1995; Salzer and Kipfmueller 2005; Stahl and others 2000). Specifically, high magnitude and/or persistent drought (1128 to 1160, 1584 to 1592, and 1776 to 1792) or wet conditions (1304 to 1360 and 1904 to 1920) identified in this analysis coincided with warm/dry or cool/wet periods documented for the southern Colorado Plateau, by Salzer and Kipfmueller's (2005). Additionally, the 16th century megadrought has been documented to have coincided with the abandonment of "a dozen" pueblos in New Mexico (Stahle and others 2000).

Comparison of the pattern of dry and wet events for specific climate division with PNVT shows that climate divisions AZ3, AZ6, AZ7, NM7, and NM8 all experienced drought events greater than the regional 1950's drought range. This pattern of higher severity events occurring within southeastern Arizona and southern New Mexico suggests that PNVTS predominantly located within this area (ie the semi-desert grasslands, Madrean pine oak woodland, Madrean encinal, and interior chaparral) historically have a pattern of the highest severity events. This regional pattern is also seen in the PDSI data set where grid point locations 105, 120, and 134 had the lowest magnitude of wet events along with drought magnitudes greater than the regional 1950's range.

The results of both the year to year climate variability (percent of years in a given year type; Figures 1-2 and 1-3) and event variability analysis (Figures 1-4 and 1-5) reveal that dry, wet, and normal years and events, of all magnitudes, are all common historically in the Southwest. For example, a drought event of any magnitude historically occurred on average every 14.5 years while wet events, of any magnitude, occurred on average every 19.4 years. This suggests that managing for an "average" year or period is less advantageous than management practices that are variable and responsive to the continually changing climate conditions that typify the Southwest. Additionally, the knowledge that extreme events, of greater magnitude than we have an ecological understanding of, have occurred in the past suggests that land managers need to be aware of and plan for the possibility of a recurrence of such events.

Finally, while having an understanding of historic climate patterns is helpful, recent research on global climate change suggests that future events may be nothing like those seen historically (Nielson and Drapek 1998; IPPC 2001). Research by Breshears and others (2005) begins to demonstrate the need to look at the change in effect of events given changing climate factors. Given the possible discrepancies between the pattern and/or magnitude of events as well as the effect of future events on vegetation, it is important to use historic climate information as a starting point for understanding trends in vegetation dynamics with the understanding that changing climatic factors as well as variability within the historic record, such as the Little Ice Age, also need to be evaluated (Millar and Woolfenden 1999).

Expert Opinion - We did not utilize expert opinion in developing our HRVs but instead relied on published empirical data. Limitations to expert opinion include lack of rigor, inclusion of bias, lack of repeatability, and limitation of spatial or temporal record (Morgan and others 1994). We did consult with subject experts extensively, however, in helping to identify data sources and reports not available in standard periodicals or journals.

Negative Data or Missing Information - Many pieces of historical information are lacking from the historical record (White and Walker 1997). When information is lacking, rather than not include this information in the HRV, we explicitly state that there is no information on the topic to indicate that we searched for, and were unable to find any relevant studies.

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Chapter 2 - Semi-Desert Grassland

2.1 General Description

Semi-desert grassland occurs throughout southeastern Arizona, southwestern New Mexico, northeastern Sonora, and northwestern Chihuahua at elevations ranging from 3,000 to 4,500 ft (Wright 1980). These grasslands are bounded by Sonoran or Chihuahuan desert at the lowest elevations and woodlands or chaparral at the higher elevations. Species composition and dominance varies across the broad range of soils and topography that occur within the semi-desert grasslands. However, there are some general associations/dominance types that can be identified for Arizona and New Mexico. Black grama (*Bouteloua eriopoda*) dominated grassland is located predominantly in New Mexico at lower precipitation levels (9.0 in), blue grama (*Bouteloua gracilis*) dominated grassland is associated with higher precipitation levels (18 to 20 in) and deep valley bottom soils such as the San Rafael or Animas valley, tobosa (*Hilaria mutica*) dominated grassland is usually located on clay soils and is found throughout the two states, while sacaton (*Sporobolus wrightii*) dominated grassland is located along water courses, requiring high water tables to regenerate and survive. There are also many areas throughout the two states that demonstrate a mix of native perennial grasses including *Aristida* sp., sideoats grama (*Bouteloua curtipendula*), spruce top grama (*Bouteloua chondrosioides*), black grama, blue grama, curly mesquite (*Hilaria belangeri*), and *Muhlenbergia* sp., as well as areas dominated by non-native perennial grasses such as Lehmann lovegrass (*Eragrostis lehmanniana*) and Boer lovegrass (*Eragrostis curvula var. conferta*). Boer lovegrass is limited to cooler and wetter locations whereas Lehmann lovegrass dominates on sandy soils in drier locations (Robinett pers. Comm.). Shrubs also occupy these grasslands and their abundance and species composition also varies with soil characteristics, elevation, occurrence of fire and climatic factors. The predominant shrubs include mesquite (*Prosopis glandulosa* and *Prosopis velutina*), broom snakeweed (*Gutierrezia sarothea*), burroweed (*Isocoma tenuisecta*), creosote bush (*Larea tridentata*), waite-a-minute bush (*Mimosa biuncifera*), and cat claw acacia (*Mimosa dysocarpa*).

2.2 Historic Range of Variation of Ecological Processes

Vegetation Dynamics – The semi-desert grasslands within Arizona and New Mexico vary greatly based on soils, topography, and precipitation and hence are challenging to describe as a single unit. Given the level of complexity within this PNVT we have identified 3 main disturbance/soil regimes, for which there is empirical information, that exist and reinforce the main grassland types (mixed native grasslands, valley bottom grasslands (dominated by blue grama or tobosa grass), and black grama dominated grasslands). Below is a discussion of vegetation dynamics specific to each grassland type.

Mixed native grasslands are the dominant grassland type within the uplands of Arizona and have been shown to trend from open grasslands with low shrub canopy cover (less than 10% = state) towards higher shrub cover and ultimately to convert (> 35% total canopy cover and > 15% mesquite or juniper cover) to shrublands without frequent fire (Gori and Enquist 2003). While we know frequent fires, on the order of every 2.5 to 10 years, to have historically maintained these grasslands in an open, shrub-free state, it is unclear exactly how many missed fire cycles will generate shrub conversion or how drought and livestock grazing interact and affect the rate of shrub increase (Brown and others 1997; Cable 1971; McPherson 1995; Robinett 1994; Thornber 1907 in Humphrey 1949; Wright 1980). Wet winters have been correlated with increases in woody species density and cover; hence prolonged wet periods also act to increase shrub density and cover of the dominant shrub species (mesquite, juniper, creosote, and burroweed) (Barton and others 2001; Grissino-Mayer and Swetnam 2000; Miller and Rose 1999; Savage 1991; Swetnam and Betancourt 1998). Shrubland conversion occurs when total shrub canopy cover gets above 35% (or mesquite or juniper cover >15%) and results in the loss of perennial grasses which increases the amount of bareground exposed to wind and water (Gori and Enquist 2003; Whitford 2002). Increases in soil exposure can result in losses of topsoil and A horizons, ultimately making it difficult for grasses to re-colonize a site even if shrub cover is decreased. However, the amount of erosional loss varies by soil type and location and, while loss of soil transforms some areas into shrublands, areas where erosion is less of a factor (ie cobble protected uplands) and water infiltration occurs at sufficient depths to promote shrub growth, fire is key for maintaining these low shrub grasslands (McAuliffe 1995).

Some valley bottom, or basin floor, grasslands with deep argillic horizons, isolated within both states (San Rafael valley in Arizona and Animas valley in New Mexico), have not shown shrub or tree encroachment and/or conversion in the absence of fire or presence of livestock grazing (McAuliffe 1995; Muldavin and others 2002). These deep soil systems have maintained open grassland characteristics despite fire suppression, drought, and livestock grazing due to the maintenance of soils that prevent shrub and tree establishment (McAuliffe 1995). However, there are other valley bottom areas that once supported grasslands, such as the San Simon valley, that have been converted to shrublands due to soil erosion. It is unclear exactly what mechanisms are responsible for the resilience seen in some areas and not in others, however, higher average precipitation in the San Rafael and Animas valleys may be one factor.

Black grama dominated grasslands within New Mexico, usually located on sandy soils and receiving less than 250 mm of precipitation (Bestelmeyer pers. Comm.), have been shown to trend towards shrublands over the last 100 years (Buffington and Herbel 1965; Gibbens and others 2005). It is unclear if the loss of grass and replacement by shrub species (primarily mesquite and creosote bush) is due to the absence or presence of fire or due to grazing and/or drought stress. In contrast to the mixed native type where shrub cover increases are primarily tied to a lack of fire events, shrub increase within black grama dominated grasslands have been seen following disturbances that have caused grass cover to drop, allowing shrub seedling establishment and soil erosion to occur (Whitford 2002). Disturbances such as drought, fire, and livestock grazing have all been shown to decrease black grama cover as well as cause mortality within this perennial grass (Buffington and Herbel 1965; Drewa and Havstad 2001; Gosz and Gosz 1996; Reynolds and Bohning 1956). The recent (last 120 years) spread of mesquite has been tied to increased seed dispersal by livestock as well as a sharp decline in mesquite use by Native Americans due to their declining population size (Frederickson and others 2006). As with vegetation dynamics within the mixed native type, areas converted to shrublands or dunelands are difficult to move back into grassland states as scattered nutrients and high erosion rates characteristic of the former reinforce a shrub/duneland system (Whitford 2002).

Disturbance Processes and Regimes

Below is a discussion of the frequency, intensity, severity, seasonality, and spatial and temporal scale of disturbances that occur within the semi-desert grassland vegetation type.

Climate – See Chapter 1, climate analysis section.

Fire - It is documented, through direct and indirect evidence, that fire played a key role in semi-desert grasslands in southeastern Arizona before 1890 (Bahre 1985; Humphrey 1952; Kaib and others 1996; McPherson 1995; Wright 1980). Specifically, stand replacing fires swept across the grasslands **between June and July** at intervals of every 2.5 to 10 years and covered hundreds of square miles at a time (Kaib and others 1996; McPherson 1995; Bahre 1985). While Native American fire use has been documented to have contributed to fires in these grasslands, natural ignitions (lightning) account for the majority of fires (Swetnam and Baisan 1996). The southwest has a pattern of cool season moisture leading into an arid foresummer followed by pre-monsoonal lightening storms which are primarily responsible for consistently producing fires in June and July (Swetnam and Betancourt 1990).

Direct lines of evidence include fire scar data from canyons connected by semi-desert grasslands along with historic (1859 to 1889) newspaper accounts of grassland fires. Fire scar analysis conducted by Kaib and others (1996) found that the mean fire interval for all trees scarred in the paired canyons was 2.5 years, mean fire interval for more than two trees scared was 8.0 years. Historical accounts corroborate frequent fires in semi-desert grasslands as well as the foresummer timing of fires and their large size. For example, one of the early (1874) news paper accounts of a fire burning outside of Tucson stated,

> Fires have been raging south and southeast of here during the past week. Millions of acres of excellent grass land have been burned over but thanks to the abundance of our grazing lands we have plenty left. As soon as the rainy season sets in, which will be about the first of next month, the whole country will again be covered with green grass (Bahre 1985)

In total, Bahre (1985) found 13 mentions of southeastern Arizona grassland fires in local newspapers between 1859 and 1889. Of those 13 reported grassland fires, 9 occurred in during the foresummer (between May 28th and June 23rd), 1 occurred after the start of the monsoon (September 27th), and 3 occurred in the early spring (between March 17th and April 16th) with one of the 3 reported to have been set by the Apache Indians (Bahre 1985).

Indirect lines of evidence, such as fire ecology of dominant species and vegetation changes over the last 115 years, support direct lines of evidence. In fact, frequent fire was identified as essential for limiting the growth and expansion of shrubs and maintaining a grassland's open character as early as 1907 (Thornber 1907 in Humphrey 1949) and has continued to be recognized throughout the last hundred years (Cable 1971; McPherson 1995; Robinett 1994; Wright 1980). Indeed, many researchers have demonstrated the effect of fire in reducing shrub cover and increasing perennial grasses in southeastern Arizona grasslands (Bock and Bock 1992; Robinett 1994; Uchytil 1988; Gori and Backer, in press). In addition, fire ecology of the dominant shrubs in semi-desert grasslands concurs with the observations that frequent fire is needed to maintain shrub free grasslands as most semi-desert grassland shrubs are easily killed by fire, at least as seedlings or young plants, and do not produce seeds until they are at least 10 years of age

(McPherson 1995). Specifically, many researchers have found that mesquite, a common shrub increasing within the semi-desert grasslands, is easily killed when its diameter is below 2”, however, after mesquite reaches larger diameters fire becomes less effective at eliminating the plant (Cable 1965; Reynolds and Bohning 1956). In addition, many studies have documented large reductions in the cover of many common semi-desert grassland shrubs such as, broom snakeweed, burroweed, and cacti (Bock and Bock 1997; Humphrey and Everson 1951; Reynolds and Bohning 1956) following fire events. Specifically, Reynolds & Bohning (1956) found that a hot June fire killed 9 % of mesquite, 28 % to 67 % of cacti species, and 88 % of burroweed.

Along with a documented reduction in shrub cover, studies have also shown fire to have little negative effect on most perennial grasses, with recovery happening 1-2 growing seasons after a fire (Bock and Bock 1992; Gosz and Gosz 1996; Cable 1972; Martin 1983; Wright 1980). Drought conditions extended this recovery time to 3-4 growing seasons post fire, but ultimately showed fire to have no negative effects on the grasses themselves except for black grama (Cable 1965; Reynolds and Bohning 1956; Valone and Kelt 1999; Wright 1980).

The role of fire in New Mexico’s black grama dominated grasslands is unclear, as studies of historical records do not document fires in these grasslands (Branscomb 1956 in Buffington and Herbal 1965; Buffington and Herbal 1965; Wright 1960). In addition, in contrast to grasslands in Arizona where fire has been shown to have null to positive effects on perennial grass cover and a negative effect on shrub cover, fire has been shown to decrease black grama cover (Buffington and Herbel 1965; Drewa and Havstad 2001; Gosz and Gosz 1996; Reynolds and Bohning 1956) and have no effect on *Gutierrezia sarothrae* (Drewa and Havstad 2001) in times of drought and do not kill mesquite (Drewa 2003). Similarly, several New Mexico studies have shown that black grama decreases with other disturbances, such as drought, livestock grazing, and clipping, recovering slowly if at all after such events (Buffington and Herbel 1965; Drewa and Havstad 2001; Gibbens and Beck 1988; Gibbens and others 2005; Gosz and Gosz 1996; Whitford and others 1999). While drought was a conflicting factor in many of these studies, it is important to note that studies in Arizona were also conducted during times of drought and resulted in longer recovery times not a lack of recovery in perennial grasses.

The recent historical data, along with fire scar data from the Chiricahua mountains, and information regarding the fire ecology of the dominant plant species all support a very frequent historical fire regime for the semi-desert grasslands of southeastern Arizona. In contrast, information regarding the negative response of black grama to fire coupled with a lack of historical fire occurrence accounts suggest that black grama dominated grasslands may have had a less frequent fire regime. However, more research needs to be carried out to determine fire’s effect both with and without grazing and drought stress.

Hydrology – We found one study of arroyo formation in southern Arizona that documented hydrologic changes within the semi-desert grassland vegetation type (Cooke and Reeves (1976). Additional information on erosion is covered in the *Erosion* section below.

The formation of arroyos along valley bottoms in southern Arizona occurred between 1865 and 1915, since this time arroyos have become deeper, wider, and longer (Cooke and Reeves 1976). The change from broad flat valley bottom drainages to incised arroyo channels altered the hydrology of many semi-desert grassland systems resulting in lower water tables and decreased water availability for vegetation. Consequences of these changes included loss of fertile land, increased sediment movement, altered hydrologic relationships, changes from lush grass and riparian vegetation to more xeric vegetation, and effects on settlements (Cooke and Reeves 1976). There is great debate over the causes of arroyo formation with causes ranging from climatic changes, land use, changes in vegetation, and livestock grazing. Cooke and Reeves' (1976) comprehensive analysis of arroyo formation in southern Arizona sheds light on key factors associated with these hydrologic changes as well as some over all patterns. First, they note that arroyo formation occurred in many but not all valley floors, second, even in areas where arroyo formation had occurred, there was not consistent entrenchment along the entire valley floor, instead entrenchment was intermittent. Most importantly, Cooke and Reeves (1976) determined that there is strong evidence that valley floor changes, "such as the cutting of ditches and canals, the creation of roads, and the building of embankments" were key in initiating arroyo formation. Additionally, they did not find evidence that climatic changes over the last 100 years, nor overgrazing were key factors in arroyo formation, but that both may have added to the problem. Finally, there is some evidence to suggest that arroyo formation is not isolated to the late nineteenth century, and that arroyos have formed and then filled at some unknown frequency and due to unknown causes (Cooke and Reeves 1976).

Herbivory - Native herbivores in the semi-desert grasslands range from insects and rodents to pronghorn and deer (Finch 2004). Historically, pronghorn (*Antilocarpa Americana*) ranged across all of North America's grasslands (Berger 2004). In Arizona, pronghorn currently inhabit 20,077 mi^2 of grasslands in the northern, central, and portions of the southeastern parts of the state, having their greatest presence in the northern part of the state. Historically, pronghorn were present in much of the state, but by 1922 were extirpated from many of its grasslands (Ockenfels and others 2000). Pronghorn were historically abundant herbivores on the landscape in Arizona and New Mexico, however there is no information available to indicate what level of impact these animals had on vegetation structure or composition. There is, however, information to suggest their effects are different than livestock. A habitat management guide produced for pronghorn in northwestern America in 1980 emphasized that pronghorn utilize less than 1 % of the available range forage resource (Neff 1986). In addition, studies in west Texas and eastern New Mexico showed that "it took 38 pronghorn to eat as much cattle forage as 1 cow" (Neff 1986).

Rodents, such as prairie dogs and kangaroo rats have both been identified as vegetation modifiers in the semi-desert grassland system (Finch 2004; Miller and others 1994). As vegetation modifiers, prairie dogs have been shown to alter nutrient cycling, increase plant and animal diversity, and decrease mesquite seedling establishment through seed pod and seedling herbivory, while kangaroo rats have been shown to modify soil structure and water infiltration and alter vegetation structure and composition (Brown and Hesky 1990; Finch 2004; Miller and others 1994: Weltzin and others 1997). Specifically, Brown and Hesky (1990) showed that the removal of kangaroo rats caused a shift from desert shrubland to grassland resulting from the increased establishment of annual grass along with non-native Lehmann lovegrass. Currently, kangaroo rats along with a host of other rodents still thrive in semi-desert grasslands in Arizona and New Mexico (Moroka and others 1982). The prairie dog does not.

In the early 1900's prairie dog species occupied between 40 and 100 million ha of grassland in western North America, by the 1960's the area they occupied had been reduced, by 98 %, to 1,482,630 ac (Miller and others 1994). This drastic reduction in prairie dogs occupation was due in large part to a western-states-wide poisoning campaign based on the inaccurate idea that prairie dogs and cattle competed strongly for the same resources (Finch 2004; McPherson 1997; Miller and others 1994). The black tailed prairie dog, which is native to the semi-desert grasslands, was extirpated from Arizona by 1960 and its range was reduced by 25 % in New Mexico where they were common on basin floor soils (Finch 2004; Bestelmeyer pers. Comm.). Despite research that suggests a low level (4 to 7 %) of competition between prairie dogs and livestock for forage resources as well as the preference of cattle to graze near prairie dog colonies due to higher palatability of forage, the prairie dog is still seen as a pest in many areas and receives little legal protection (Miller and others 1994).

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and/or introductions as important ecological determinants for the semi-desert grassland vegetation type.

Insects and Pathogens - We found no studies that documented historic insect or pathogen disturbances within the semi-desert grassland vegetation type. However, current research on invertebrates may be useful in understanding current conditions and possible effects of historic disturbance events.

Invertebrate species numbers are extremely high in desert grasslands, with species numbers ranging in "the thousands to tens of thousands", and include single celled protozoans, bacterial and fungal feeding nematodes, soil mites, arachnids, millipedes, cockroaches, crickets, grasshoppers, ants, beetles, butterflies, flies, bees, wasps, and true bugs (Whitford and others 1995). While it is understood that both above and below ground invertebrates are important elements of this grassland system and are critical in nutrient cycling, little research has been done on many of these species. However, given their abundance and large size, ants, subterranean termites, and grasshoppers are fairly well studied and hence they will be discussed below in more detail (Other sources; Whitford and others 1995).

Both ants and termites are known for their ability to alter nutrient cycling. In particular, Whitford (1991) determined that termites consumed "50 % or more of all photosynthetically fixed carbon" (Whitford and others 1995). Ants, on the other hands, increase the heterogeneity of nutrients and microtopography yielding areas of with higher productivity and compositional differences (Whitford and others 1995). Additionally, ants cultivate the soil by transporting new soil to the surface; in a study of a variety of soil and grass types at the Jornada Experimental Range, ants were shown to move anywhere from 0.1 kg/ha (Alkali sacaton swale on clay loam soil) to 85.8 kg/ha (Black grama –mesa dropseed on sandy loam soil) of soil while cleaning and constructing nests (Whitford and others 1995). The role of grasshoppers within grasslands is less clear. Bock and others (1992) investigated the effects of increased grasshopper density on vegetative cover and species composition and found no difference when compared to controls over a 4 year period.

Finally, there is some information regarding the distribution and abundance of ants and grasshoppers within the desert grasslands. In general, ants and grasshoppers increase in abundance when food and habitat are optimal (Whitford and others 1995). Specifically, grasshoppers have been shown to decrease in abundance following wildland fires, but return to pre-burn levels 2 years after the event. Additionally, grasshopper species composition shifts in response to fire and grazing disturbances, with species that prefer open ground or herbs increasing with such disturbances (Bock and Bock 1991; Jepson-Innes and Bock 1989). On the other hand, in a study by Whitford and others (1999), ant "community composition, relative abundances of species, and species richness" did not change in response to livestock grazing at various intensities or vegetation removal by herbicide and mechanical treatments. However, there were decreases in large seed harvesting ants, *Pogonomyrmex* spp., in response to dominance of a site by Lehmann lovegrass (Whitford and others 1999). Additionally, Valone and others (1994) found two southern Arizona ant species, *Pheidole xerophila* and *Pogonomyrmex desertorum*, to be sensitive to the removal of rodents within the grassland sites, with *Pheidole xerophila* showing an increase in foraging workers and *Pogonomyrmex desertorum* showing a decrease in colony numbers.

Nutrient Cycling - We found no studies that documented historic nutrient cycling within the semi-desert grassland vegetation type. However, current research on this process may be useful in understanding current conditions and possible effects of historic disturbance events.

In regards to nutrient cycling and availability within the semi-desert grasslands, much work has been carried out on the Jornada Experimental Range (JER) in New Mexico, while a few studies have occurred at Sevilleta Wildlife Refuge in New Mexico and at Fort Huachuca in southeastern Arizona (Bestelmeyer and others 2003; Connin and others 1997; Corkidi and others 2002; Gallardo and Schlesinger 1995; Herman and others 1995; Kieft and others 1998; Parsons 2003; Reynolds and others 1999; Schlesinger and others 2000; Snyder and others 2002; Whitford and Kay 1999; Wilson and others 2005). Most of this work is focused on comparing nutrient and erosion patterns in New Mexico grasslands and shrublands, in an effort to understand the effects of the last centuries large scale shift from semi-desert grassland to shrubland. A key finding common to many of these studies is that nutrients are spatially distributed under plants, which results in a heterogeneous soil resource distribution in shrub dominated areas and a more even distribution within grasslands (Connin and others 1997; Herman and others 1995; Kieft and others 1998; Parsons 2003; Reynolds and others 1999; Schlesinger and others 1996). This difference in the pattern of resource distribution between shrublands and grasslands is due to the difference in rooting depth, plant distribution, and associated loss of nutrients via wind and water erosion. In shrublands, plants have deeper rooting depths which allows for the translocation of nutrients deeper into the soil profile than in grasslands (Connin and others 1997). Additionally, plants within a shrubland are spaced further apart than those in grassland systems which allows for greater loss of soil and nutrients, from wind and water erosion, than in grassland communities (Connin and others 1997; Herman and others 1995; Kieft and others 1998; Parsons 2003; Reynolds and others 1999; Schlesinger and others 1996).

Within this general resource distribution pattern, nutrient availability can still vary. Whitford and Kay (1999) determined that small mammals increased heterogeneity of resources in desert grasslands through the creation of holes that accumulate litter and allow for greater water infiltration resulting in a patch work of high nutrient areas. Similarly, Snyder and others (2002) and Bestelmeyer and others (2003) determined that ants play a key role in the movement of resources within a grassland system and ultimately increase nutrient heterogeneity. Microbial studies suggest that bacteria presence and abundance is correlated to nutrients (Herman and others 1995). Additionally, microbial activity and nutrient cycling can vary based on the availability of carbon and nitrogen within the soil (Gallardo and Schlesinger 1995). An experimental study on the JER showed an increase in microbial biomass following carbon fertilization within shrublands (creosote, mesquite and tarbush) but not within grasslands and biomass increases following nitrogen fertilization only within mesquite shrublands and grasslands (Gallardo and Schlesinger 1995). Disturbance processes also effect nutrient accumulation. A comparison of nutrient accumulation within a recently burned (within 20 years) and less recently burned (greater than 50 years) mesquite grassland, in Fort Huachuca Arizona, identified different nutrient patterns. Nutrients were found to be more localized under mesquite trees where fire occurred more than 50 years ago, where as nutrients were more diffuse on the site that had been burned recently (Wilson and others 2005).

Windthrow - Not an applicable category for a grassland system

Avalanche - Not an applicable category for a grassland system

Erosion – We found no studies that documented the historic process of erosion within the semi-desert grassland vegetation type. However, current research from the Jornada Experimental Station on this process may be useful in understanding current conditions and possible effects of historic disturbance events.

Results of studies from the Jornada Experimental Range (JER) show that erosion due to wind and water is negatively correlated with the amount of vegetative or protective soil cover (Devine and others 1998; Gibbens and others 1983; Hupy 2004; Parsons and others 2003; Nash and others 2003; Neave and Abrahams 2001; Parsons and others 2003; Wainwright and others 2002). A field experiment comparing runoff and sediment transport in tobossa dominated areas versus burrograss dominated areas revealed that the higher cover values (69.4 % +/- 4.7 % in the spring of 1986 and 76.1 % +/- 3.3 % in the fall 1986 for tobossa and 31.6 % +/- 5.7 % in the spring of 1986 and 65.0 % +/- 6.4 % in the fall 1986 for burrograss) and subsequently lower bareground values (10.0 % +/- 3.7 % in the spring of 1986 and 2.8 % +/- 1.2 % in the fall 1986 for tobossa and 63.0 % +/- 5.4 % in the spring of 1986 and 29.2 % +/- 6.1 % in the fall 1986 for burrograss) associated with tobossa grass decreased runoff and sediment loss by more than half in both spring and fall water runoff trials (Devine and others 1998). Similarly, a study of rodent impact on soil erosion processes by Neave and Abrahams (2001) also identified the importance of cover in reducing water runoff and rodent activity in increasing sediment movement. Specifically, they found that intact grasslands and shrublands had lower rates of water runoff (1.32 and 1.02 $cm^3/s/cm^2$ respectively) than degraded grasslands and shrub interspaces which had similarly high rates (2.34 and 2.37 $cm^3/s/cm^2$ respectively) (Neave and Abrahams 2001). Additionally, they found that the highest amounts of sediment transport were coming from open areas that had been disturbed by small mammals (Neave and Abrahams 2001).

Along with vegetative cover, Hupy (2004) documented the importance of any type of surface protection, such as soil crusts, gravel, or vegetative cover, on decreasing wind generated erosion. Specifically, Hupy (2004) found that the highest amounts of dust came from mesquite dunes with similarly lower dust amounts collected from surfaces with weakly developed desert pavements or forb/grass cover. The amount of dust collected varied by height of collection (between 5 cm and 100 cm above ground) and type of site (coppice dune, forb/grass cover, pavements); the greatest amount of dust was collected at the 5 cm height (Between 16 and 1 grams depending on site) while relatively small amounts were collected at the 100 cm height (between slightly over 0 and 1 gram).

Another key result of JER erosion studies is that erosion is a dynamic process that changes with conditions and over time (Gibbens and others 1983; Wainwright and others 2002). For example, Gibbens and others (1983) looked at the change in soil levels due to wind and water erosion in mesquite duneland and duneland/grassland sites between 1935 and 1980 and found that on large mesquite dunelands there was a maximum gain of 86.9 cm, a maximum loss of 64.6 cm with an overall gain of 1.9 cm across the 259 ha site. On another 259 ha mixed mesquite duneland/grassland site they found a 4.6 cm net loss of soil and transition to complete duneland type by 1980 (Gibbens and others 1983). Similarly, Wainwright and others (2002) describe a dynamic erosion process within rills on the Jornada bajada. The build up of sediment leads to the creation of a water and nutrient rich “bead” within a rill. These beads are subject to erosion under large precipitation events, hypothesized to occur every 30 years (Wainwright and others 2002).

In addition to highlighting the dynamic nature of erosion, Wainwright and others (2002) study also points out the connection between erosion processes and nutrient availability. Specifically, they found that the bajada “beads” created a place for water and nutrients to collect (% carbon, hydrogen, and nitrogen within bead was 0.79 % +/- 1.01, 0.24 % +/- 0.17, and 0.07 % +/- 0.1 compared to 0.32 % +/- 0.16, 0.18 % +/- 0.03, and 0.03 % +/- 0.01 outside the bead) and subsequently were refuges for perennial grasses in a sea of creosote bush degraded grassland (Wainwright and others 2002). Other studies have shown links between factors associated with erosion and nutrient availability (Nash and others 2003; Neave and Abrahams 2001). Specifically, factors that decrease soil movement and increase water infiltration, such as vegetative cover and microtopography, also increase nutrient capture within the semi-desert grassland (Nash and others 2003; Neave and Abrahams 2001).

2.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation **–** Forty eight historic photographs and accompanying annotations, taken between 1880 and 1905 within the semi-desert grassland vegetation type, were analyzed for vegetation condition and species composition. Photographic information came from 1) Jornada Experimental Range - Las Cruces, New Mexico; 2) Lincoln county, New Mexico; 3) the Santa Rita Experimental Range, Arizona; and 4) southeastern Arizona. While these data do not give a range of historic values for vegetation characteristics over time, by synthesizing information from multiple locations we can get some idea of the range of vegetation conditions and species that existed around the turn of the century. Below is a summary of information collected from each of the four locations based on all photographs available near pre-European settlement (1880 to 1905 time period) times. All photographs and information for the following sections comes from the SWFAP photographic database, for a discussion of the methodology behind the creation of this database see chapter 1, *Methods Used in Determining HRV* section.

Overstory –

1) **Jornada Experimental Range, Las Cruces, New Mexico (http://usda-ars.nmsu.edu/general/historicalphotos.htm)**

General description of photographs:
Photographs show open grassland valleys with scattered shrubs on hillsides. There were not shrub or grass species identified in the photograph annotations. Information comes from 2 photographs taken circa 1890 (Figure 2-).

Shrub species:
Not mentioned by name

Perennial grass species:
Not mentioned by name

2) **Lincoln County, New Mexico (Fuchs 2002)**

General description of photographs:
Photographs show open grassland valleys with scattered to moderate shrub cover on hillsides and in drainages. A list of shrub and grass species identified in photographs is listed below. Information comes from 3 photographs taken in 1899 (Figure 2-).

Shrub species:
Bigelow sage (*Artemisia bigelovii*)
Cholla (*Opuntia* sp.)
One-seed juniper (*Juniperus monosperma*)
Skunkbush sumac (*Rhus trilobata*)
Wavy leaf oak (*Quercus undulata*)
Yucca (*Yucca* sp.)

Perennial grass species:
Not referred to by name

3) **Santa Rita Experimental Range, southeastern Arizona (http://ag.arizona.edu/SRER/photos.html)**

General description of photos:
Photographs show open grassland valleys with scattered shrubs on hillsides and moderate to dense shrub cover in drainages and washes. A list of shrub and grass species identified in photographs is listed below. Information comes from 15 photographs taken between 1902 and 1905 (Figure 2-).

Shrub species list:
Catclaw acacia (*Acacia gregii*)
Condalia (*Condalia sp.*)
Mesquite (*Prosopis sp.*)
Palo Verde (*Cercidium sp.*)

Perennial grass species:
Not referred to by name

Annual grass species:
Needle grama grass (*Bouteloua aristidoides*)
Annual aristida (*Aristida americana*)

4) Southeastern Arizona (Turner and others 2003)

General description of photos:
Photographs show open grassland valleys with scattered to moderate shrub cover on hillsides and moderate to dense shrub cover in drainages and washes. A list of shrub and grass species identified in photographs is listed below. Information comes from 28 photographs taken between 1880 and 1892 (Figure 2-).

Shrub species:
Agave (*Agave* sp.)
Arizona rosewood (*Vauquelinia californica*)
Arizona white oak (*Quercus arizonica*)
Bear grass (*Nolina microcarpa*)
Blue yucca (*Yucca bacata var. brevifolia*)
Burrobrush (*Hymenoclea monogyra*)
Chamiso (*Atriplex canescens*)
Cottonwood (*Populus fremontii*)
Desert willow (*Chilopsis linearis*)
Emory oak (*Quercus emoryii*)
Fairyduster (*Calliandra eriophylla*)
Gray thorn (*Condalia lycioides*)
Little leaf sumac (*Rhus microphylla*)
Mexican blue oak (*Quercus oblongifolia*)
Morotonia (*Morotonia scabrella*)
Mesquite (*Prosopis velutina*)
Mexican tea (*Ephedra trifurcata*)
Netleaf hackberry (*Celtis reticulata*)
Ocotillo (*Fouquieria splendens*)
Palmillas (*Yucca elata*)
Soapberry (*Sapindus saponaria*)
Sotol (*Dasylirion wheeleri*)
Whitethorn (*Acacia constricta*)
Velvet ash (*Fraxinus pennsylvanica*)
Yucca (*Yucca sp.*)

Perennial grass species:
Arizona cotton top (*Digitaria californica*)
Cane beardgrass (*Bothriochloa barbinodis*)
Grama grass (*Bouteloua* sp.)
Sacaton (*Sporobolus airoides*)
Tobosa (*Hilaria mutica*)

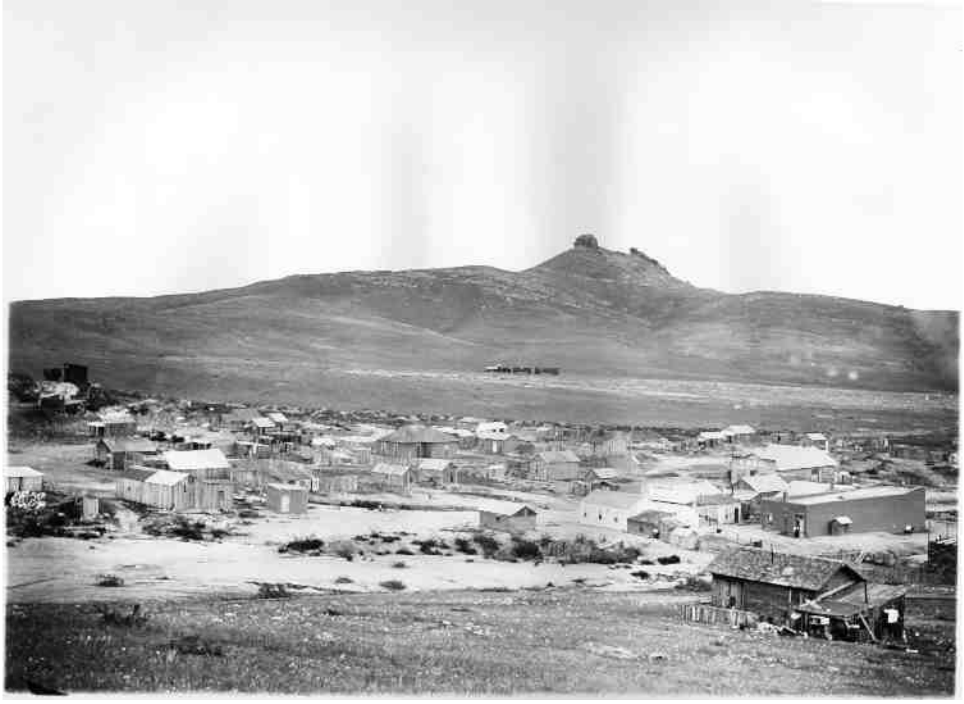

Figure 2-1. 1890's grassland photos taken near Lake Valley, New Mexico. Both photographs depict low shrub cover grasslands (Photographs courtesy Jornada Experimental Range).

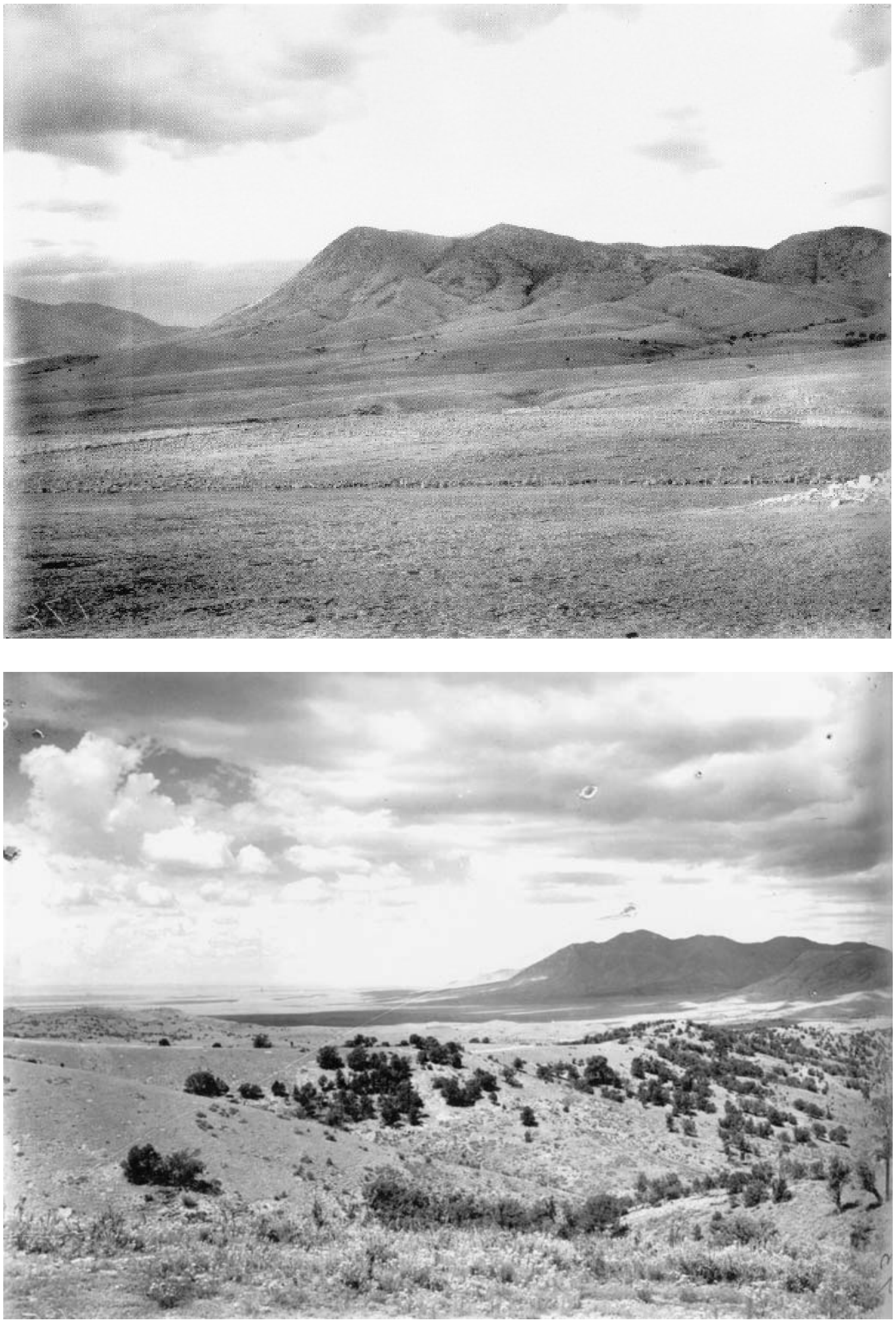

Figure 2-2. 1899 photographs of grasslands in Lincoln county showing open low shrub cover valleys with increasing shrubs and one-seed juniper on hillsides and drainages (Photographs courtesy of United States Geological Survey and Hollis Fuchs 2002)

Figure 2-3. 1902 photographs from the Santa Rita Experimental Range depicting low shrub cover grasslands with shrubs, particularly mesquite, localized to drainages (Photographs courtesy of the Santa Rita Experimental Range).

Figure 2-4. 1895 photograph of Red Rock Canyon east of Patagonia Arizona (top) and 1890 photograph of Guevavi Canyon (bottom) depicting grasslands with low shrub cover except on hill slope drainages. Additionally, bottom photograph depicts short cropped grass and exposed soil resulting from heavy livestock grazing and drought (Photographs courtesy of Unites States Geological Survey and Turner and others 2003).

Understory - Not an applicable category for a grassland system.

Herbaceous Layer – We found no studies that documented the historic herbaceous species component for the semi-desert grassland vegetation type.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure – We found no studies, in addition to those cited in the **Overstory** section, that documented historic canopy cover for the semi-desert grassland vegetation type.

Structure Class (Size Class) - We found no studies that documented the historic structure class of trees for the semi-desert grassland vegetation type.

Life Form – Based on guidelines from the Southwest Region office's mid-scale vegetation mapping effort, life form of vegetation (≥ 10% tree cover = tree, ≥ 10% shrub cover = shrub, ≥ 10% herbaceous cover = herbaceous) was visually estimated for each of the 48 pre-1905 photographs available for the semi-desert grassland vegetation type from the SWFAP photographic database. Results of this analysis revealed that 28 of the photographs depict an herbaceous life form, 15 depict an herbaceous life form in the valleys with a shrub life form on hillsides, drainages, or in washes, and 5 depict a shrub life form only. While there are biases associated with the number of photographs taken for each geographic area, where photographs were taken, and what photographs were taken of, the majority of photographs taken around the turn of the century within the semi-desert grassland vegetation type depict that the majority of the landscape had less than 10% shrub cover with higher shrub cover values on hillsides and in drainages and washes.

Density - We found no studies, in addition to those cited in the **Overstory** section, that document the historic density of trees for the semi-desert grassland vegetation type.

Age Structure - We found no studies that documented the historic age structure of trees for the semi-desert grassland vegetation type.

Patch Dispersion – Based on the above mentioned photographic analysis, the localization of mesquite to washes at the turn of the century is documented (Figure 2-1, Figure 2-2, Figure 2-3, Figure 2-4). Likewise, the lack of shrub cover in grassland valley bottoms and higher shrub cover, especially of juniper, in drainages is also documented.

Reference Sites Used

Limitations – Information on semi-desert grasslands comes primarily from 3 locations within Arizona and New Mexico; the Santa Rita Experimental Range in Arizona, the Jornada Experimental Range in Las Cruces, New Mexico, and the Sevilleta National Wildlife Refuge in central New Mexico. The limited number of sites from which to draw information for such a large geographic region is certainly a limitation. Additionally, all three locations were/are subject to livestock grazing and lack natural fire regimes, hence they are not ideal reference sites.

Characteristics of Applicable Sites – Ideally reference sites for the semi-desert grassland vegetation type would have intact historic disturbance processes, most notably frequent fire, and would never have been grazed by livestock. However, since heavy livestock grazing was ubiquitous around the turn of the century and fire suppression has been the norm for the past 120 years, an ideal reference site may not exist anymore. The addition of reference sites and research locations from geographic regions not currently represented in the literature, such as valley bottom grasslands, as well as the introduction of natural fire regimes into reference sites would greatly increase our understanding of this system.

2.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory - Large herbivores were not present within the semi-desert grasslands for the last 10,000 to 12,000 years until their introduction by the Europeans to New Mexico and Arizona as early as 1598 and the late 1600's respectively (Finch 2004). However, negative livestock impacts were not noted until the 1870's when overstocking and overgrazing began, with livestock numbers peaking in the late 1880's to early 1890's (Bahre 1991; Finch 2004). During this peak, Arizona and New Mexico were recorded to have 4.5 and 9 million animal units, respectively, grazing on rangelands (Finch 2004). These high stocking rates along with summer drought in 1891 and 1892 caused severe impacts to semi-desert grasslands throughout the states followed by a dramatic decline in livestock numbers (Finch 2004). While the number of livestock grazing in Arizona and New Mexico has declined considerably since the early 1890's, the effects of livestock grazing on semi-desert grasslands continue.

The impacts of livestock grazing vary considerably based on the intensity and seasonality of use and most importantly with respect to average annual precipitation (Holecheck and others 1998; Van Poolen and Lacey 1979). Holecheck and others (1998) identify impacts of heavy grazing on grasses and grasslands as the following:

- Decreased photosynthesis
- Reduced carbohydrate storage
- Reduced root growth
- Reduced seed production
- Reduced ability to compete with ungrazed plants
- Reduced mulch accumulation. This decreases soil water infiltration and retention. Mulch is also necessary to prevent erosion.

Holecheck and others (1998) as well as other researchers have also noted that, commensurate with grazing intensity, livestock grazing acts to reduce water infiltration rates, increase surface runoff, and increase soil erosion via decreasing plant cover and increasing soil compaction (Holecheck and others 1998; other erosion resources). Additionally, researchers have noted shifts in grassland species composition from more palatable to less palatable species due to livestock grazing, with most pronounced differences in species composition occurring under heavy grazing practices (Holecheck and other 1998; McClaran 2003; McPherson 1997; Ruyle 2003). Shifts towards woody species dominance have been documented under livestock exclusion, suggesting that fire suppression was the critical factor in this compositional shift within semi-desert grasslands (Brown and Archer 1989; Holecheck and others 1998; McClaran 2003; McPherson and Weltzin 1998).

There is some discussion that “light to moderate” grazing may have beneficial effects on rangeland plants (Holecheck and others 1998; Whitford 2002). Holecheck and others (1998) identify the “possible positive effects of light to moderate grazing on range plant physiology” as follows:

- Increased photosynthesis
- Increased tillering
- Reduced shading
- Reduced transpiration
- Inoculation of plant parts with growth-promoting substances
- Reduction of excessive mulch accumulation that may physically and chemically inhibit vegetative growth. Excessive mulch can provide habitat for pathogens and insects that can damage forage plants.

While the above mentioned negative effects of heavy grazing are well documented in studies within the southwest, positive effects are not well documented and Holecheck and others (1998) state that these positive effects are “most likely (to occur) in areas receiving over 400 mm of average annual precipitation. Below this level of precipitation, excessive accumulations of vegetation usually do not occur, due to aridity”. Additionally, they note that forage production, grazing resistance of grasses, and grassland recovery following heavy livestock grazing is lower in arid grasslands (areas, such as the semi-desert grasslands, that receive less than 300 mm average precipitation per year). Specifically, studies from New Mexico desert grasslands showed heavy grazing to have large impacts within a couple of years but recovery after 20 years was slow especially on sites with loss of topsoil (Holecheck and others 1998).

Several studies from New Mexico and Arizona documented rangeland improvement or maintenance of perennial grasses under light (35% to 40% or unindentified) livestock utilization levels (Cable and Martin 1975; Holecheck 1998; McClaran 2003). Based on these studies and others, Holecheck and others (1998) recommend 30% to 40% livestock utilization for semi-desert grass and shrubland systems in order to maintain “critical minimum” residual grass cover. While there is documentation that light grazing may have minimal to no effects on semi-desert grassland, it is important to note that it is difficult to maintain these low levels of utilization over time. Even the Santa Rita Experimental Range has not been able to reduce utilization to this level despite “repeated attempts” (McClaran 2003).

Silviculture - Not an applicable category for a grassland system

Fragmentation - Population expansion in southeastern and central Arizona over the last 70 years has led to increased urban development in the surrounding grassland and oak woodlands (Bahre 1991). The lure of temperate weather, pastoral views, and open space draws many people, especially retired persons, to Arizona and its grassland communities (McPherson 1997). In fact, Arizona lost 403,000 acres to rural development between 1982 and 1997, this 37 % loss of rural lands was 3 % greater than the national average of 34 % (Sprawl City http://www.sprawlcity.com/studyAZ/index.html). The problem is so great that multiple studies have noted the negative effects of urban expansion on grassland communities and their associated species (Bahre 1991; Bock and Bock 2002; Finch 2004; McPherson 1997; Turner and others 2003) and many have even identified it as the greatest threat to grasslands (Finch 2004; McPherson 1997; Neff 1986; Ockenfels and others 1994; van Riper 1998). Urban expansion has lead to the loss and fragmentation of grassland vegetation and the disruption of historic processes, such as fire, that maintained the vegetation through increased fencing, road access, recreation, introduction of non-natives and home building (Bahre 1991; Finch 2004; McPherson 1997). For wide-ranging grassland species, such as pronghorn, development and fragmentation has had drastic impacts on their abundance and distribution (Neff 1986; Ockenfels and others 1994; van Riper 1998).

Mining - We found no studies that documented the effects of mining within the semi-desert grassland vegetation type. However, mining effects documented in Madrean encinal section may be useful.

Fire Management - Passive fire suppression, through livestock grazing beginning in the late 1800's as well as active suppression increasing over the last 120 years, has resulted in reduced fire return intervals in semi-desert grasslands (Davis and others 2002; Kaib and others 1996; McPherson 1995). This decrease in fire frequency for southeastern Arizona was reported in Kaib and others' (1996) fire scar study as well as by Davis and others' (2002) sediment study. Specifically, Kaib and others (1996) investigated fire scar data for 2 southeastern Arizona canyons linked by grasslands, they found that fires dropped in occurrence from every 4 to 8 years on average, between 1600 and 1899 in both canyons, to every 25 years in one canyon with no fires occurring in the second canyon. Similarly, Davis and others (2002) found a 4 to 120 fold decrease in charcoal abundance (circa 200 years B.P) from sediment cores taken from grassland cienegas in southeastern Arizona.

While there have been some wildland fire or prescribed burns that have occurred within semi-desert grasslands in the last 120 years, it is only recently that national attention has been focused on returning fire to fire adapted ecosystems and that discussions and planning for prescribed and wildland fire use within semi-desert grasslands have truly begun (National Fire Plan 2000). Sayre (2005) outlines some of the current obstacles associated with applying fire in semi-desert grasslands of southeastern Arizona, they are landownership, livestock grazing, and proximity of human developments. While fire re-introduction is beginning to take place, little attention is being paid to the season in which fires occurr. We know fires historically occurred between June and July when flammable fine fuels and dry lightning strikes were abundant (Kaib and others 1996; McPherson 1995; Bahre 1985). However, most prescribed fires occur earlier in the spring or later in the fall when fires are easier to control but when they may have unknown or negative effects on the grassland system (McPherson pers. Comm.).

Exotic Introductions (Plant & Animal) - There are two invasive non-native perennial grasses that occur throughout the semi-desert grassland region, Lehmann lovegrass and Boer lovegrass. The most common and abundant is Lehmann lovegrass which is a drought tolerant perennial grass from South Africa (Crider 1945; Gori and Enquist 2003). Boer lovegrass is also a native of South Africa, but is adapted to cooler, slightly wetter conditions than Lehmann lovegrass (Ruyle and Young 1997). In the 1930's, both grasses were seeded along roadsides and on rangelands in southeastern Arizona by the Soil Conservation Service in an effort to stop soil loss (Cox and Ruyle 1986).

Both non-native perennial grasses are adapted to frequent fire and recover quickly from fire disturbance. In many cases, these non-native grasses increase more rapidly than the native perennial grasses (Anable and others 1992). In particular, Lehmann lovegrass is adapted to germinate on open bare soil, and increases on sites following disturbances such as fire and drought (Anable 1990; Angel and McClaran 2001). Finally, both non-native grasses produce higher amounts of biomass than native grasses hence they can carry fires more easily and produce hotter fires than native grasslands (McPherson and Weltzin 1998). With the continued spread of the grasses, fire regimes in invaded areas may increase in frequency and intensity.

Additionally, Lehmann lovegrass has been implicated in contributing to decreased plant and animal species richness (Cable 1971; Bock and others 1986; Medina 1988), alteration of ecosystem processes, such as soil carbon and nitrogen ratios, water infiltration rates, and fire regimes (Cable 1971; Bock and others1986; Williams and Baruch 2000) as well as modification of plant community composition (Cable 1971; Anable and others 1992; Kuvlesky and others 2002). Both Boer lovegrass and Lehmann lovegrass are currently found along roadsides and in scattered to rare abundance throughout semi-desert grasslands in southeastern Arizona; they are now common to dominant on 1,469,319 acres there (Gori and Enquist 2003).

2.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory - There have been many studies that have investigated vegetation changes in the semi-desert grasslands over the last 150 to 200 years. These studies range in location from the Santa Rita Experimental Range (SRER) in southeastern Arizona; Jornada Experimental Range (JER) and Chihuahuan Desert Range Research Center (CDRRC) in southern New Mexico; Lincoln county New Mexico; Malpais borderlands in southeastern Arizona and southwestern New Mexico; southeastern Arizona/northern Mexico; to all semi-desert grasslands in Arizona, southwestern New Mexico and northern Mexico (Buffington and Herbel 1965; Davis and others 2002; Gori and Enquist 2003; Hennessy and others 1983; Humphrey and Mehrhoff 1958; Muldavin and others 2002; Turner and others 2003). Strikingly, all these studies concluded that mesquite (*Prosopis velutina* and *glandulosa*) increased in acreage and cover within semi-desert grasslands while native perennial grass dominated areas decreased in acreage (Figure 2-5 to 2-8).

Specifically, on the JER and CDRRC, areas identified in 1858 as fair to very good grass covered 98 % and 67 % of the land, respectively, at the two sites, with 45 % and 18 % of the two areas classified as shrub free (Gibbens and others 2005). By 1998 mesquite and creosote bush had become dominant on the JER covering 59 % and 27 % of the JER respectively; their dominance on the CDRRC amounted to 37 % and 46 % of the area (Gibbens and others 2005, Figure 2-6). These field based studies were corroborated by Laliberte and others (2004) remotely sensed study of the CDRRC which found a 0.2% per year increase in the percent of shrub cover between 1937 and 2003 with most of the increase occurring after the 1950’s drought. Similarly, Turner and others (2003) found mesquite to be increasing at all 28 southeastern Arizona grassland photo stations between the early 1900's and 1962, and by the 1990's, mesquite had continued to increase on 18 of the 28 grassland photo stations (Figure 2-8). In addition, Turner and others (2003) also noticed an increase in one seed juniper (*Juniperus monosperma*) in semi-desert grasslands within Arizona.

Taking a broader look, The Nature Conservancy's regional grassland assessment, identified a total of 13,115, 000 acres of semi-desert grasslands in southeastern Arizona, southwestern New Mexico and northern Mexico. Thirty six percent of these grasslands were historic grasslands that are now converted to shrubland, another 32 % of extant and former grasslands have between 10 % and 35 % shrub cover, while 12 % have non-native perennial grasses as common or dominant (Gori and Enquist 2003). Only 17 % of extant and former grasslands within the region can be classified as open (less than 10 % shrub cover) native grasslands (Gori and Enquist 2003). While we do not know what percent of the landscape would have historically been in an open native condition, based on our knowledge of vegetation dynamics within the system and historic photographs, it appeared that the majority of the semi-desert grasslands would have historically fallen into this category. Additionally, it is important to note that the 32 % of the regional grasslands identified as having 10 % to 35 % shrub cover are potentially restorable, to lower shrub cover levels, through prescribed or wildland fire.

Lack of fire has been implicated in the increased density and cover of mesquite, juniper, broom snakeweed, burroweed, creosote bush, and cacti (Buffington and Herbel 1965; Gori and Enquist 2003; Hennessy and others 1983; Humphrey and Mehrhoff 1958; Muldavin et al. 2002; Turner et al. 2003). This increase in woody species has been documented both with and without the presence of livestock grazing and has not been convincingly tied to climatic changes (McPherson and Weltzin 1998; Turner and other 2003). Ultimately, the increase in trees and shrubs has changed vegetation in the semi-desert grasslands from a predominantly open perennial grass system to mixed shrub, tree, and perennial grass system with multiple areas having been converted to shrublands (Gibbens and others 2005; Gori and Enquist 2003).

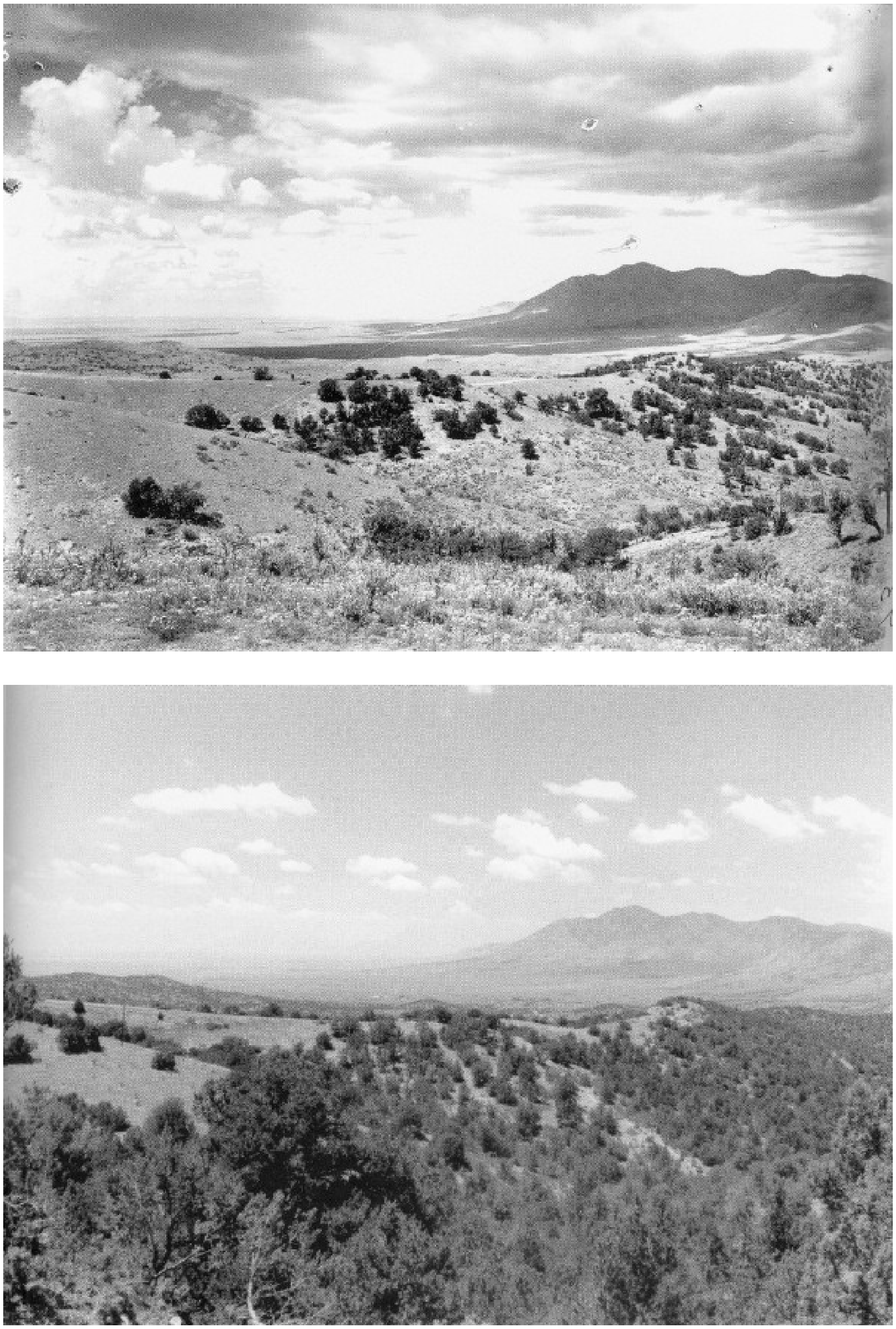

Figure 2-5. Repeat photography sequence taken in 1899 (top) and 1996 (bottom) at Fort Stanton, New Mexico. Photograph depicts expansion of juniper from the hillsides out into the open grassland valley bottom as well as increasing juniper cover on hillside (Photographs courtesy of Hollis Fuchs 2002).

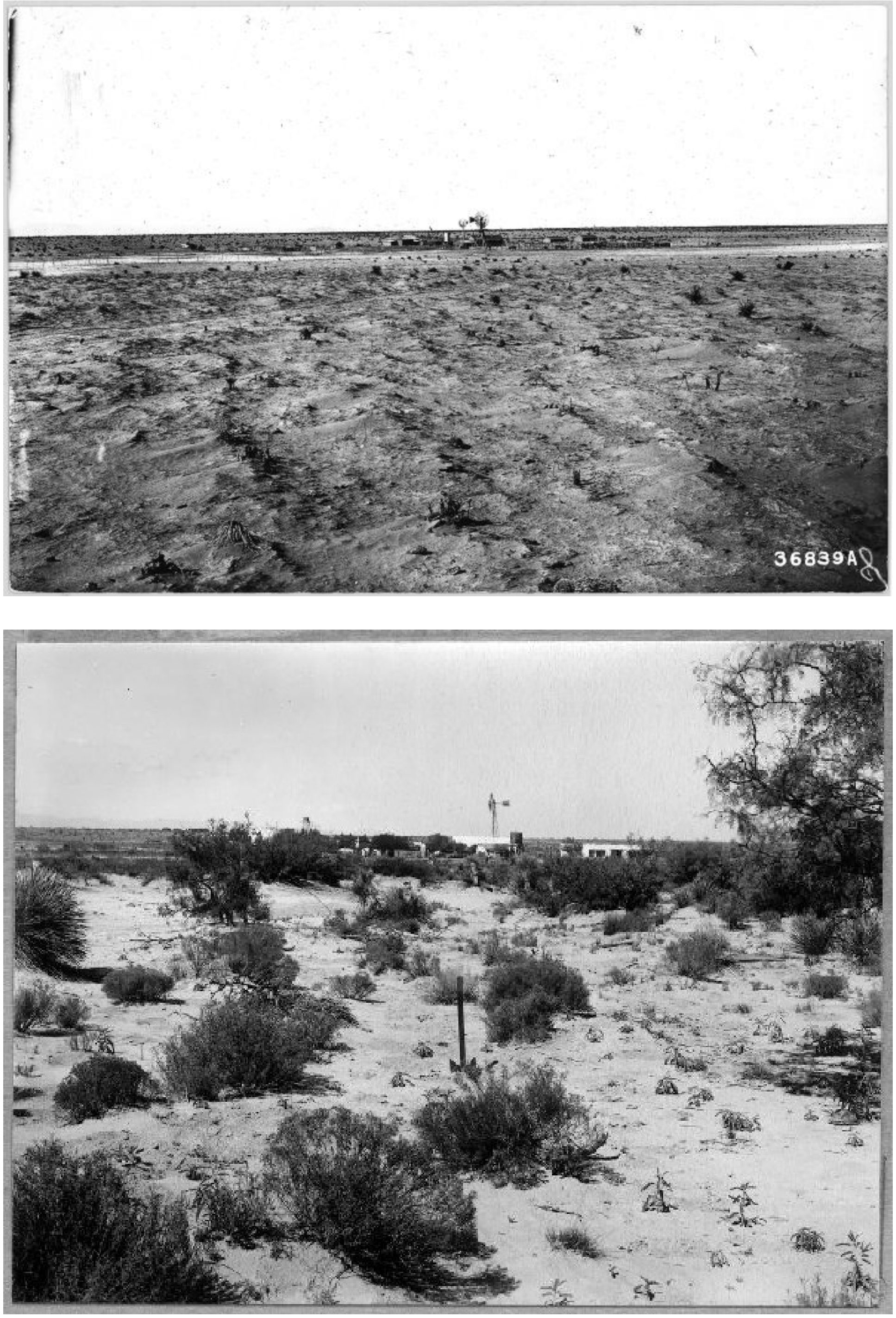

Figure 2-6. Repeat photography sequence taken in 1918 (top) and 1931 (bottom) at the Jornada Experimental Range in Las Cruces, New Mexico. Photograph depicts the transition from a open grassland to a dune shrubland. (Photographs courtesy of Jornada Experimental Range).

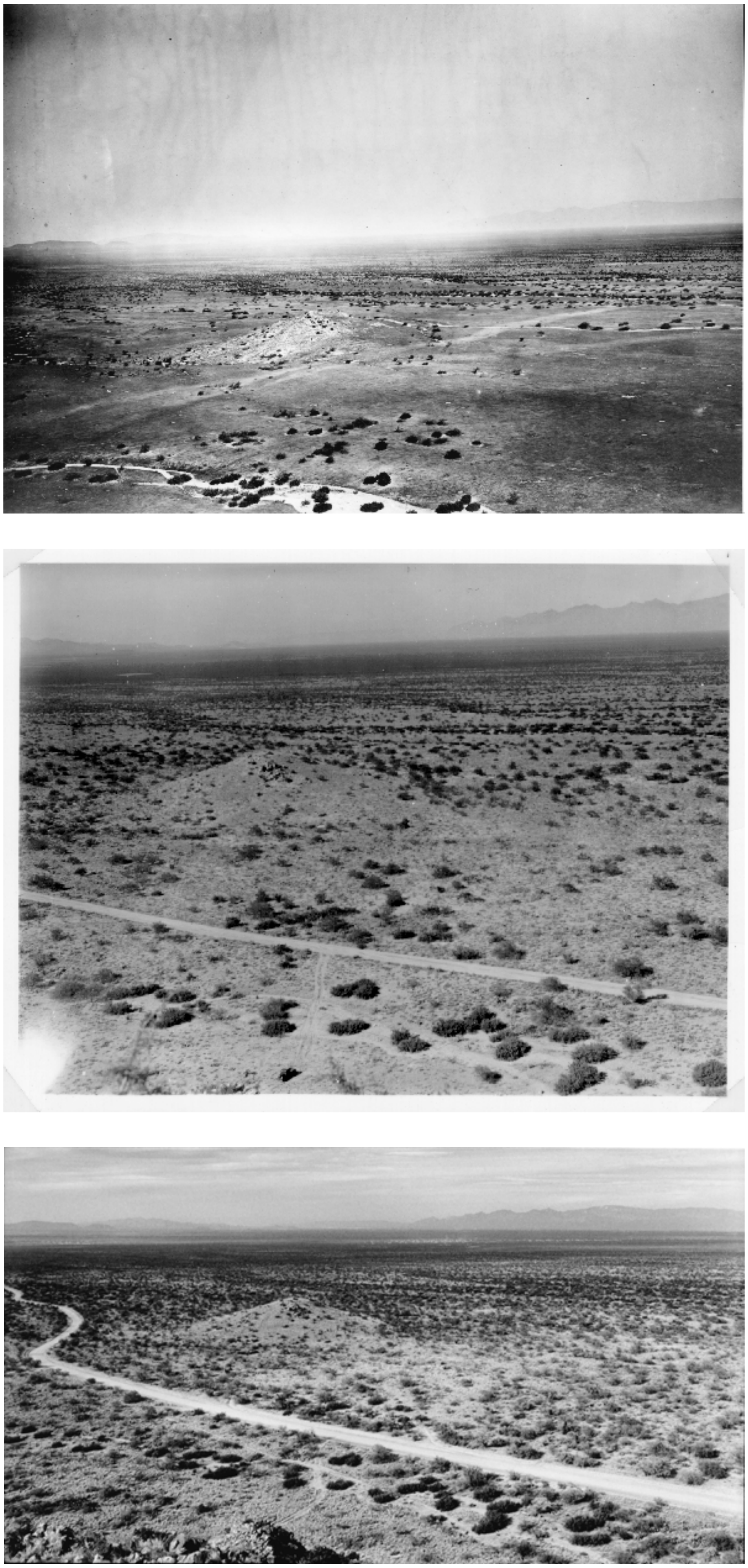

Figure 2-7. Repeat photography sequence taken in 1902 (top) and 1950 (middle) and 2000 (bottom) at the Santa Rita Experimental Range, southeastern Arizona. Photographs depict the change in cover and patch distribution of shrubs over the last 100 years on the SRER. Specifically, it is easy to see the expansion of mesquite out of the drainages and onto the open grassland (Photographs courtesy of Santa Rita Experimental Range).

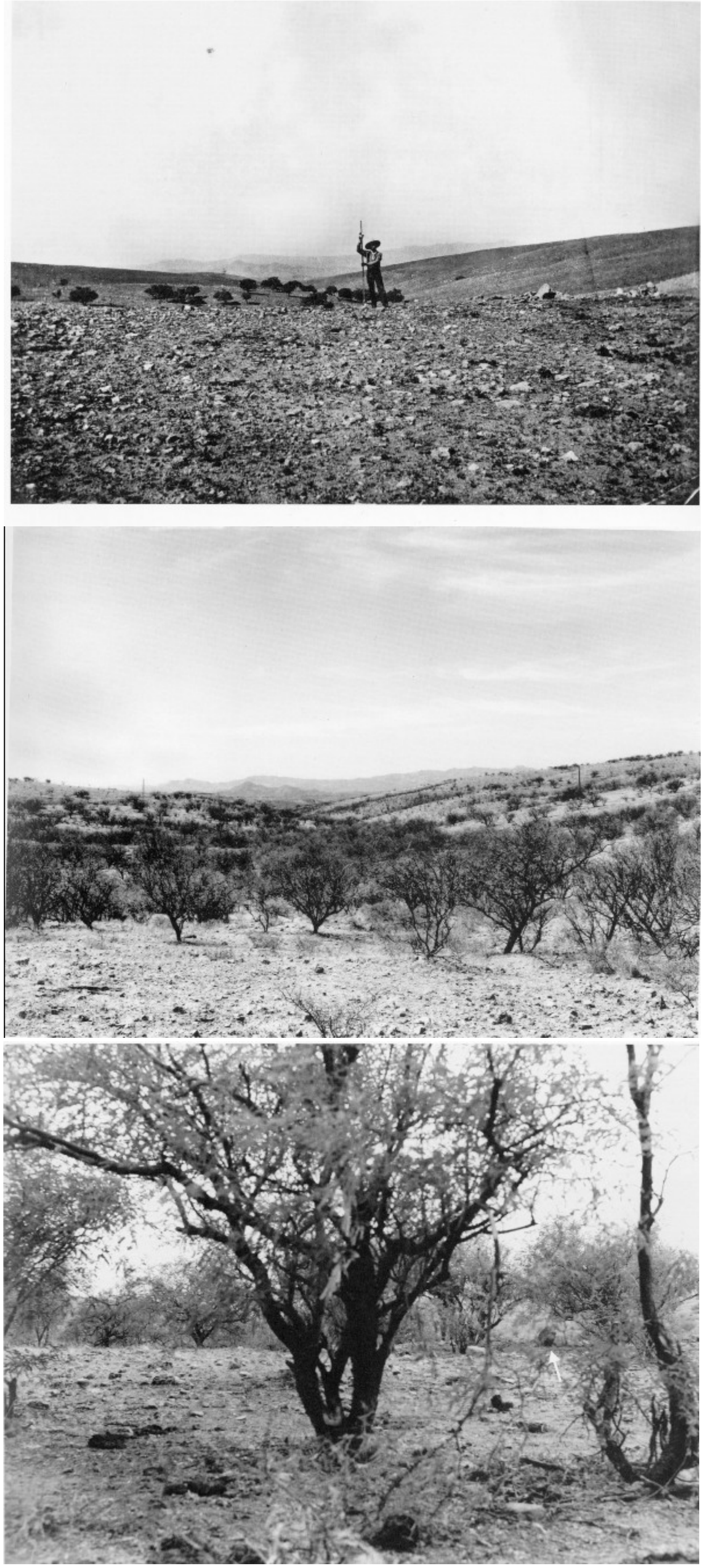

Figure 2-8. Repeat photography sequence taken in 1890 (top) and 1962 (middle) and 1996 (bottom) in Guevavi Canyon, Arizona. Photograph depicts the transition from an open grassland to a mesquite woodland. (Photographs courtesy of Unites States Geological Survey and Turner and others 2003).

Understory - Not an applicable category for a grassland system.

Herbaceous Layer - We found no studies, in addition to those cited in the **Overstory** section, that documented changes within the herbaceous layer for the semi-desert grasslands vegetation type.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure - We found no studies, in addition to those cited in the **Overstory** section, that documented changes canopy cover for the semi-desert grassland vegetation type.

Structure Class (Size Class) - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in tree size classes for the semi-desert grassland vegetation type.

Life Form - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in life form for the semi-desert grassland vegetation type.

Density – We found no studies, in addition to those cited in the **Overstory** section, that documented changes in tree density for the semi-desert grassland vegetation type.

Age Structure - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in tree or grass age structure for the semi-desert grassland vegetation type.

Patch Dispersion - Multiple studies have noted the movement of mesquite and other shrubs out from washes and drainages and into open grasslands (Fuchs 2002; McClaran 2003; Turner and others 2003). Studies of the Santa Rita Experimental Range in Arizona showed mesquite, catclaw acacia, blue palo verde trees, and creosote bush to be most abundant below 3,280 ft in large washes in the early 1900's (McClaran 2003). By 1915 mesquite was noted occurring between washes within the open grassland, and by the 1950's mesquite densities had increased within the grasslands and moved up in elevation to about 3,440 ft with expansion to 4,430 ft by the 1980's (Figure 2-7). Similarly, Turner and others (2003) found mesquite to be increasing at all 28 southeastern Arizona grassland photo stations between the early 1900's and 1962; by the 1990's, mesquite had continued to increase on 18 of the 28 grassland photo stations. Additionally, Fuchs' (2002) repeat photography study showed the expansion of one-seed juniper from drainages into open grassland areas (Figure 2-8).

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Chapter 3 – Madrean Encinal

3.1 General Description

Madrean encinal, or oak woodland, occurs throughout southeastern Arizona, southwestern New Mexico, northeastern Sonora, and northwestern Chihuahua, discontinuously distributed in the foothills of isolated mountain ranges at elevations ranging from 3,600 to 6,500 ft. These woodlands grade into semi-desert grasslands at lower elevations and pine-oak woodlands in higher elevations (McPherson1997; Turner and others 2003). The majority of these oak woodlands occur in Mexico (90 %) with only a scant 10% occurring in the United States (McPherson 1997). Emory oak (*Quercus emoryi*) is present throughout the range of Madrean encinal. However, Mexican blue oak (*Quercus oblongifolia*), Arizona white oak (*Quercus arizonica*), and gray oak (*Quercus grisea*) are the most common oak species in the United States and northern Mexico while Chihuahua oak (*Quercus chihuahensis*), Cusi oak (*Quercus albocincta*), *Quercus chuchuichupensis*, and *Quercus santa-clarensis* are common in the Mexican states of Chihuahua and Durango (McPherson 1997). Warm season perennial bunch grasses, such as sideoats grama (*Bouteloua curtipendula*), blue grama (*Bouteloua gracilis*), hairy grama (*Bouteloua hirsute*), purple grama (*Bouteloua radicosa*), Plains lovegrass, Mexican lovegrass (*Eragrostis mexicana*), deer grass (*Muhlenbergia emersleyi*), and longtongue muhly (*Muhlenbergia longiligula*), dominate the understory.

3.2 Historic Range of Variation of Ecological Processes

Vegetation Dynamics – Madrean encinal woodlands were historically dominated by an open stand of oaks with denser stands of oaks on north facing slopes and in drainages. Perennial grasses composed the understory and provided fuel for surface fires. Over the last 150 years, Madrean encinal has trended away from these open woodlands and towards woodlands with higher canopy cover and higher abundances of mesquite and juniper trees (Turner and others 2003). Regeneration of the dominant oak species is primarily due to re-sprouting following a disturbance with little regeneration from acorns due to dry conditions (Germaine and McPherson 1999). Additionally, drought has been suggested to cause mortality in oaks at the lower elevational bound of this PNVT (Turner and others 2003).

Disturbance Processes and Regimes

Below is a discussion of the frequency, intensity, severity, seasonality, and spatial and temporal scale of disturbances that occur within the Madrean encinal vegetation type.

Climate – See Chapter 1, climate analysis section.

Fire - Frequent fires, occurring primarily between April and June, have been well documented for the semi-desert grasslands and Madrean pine-oak woodlands that border the Madrean encinal, however, little is known about the frequency of fire in southwestern oak woodlands (Bahre 1985; Fulé and others 2005; Humphrey 1952, Kaib et al 1996, McPherson 1995, Swetnam and Baisan 1996; Swetnam and others 1992; Wright 1980). Given that fires in semi-desert grasslands occurred on average every 2.5 to 10 years, covered hundreds of square miles, and often spread to upper elevation Madrean pine-oak systems, which experienced fires on average every 3 to 7 years, it seems likely that Madrean encinal also experience frequent fires (Bahre 1985; Fulé and Covington 1998 b; Fulé and others 2005; Kaib et al. 1996; McPherson 1995; Swetnam and Baisan 1996; Swetnam and others 1992). The ability of the dominant vegetation, evergreen oaks and perennial bunch grass, to recover quickly after fire also supports the idea of frequent fire in oak woodlands (Bock & Bock 1992; Cable 1972; Caprio and Zwolinski 1992; Martin 1983).

The timing of fires was probably similar to that of other Madrean PNVTs and occurred predominantly between April and June due to dry lightning strikes that precede the summer monsoon (Swetnam and Betancourt 1990). The size and intensity of these fires is however largely unknown. Research by Caprio and Zwolinski (1992) suggests that the differences between Mexican blue oak and Emory oak in sprouting and live crown survival following fire may relate to different fire frequencies and intensity experienced. Specifically, they suggest that the higher rate of live crown survival seen in Mexican blue oak compared to Emory oak along with its thicker bark may make Mexican blue oak more able to survive the frequent low intensity fires associated with grasslands, while Emory oak may be restricted to areas with less frequent higher intensity fires (Caprio and Zwolinski 1992).

Given that little information regarding fire frequency, size, or intensity is directly known, caution should be taken in extrapolating information from surrounding vegetation (Guy McPherson personal communication). As one way of adding caution, it is important to remember that there is variability in fire occurrence, and while mean fire return intervals were highlighted above, the variation around the mean is also important and has impacts on the establishment of woody species. For example, research by Barton and others (2001) found that in Madrean pine-oak woodlands, fire free periods of between 20 and 30 years in length were necessary to allow Arizona pines to establish while subsequent years of frequent fire were responsible for allowing mature pines to remain dominant. The range of variation in fire frequency (between 1 and 38 years between any given fire) identified in Madrean pine-oak woodlands supports this idea (Fulé and others 2005; Kaib and other 1996; Swetnam and Baisan 1996; Swetnam and others 1992). Additionally, a study by Borelli and others (1994) found that most oak regeneration came from stump sprouts (56 %) and root sprouts (25 %), with only 19 % of seedlings coming from seed. This infrequent regeneration by seed may highlight the importance of climate and herbivory in regulating acorn germination and may be an important factor to consider when evaluating appropriate fire regimes for this PNVT.

Hydrology – A synthesis study of hydrology and watershed management in Madrean encinal by Lopes and Ffolliott (1992) documents that there is little known about the hydrology of this system. What is known is that surface runoff and soil erosion increases with increasing intensity and/or duration of rainfall or snowmelt events and that vegetative cover, provided by both trees and herbaceous plants, increases water infiltration thus decreasing surface runoff and erosion (Lopes and Ffolliott 1992).

Herbivory - Herbivory from native insects and mammals has been documented to have significant impacts in grassland vegetation in the southwest. Rodents, such as prairie dogs and kangaroo rats, have been shown to have large impacts on the structure and composition of vegetation in grassland and desert ecosystems in the southwest. These impacts are primarily due to rodents preferentially eating the seeds and/or herbaceous parts of plants and/or their ability to cache seeds of woody plants at optimal germination depths (Brown and Hesky 1990; Finch 2004; McPherson 1997; Miller and others 1994; Weltzin and others" 1997; Parmenter and Van Devender 1995). While it is not clear if results from lower elevation grassland studies are directly applicable to the oak woodlands, it seems likely that there was some overlap of native herbivores at least in low elevation woodlands. For a more detailed discussion of grassland herbivores, see semi-desert grassland **Herbivory** section. Vertebrate and invertebrate herbivory has been documented to be an "extremely variable and unpredictable" cause of seedling mortality within Emory oaks in southeastern Arizona with mortality of seedlings primarily due to desiccation (Germaine and McPherson 1999). Additionally, vertebrate herbivores, mostly birds, can have negative and positive effects on germination via consumption of large numbers of acorns and the caching of acorns at optimal germination depths and long distances from the original seed source (McClaran and McPherson 1999).

Ultimately, while we know that herbivores historically played a role in seedling establishment, mortality, and distribution within the oak woodlands, there is little information that identifies the spatial or temporal extent of these interactions or the effect that herbivores have on species composition or the dynamics of plant populations in the Madrean encinal.

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and/or introductions as important ecological determinants for the Madrean encinal vegetation type.

Insects and Pathogens - We found no studies, in addition to those mentioned in the herbivory section, that implicated insects and/or pathogens as important ecological determinants for the Madrean encinal vegetation type.

Nutrient Cycling - We found no studies that document nutrient cycling in the Madrean encinal vegetation type.

Windthrow - Not an applicable category for this system

Avalanche - Not an applicable category for this system

Erosion – We found no studies that document erosion in the Madrean encinal vegetation type.

3.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation **–** Nineteen historic photographs and accompanying annotations from Turner and others (2003) book "The changing mile revisited" were analyzed for vegetation condition and species composition. All 19 photographs were taken in southeastern Arizona between 1887 and 1895. While these data do not give a range of historic values for vegetation characteristics over time, by synthesizing information from multiple locations we can get some idea of the range of vegetation conditions and species that existed around the turn of the century. Below is a summary of information collected from the 19 photographs.

Overstory – General description of photos: Photographs show open grassland valleys on south facing slopes with moderate tree cover in drainages and on north facing slopes, see Figure 3-1. A list of tree, shrub and grass species identified in the photograph annotations are listed below.

Tree species:
Arizona white oak (*Quercus arizonica*)
Cottonwood (*Populus fremontii*)
Emory oak (*Quercus emoryii*)
Mexican blue oak (*Quercus oblongifolia*)
Mesquite (*Prosopis velutina*)

Shrub species:
Palmer Agave (*Agave palmeri*.)
Bear grass (*Nolina microcarpa*)
Ocotillo (*Fouquieria splendens*)
Palmillas (*Yucca elata*)
Sotol (*Dasylirion wheeleri*)
Yucca (*Yucca arizonica*)

Perennial grass species:
Not mentioned by name, but the presence of perennial grasslands are noted.

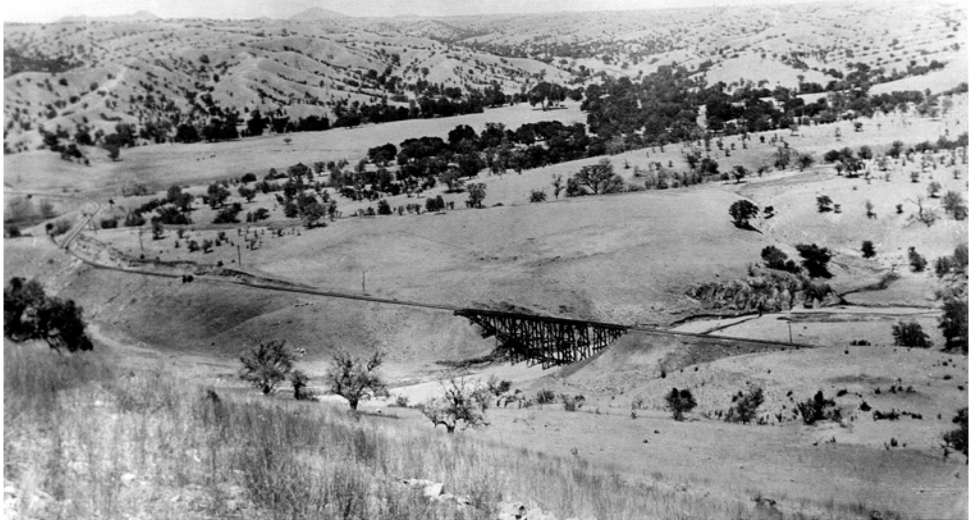

Figure 3-1. 1895 photograph of Red Rock Canyon, east of Patagonia Arizona (top), and 1895 photograph taken southwest of present day Sonoita Arizona with Santa Rita Mountains in the background (bottom). Photographs depict oak woodlands with low shrub cover except on north facing hill slopes and drainages. (Photographs courtesy of Unites States Geological Survey and Turner and others 2003).

Understory – In addition to information cited in the **Overstory** section, descriptions of photographs from Turner and others (2003) include statements such as "grassy" areas and areas dominated by "grasslands" to describe Madrean encinal photographs. Similarly, Muldavin and other's (2002) study, which coded General Land Office surveyor's notes into vegetation classes for the Malpais borderlands region, identified scattered woodlands that had grass understories. Specifics regarding species composition or cover values were not given.

Herbaceous Layer – We found no studies that documented the historic herbaceous component for the Madrean encinal vegetation type.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure - See *Patch Composition of Vegetation, Overstory* in section **2.5 Effects of Anthropogenic Disturbance.**

Structure Class (Size Class) - We found no studies that documented the historic structure class of trees for the Madrean encinal vegetation type.

Life Form – Based on guidelines from the Southwest Region office's mid-scale vegetation mapping effort, life form of vegetation (≥ 10% tree cover = tree, ≥ 10% shrub cover = shrub, ≥ 10% herbaceous cover = herbaceous) was visually estimated for each of the 19 pre-1900 photographs available for the Madrean encinal vegetation type from Turner and others (2003) book. Results of this analysis revealed that 18 of the photographs depict an herbaceous life form, in the valleys with a tree/shrub life form on the north slopes of hillsides and in drainages while only 1 photograph depicted a tree/shrub life form dominating the photograph. While there are biases associated with where the photographs were taken and what the photographs were taken of, the majority of photographs taken before the turn of the century within the Madrean encinal vegetation type depict the majority of the landscape to be open grassland with trees/shrubs on north facing slopes and drainages, with oaks being the dominant tree species. Mesquite was identified in 3 of the 19 photographs, while oaks were identified in 16 of the 19 photographs.

Density - We found no studies, in addition to those cited in the **Overstory** section, that document the historic density of trees within the Madrean encinal vegetation type.

Age Structure - We found no studies that documented the historic age structure of trees for the Madrean encinal vegetation type.

Patch Dispersion – We found no studies that documented the historic patch dispersion of trees for the Madrean encinal vegetation type.

Reference Sites Used

Limitations – Information for HRV of vegetation characterisitics in the Madrean encinal vegetation type comes largely from Turner and others (2003) repeat photography analysis in "The changing mile revisited." While this book offers good information for around the turn of the century, the temporal and spatial limits of the study are limitations in understanding a true range of characteristics. There are also limits associated with information on the historical processes, as much of this information comes from neighboring vegetation types and not studies from the Madrean encinal.

Characteristics of Applicable Sites – Ideal reference sites within the Madrean encinal would have a historical fire regime, would be free from livestock grazing and fuel wood cutting, and would be scattered throughout the encinal in southeastern Arizona, southwestern New Mexico, and northern Mexico. Ideally information surrounding vegetation response to fire, native herbivory, and climatic factors would be obtained from such sites.

3.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory - Livestock grazing in Madrean encinal is currently a common practice in both the United States and Mexico with grazing occurring in virtually all of Mexico’s and in roughly 75 % of the United States’ oak woodlands (McPherson 1997). For a detailed discussion of livestock impacts in rangelands see Chapter 2 – Semi-desert grassland.

Silviculture - Fuel wood cutting for mining and domestic use in Madrean encinal was common in southeastern Arizona until the late 1800’s, and is still common in Arizona and northern Mexico today (Bahre 1991; Bennet 1992; Webster and Bahre 2001). In southeastern Arizona, mesquite, oak, and juniper at low elevations, along with pines from higher elevations were the dominant species cut for fuel wood and timber for mining and domestic uses. Intense harvesting in and around mining towns in southeast Arizona between 1870 and the 1890’s (peak years) had a devastating effect on trees in the surrounding watershed (Bahre 1991). For example, Bahre (1991) estimated that the total amount of fuel wood cut from the Tombstone watershed between 1878 and 1940 was “more than the total cordage presently reported by the Forest Service for the evergreen woodlands of the Tombstone woodshed”. While it is certain that this short duration, high intensity utilization of woodlands had a devastating effect on Madrean encinal and pine-oak woodlands, the disturbance was highly variable and predominantly effected watersheds near mining towns (Bahre 1991: Turner and others 2003). In addition, based on repeat photography, Turner and others (2003) concluded that while fuel wood harvesting had a dramatic effect, its “influence was local and transient”.

Additionally, a study by Sharman and Ffolliott (1992) documented the effects of fuel wood cutting on the stand structure of oak woodlands in southeastern Arizona between 1972 and 1991. Specifically, they documented that cutting intensities of 80% decreased tree height, crown depth, crown volume, and foliage volume while increasing stems per tree when compared to an undisturbed area (Sharman and Ffolliott 1992). Cutting intensities of 50% only resulted in decreased tree height and crown depth (Sharman and Ffolliott 1992).

Fragmentation - Population expansion in southeastern and central Arizona over the last 70 years has led to increased urban development in the surrounding grassland and oak woodlands (Bahre 1991). The lure of temperate weather, pastoral views, and open space draws many people, especially retired persons, to Arizona and its grassland communities (McPherson 1997). In fact, Arizona lost 403,000 acres to rural development between 1982 and 1997, this 37 % loss of rural lands was 3 % greater than the national average (34 %) (Sprawl City http://www.sprawlcity.com/studyAZ/index.html). The problem is so great that multiple studies have noted the negative effects of urban expansion on grassland communities and their associated species (Bahre 1991; Bock and Bock 2002; Finch 2004; McPherson 1997; Turner and others 2003) many have even identified it as the greatest threat (Finch 2004; McPherson 1997; Neff 1986; Ockenfels and others 1994; van Riper and Ockenfels 1988). Urban expansion has lead to the loss and fragmentation of grassland vegetation and the historic processes, such as fire, that use to maintain the vegetation through increased fencing, road access, recreation, introduction of non-natives and home building (Bahre 1991; Finch 2004; McPherson 1997). For wide ranging grassland associated species, such as pronghorn, development and fragmentation has had drastic impacts on their abundance and distribution (Neff 1986; Ockenfels and others 1994; van Riper 1988).

Mining - We found no studies, in addition to those cited in the silviculture section, that documented the effects of mining for the Madrean encinal vegetation type.

Fire Management - Passive fire suppression, through livestock grazing, beginning in the late 1800's as well as active suppression, increasing over the last 100 years, has resulted in reduced fire return intervals in both vegetation types that border the Madrean encinal woodlands. Specifically, semi-desert grasslands of southeastern Arizona have seen a drastic decline in fire occurrence, as have the Madrean pine-oak woodlands which have also seen increased fire intensity when fires do return (Fulé and others 2005; Kaib and others 1996; Swetnam and Baisan 1996). Reduced fire frequency has been implicated in the increase in woody plant densities throughout the southwest and in particular in the grasslands and pine-oak woodlands that border the Madrean encinal (Barton 1999; Buffington and Herbel 1965; Davis and others 2002; Gori and Enquist 2003; Hennessy and others 1983; Humphrey and Mehrhoff 1958; Muldavin and others 2002; Turner and others 2003). The increase in woody species in the Madrean encinal has resulted in a species composition shift, in some areas, from oak dominated woodlands to mesquite and/or juniper dominated woodlands (Turner and others 2003). Additionally, the increases in stand densities that have occurred will likely result in higher fire intensities than historically would have occurred in encinal.

Exotic Introductions (Plant & Animal) – We found no studies that documented the effects of exotic introductions for the Madrean encinal vegetation type. However, given the shared boundary between encinal and semi-desert grassland, information from the semi-desert grassland chapter dealing with *Exotic Introductions* (specifically, non-native perennial grasses) contains valid information for encinal as well.

3.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory - The most striking, and consistently agreed upon change that has occurred within Madrean encinal in the United States, since the late 1800's, is an increase in the density of woody plants, see Figures 3-2 and 3-3 (Bahre 1991; McPherson 1997; Muldavin and others 2002; Turner and others 2003). Increased density has come from both an increase in stand densities of existing oaks as well as large increases in other woody species such as mesquite and one seed juniper along with moderate increases in shrubs such as ocotillo, palmer agave, and sotol ((Bahre 1991; McPherson 1997; Turner and others 2003). Specifically, mesquite (*Prosopis sp.*) was identified by Turner and others (2003) as the main woody increaser within oak woodlands of southeastern Arizona, with all (26) of the oak woodland repeat photography stations showing increases of mesquite from around the turn of the century to 1994. In addition, they noted that by the time photos were taken in 1962 both mesquite and one seed juniper (*Juniperus monosperma*) often outnumbered oaks (Turner and others 2003). Muldavin and others (2002), in a repeat study, based on General Land Office surveyors notes, of the Malpais borderlands region between 1880 and 2002, documented a shift from open oak woodlands to denser canopied woodlands with 70% of historic scattered woodlands identified as denser mixed woodlands today.

Other possible changes that have been identified are either less well documented or have been contested by other research. These changes are 1) a shift from oak woodlands to grasslands, 2) the loss of oaks at low elevations resulting in an upslope movement of oak woodland, and 3) encroachment of oaks into grasslands resulting in a downslope movement of woodlands. The first possible change, shifts from historic oak woodlands to grasslands was documented by Muldavin and others (2002) who suggest that these changes may have been the result of fuel wood cutting in the area. While few studies within Madrean encinal document similar shifts in oak woodlands, clear cutting oak woodlands for fuel wood has been well documented and seems like a likely explanation for these results (Bahre 1991).

In regards to a possible boundary shift in oak woodlands, the jury is still out. Research on the possible boundary shift comes from repeat photography analyses and a soil organic isotope study which suggest either 1) a movement of oak woodlands upslope due to a die off of oaks below 4,500 ft (following the 1950’s drought) or 2) no change in the distribution of oak woodlands, simply increases or decreases within their historic range or 3) the encroachment of oaks into lower elevation grasslands (Bahre 1991; McClaran and McPherson 1997; Turner and others 2003). It is probable that all three events have occurred across oak woodlands in southeastern Arizona, however, given the level of debate, it can not be concluded that a trend of upslope or downslope movement exists.

What is causing the changes seen in oak woodlands over the last 150 years? There are many causes that have been reported to be responsible for the changes described above, including: livestock grazing, climate and climate change, fire suppression, and native herbivory (Bahre 1991; McPherson and Weltzin 1998; Turner and others 2003). However, concensus seems to be that a combination of livestock grazing, climatic factors, and fire suppression are responsible for changes seen in oak woodlands. A discussion of these natural and anthropogenic disturbances follows.

Understory – We found no studies, in addition to those cited in the **Overstory** section, that documented changes for the Madrean encinal understory.

Herbaceous Layer - We found no studies that documented changes within the herbaceous layer for the Madrean encinal vegetation type.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in canopy cover for the Madrean encinal vegetation type.

Structure Class (Size Class) - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in tree size classes for the Madrean encinal vegetation type.

Life Form - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in life form for the Madrean encinal vegetation type.

Density – We found no studies, in addition to those cited in the **Overstory** section, that documented changes in tree density for the Madrean encinal vegetation type.

Age Structure - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in tree or grass age structure for the Madrean encinal vegetation type.

Patch Dispersion – We found no studies, in addition to those cited in the **Overstory** section, that documented changes in tree dispersion for the Madrean encinal vegetation type.

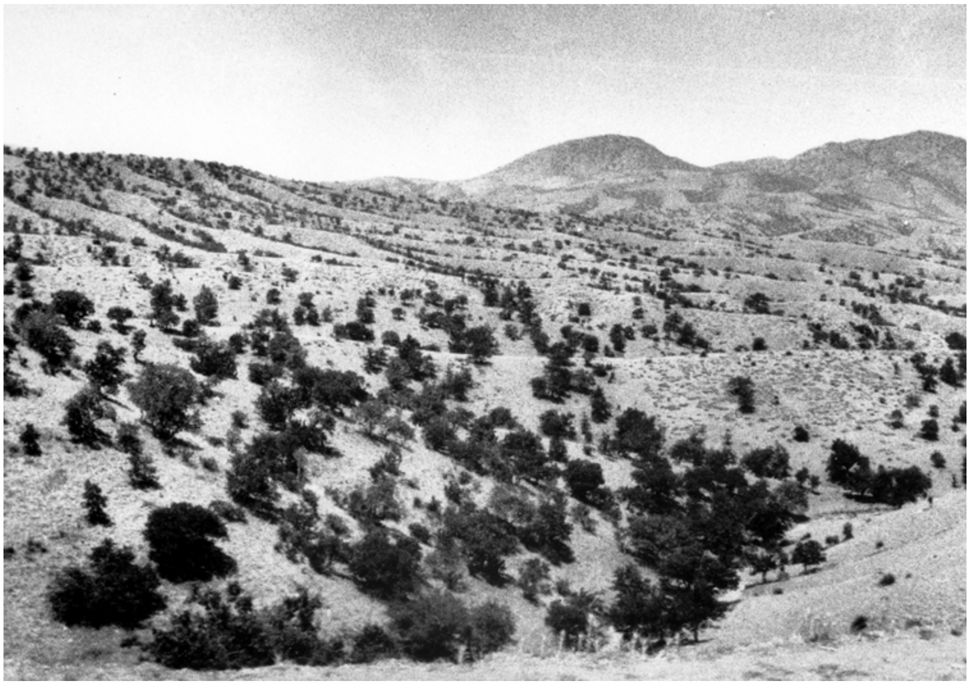

Figure 3-2. 1925 photograph (top) and 1987 photograph (bottom) of Barrel canyon near the Empire Mountains in southeastern Arizona (Photographs courtesy of Unites States Geological Survey and Turner and others 2003). Photographs show a slight increase in tree density with a large mesquite in the foreground.

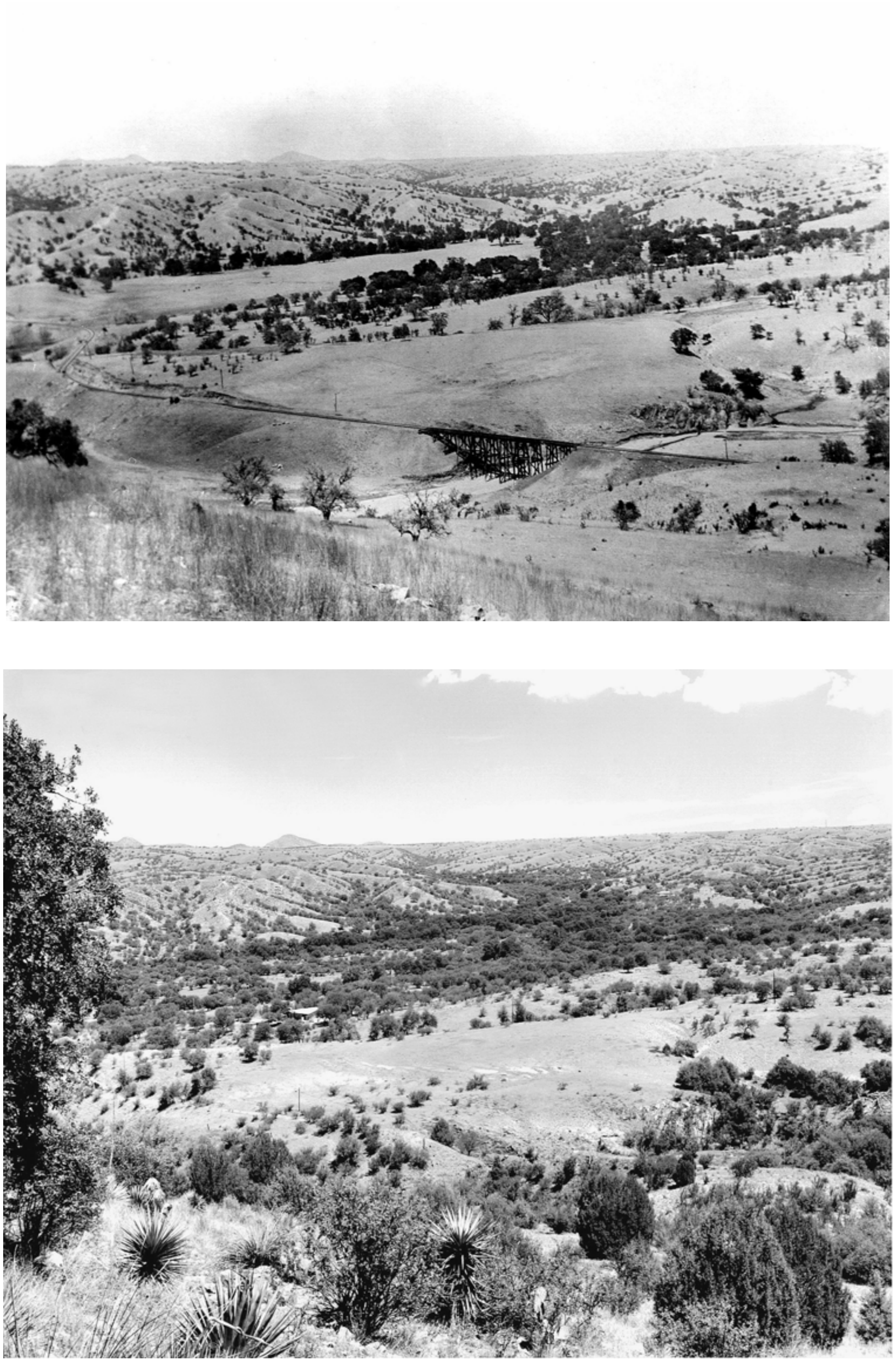

Figure 3-3. 1895 photograph (top) and 1994 photograph (bottom) of sonoita creek area in southeastern Arizona (Photographs courtesy of Unites States Geological Survey and Turner and others 2003). Photographs show an increase in tree density especially by junipers which can easily be seen in the foreground.

3.6 Madrean Encinal References

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Chapter 4 - Interior Chaparral

4.1 General Description

Arizona chaparral occurs throughout central Arizona, southwestern New Mexico, and northern parts of Mexico as a discontinuous band of vegetation. The majority of this vegetation type exists at mid elevations (3,002 ft to 6,004 ft) below the mogollon rim in Arizona and in extreme eastern Chihuahua and western Coahuila, average precipitation varies from 15 to 25 inches per year (Carmichael and others 1978; Pase and Brown 1982). It is bordered by ponderosa pine or pinyon juniper at the upper elevations, and semi-desert grassland or sonoran/mojave desert at the lower elevations (Carmichael and others 1978). Species composition and dominance varies greatly across the broad range of soils and topography that occur throughout its range. In fact, multiple researchers (Brown and Lowe 1974; Carmichael and others 1978; Darrow 1944; Swank 1958) have attempted to detail out chaparral's complex structure by grouping vegetation based on plant associations. The latest of these classifications was that carried out by Carmichael and others (1978) who broke out Arizona chaparral into 8 communities based on major plant associations within central Arizona (Table 4-1). As evidenced by Carmichael and others' (1978) classification, shrub live oak is the most common, dominant shrub within Arizona chaparral, however, a wide range of other shrubs and trees (45 species) were also found within the chaparral associations.

Table 4-1. Community associations, with scientific name of dominant shrubs, and mean elevation for each association identified by Carmichael and others (1978) for central Arizona chaparral.

Community Association	Scientific Name	Mean Elevation (ft)
Shrub live oak – birchleaf mountain mahogany	*Quercus turbinella – Cercocarpus betuloides*	3,773
Shrub live oak – mixed shrub	*Quercus turbinella* – mixed shrub	3,937
Pointleaf manzanita	*Arctostaphylos pungens*	4,265
Arizona cypress – shrub live oak	*Cupressus arizonica - Quercus turbinella*	4,429
Shrub live oak – datil yucca – yellowleaf silktassel	*Quercus turbinell – Yucca baccata – Garrya flavescens*	4,921
Yerbasanta – desert ceanothus	*Eriodictyon augustifolium – Ceanothus greggii*	4,4921
Pringle manzanita	*Archtostaphylos pringlei*	5,249
Arizona oak – yellow leaf silktassel – Emory oak	*Quercus arizonica – Garrya flavescens – Quercus emoryi*	5,577

4.2 Historic Range of Variation of Ecological Processes

Vegetation Dynamics – Interior chaparral appears to be a fairly stable vegetation type due to the majority of its species having the ability to quickly re-sprouting following disturbance events, such as fire and mechanical or chemical removal (Cable 1975; Lillie and others 1964; Pase and Ingebo 1965; Pond and Cable 1960). Additionally, the few species that regenerate from seed require fire to prepare the seedbed (Carmichael and others 1978). Historically, this led to quick recovery of chaparral following the dominant natural disturbance, fire. In current times, these same mechanisms have allowed chaparral to maintain its dense canopy cover character regardless of human disturbance.

Disturbance Processes and Regimes

Below is a discussion of the frequency, intensity, severity, seasonality, and spatial and temporal scale of disturbances that occur within the interior chaparral vegetation type.

Climate – See Chapter 1, climate analysis section.

Fire - Frequent fires, covering hundreds of square miles at a time and occurring primarily between April and June, have been well documented through direct (fire scar analysis) and indirect (ecology of dominant species) lines of evidence for the semi-desert grasslands and ponderosa pine forests that border chaparral shrublands in Arizona (Bahre 1985; Cooper 1960; Covington and Moore 1994; Dieterich 1980; Kaib and others 1996; McPherson 1995; Swetnam and Baisan 1996; Weaver 1951). Documentation regarding fire occurrence within the chaparral PNVT, however, relies most heavily on indirect information such as fire adaptation of chaparral species as well the general ecology of the system.

In particular, Carmichael and others (1978) identify chaparral as fire adapted because its deep, well developed root system allows most chaparral species to sprout rapidly following fire. Like wise, they note that non-sprouting species (desert ceanothus and manzanita) “do not germinate in the absence of heat scarification”. Based on ecological evidence, Pase and Brown (1982) and Wright and Bailey (1982) identify possible fire return interval ranges of 50 to 100 years and 20 to 80 years respectively. A quantitative study conducted by Sneede and others (2002) for the Prescott Basin within the Prescott National Forest identified an average burn interval of 30 to 40 years. In addition to knowing fires occurred somewhere on the order of every 20 to 100 years, ecologically speaking, chaparral fires are known to be high intensity stand replacing fires (Overby and Perry 1996). However, we don’t have good information regarding the size of fires that swept across this PNVT.

Hydrology – We found no studies, in addition to those cited in the *conversion to chaparral section,* that documented hydrological processes, such as flooding, as important ecological determinants for the interior chaparral vegetation type.

Herbivory - Mule deer, white-tailed deer, and black bears are key herbivores in interior chaparral. Deer eat a variety of forbs, shrubs and browse, as well as mast and other fruits (Baker 1999; Cable 1975). Conversion treatments of chaparral to grassland, were shown to increase forage for deer, elk, and cattle. However, the decrease in protective cover following conversion was also shown to negatively affect deer, especially when treatments occurred on large landscape scales (Baker 1999; Cable 1975). Additionally, cover and food for black bear is best when there are shrubs and low trees due to presence of numerous mast and fruit producing species (Baker 1999).

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and/or introductions as important ecological determinants for the interior chaparral woodland vegetation type.

Insects and Pathogens - We found no studies that implicated insects and/or pathogens as important ecological determinants for the interior chaparral vegetation type.

Nutrient Cycling - We found no studies, in addition to those cited in the *conversion to chaparral* section, that documented nutrient cycling for the interior chaparral vegetation type.

Windthrow – Not an applicable category for interior chaparral

Avalanche - Not an applicable category for interior chaparral

Erosion – We found no studies, in addition to those cited in the *conversion to chaparral section,* that documented erosion within the interior chaparral vegetation type.

4.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation – We found 5 early 1900s (1917 to 1957) photographs taken on the Apache, Gila, and Tonto National Forests (Figures 4-1 and 4-2). It is difficult to identify vegetation characteristics, other than moderate density shrub cover, from the photographs. other than the moderate cover depicting interior chaparral vegetation. We found no pre-1900 photographs or peer reviewed documentation that identified historic conditions for the interior chaparral vegetation type.

Overstory – *We found no studies that documented historic overstory conditions for the interior chaparral vegetation type.*

Understory – We found no studies that documented historic understory conditions for the interior chaparral vegetation type.

Herbaceous Layer – We found no studies that documented historic herbaceous layer conditions for the interior chaparral vegetation type.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure - We found no studies that documented historic canopy cover for the interior chaparral vegetation type.

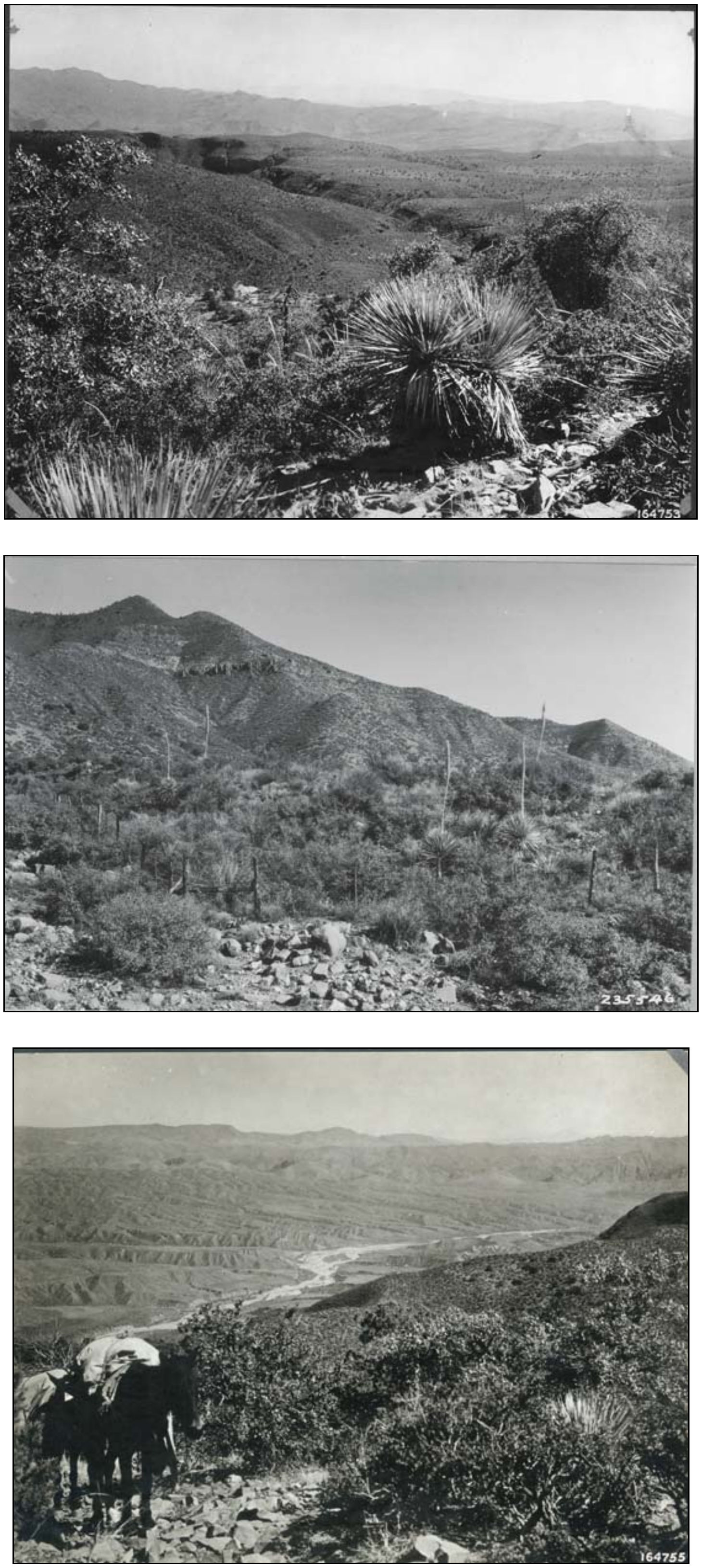

Figure 4-1. Photographs of interior chaparral taken in 1917 (top), 192? (middle), and 1957 (bottom) in the Sierra Ancha experimental range on the Tonto National Forest. Top photograph is looking at the Pinal Mountains from Mt. Baker, middle photograph is of an experimental plot, and the bottom photograph is overlooking Cherry Creek from near the summit of Sierra Ancha. Photographs courtesy of the USFS.

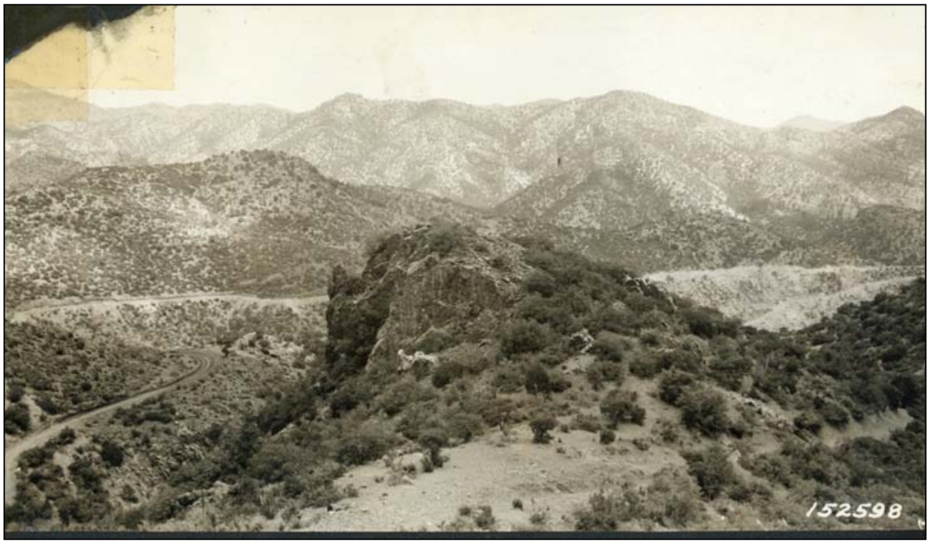

Figure 4-2. Photographs of interior chaparral taken in 1920 (top) and 1928 (bottom) on the Apache and Gila National Forests respectively. Top photograph is of the Clifton-Springerville road looking south while the bottom photograph is of the "Kneeling Nun at Santa Rita". Photographs courtesy of the USFS.

Structure Class (Size Class) - We found no studies that documented the historic structure class of trees for the interior chaparral vegetation type.

Life Form – We found no studies that documented historic life forms for the interior chaparral vegetation type.

Density - We found no studies that documented historic tree density for the interior chaparral vegetation type.

Age Structure - We found no studies that documented historic tree age structure for the interior chaparral vegetation type.

Patch Dispersion – We found no studies that documented historic patch dispersion for the interior chaparral vegetation type.

Reference Sites Used

Limitations – There is currently little information regarding chaparral vegetation near the turn of the century or for sites free from human disturbance. This definitely presents a large limitation to the extent to which historic conditions can be described.

Characteristics of Applicable Sites – Ideally, reference sites for chaparral would exist and would have intact fire regimes, be free from mechanical, chemical, or prescribed fire treatments and would include photographic documentation. Given the dense character of chaparral and the steep terrain on which it is located, identifying reference sites free from human disturbance is possible, however, finding historic site condition information for such sites is much less likely.

4.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory - Due to the steep slopes and high shrub cover (around 80 %) associated with chaparral shrubland, livestock grazing impacts have been restricted to those lower elevation sites that have both gentle slopes and relatively low shrub cover (Pase and Brown 1982). These livestock accessible sites were heavily grazed, between 1880 and 1920, and up until the 1940s they were the locations of a flourishing mohair goat industry (Pase and Brown 1982). For a more detailed discussion of the impacts of goat browsing, see Chaparral to grassland Conversion section.

Chaparral to Grassland Conversion - Around the 1950's there became a growing concern in Arizona that chaparral vegetation was taking water away from streams that could be used for agriculture and other human uses and that its' dense vegetation offered little forage value or access to wildlife or livestock (Pase and Granfelt 1977; Cable 1975).

> Chaparral in Arizona is used far below its potential. Conversions to grass can greatly increase water and grass production, and improve wildlife habitat. Management options include conversion to grass, maintaining shrubs in sprout stage, changing shrub composition, reseeding, and using goats to harvest shrub forage (Cable 1975).

These concerns resulted in 30 years of experimentation in the conversion of chaparral shrubland to a more open grassland shrub type that would use less water, be less of a fire hazard, and provide more forage for wildlife and livestock. These many experiments investigated the effectiveness of fire, herbicide, clipping, and seeding of non-native perennial grasses (Boers, Weeping, and Lehmanns lovegrasses) individually as well as in combination with each other, to eradicate chaparral vegetation. While the underlying assumption of many of these experiments, that chaparral needed to be changed in order to be beneficial to humans, is not the most ecological in nature, the results of many of these studies yielded important information regarding the ecology of this PNVT and its' dominant species.

From these studies, the most well documented characteristic of chaparral vegetation is its ability to quickly recover shrub cover to pre-disturbance levels. Multiple studies within Arizona showed that shrub cover on converted areas increased rapidly, pushing out grasses and forbs (Cable 1975; Lillie and others 1964; Pase and Ingebo 1965; Pond and Cable 1960). Studies by Pond and Cable (1960) and Lillie and others (1964) documented shrub cover to be back to pre-treatment levels within 7 years. Specifically, Pond and Cable (1960) showed that shrub live oak was very difficult to kill due to its ability to re-sprout, even 5 annual treatments of fire could not kill shrub live oak. In fact, burning, in general, increased the stem number of shrub live oak (Pond and Cable 1960). Lillie and others (1964) found that clipping of shrub live oak followed by fire lead to a slight increase in the number of stems produced, while clipping followed by chemical application of herbicide decreased stem weight, with spring time herbicide application yielding larger decreases in stem weight when compared to fall application.

Pond and Cable's work (1960) also investigated repeated fire effects on other chaparral species, they found skunkbush sumac to re-sprout erratically and suggested repeated burning appeared to be an unlikely means of eradicating this shrub species, Wrights silktassel and hollyleaf buckthorn were found to be easily killed by two years of repeat burning while desert ceanothus, manzanita, and larchleaf goldenweed were killed by just one fire, due to their inability to re-sprout (Pond and Cable 1960). Overall, Pond and Cable's work (1960) suggests that repeated burning will not get rid of the dominant shrub, shrub live oak, but can eliminate valuable forage species (Wrights silktassel and hollyleaf buckthorn) or species that do not re-sprout following disturbance. Similarly, Carmichael and others (1978) concluded that chaparral plants that reproduce through prolific production of seeds and require heat scarification to germinate (manzanita and desert ceanothus) may be lost from chaparral communities under both frequent and infrequent fire regimes. Frequent fire regimes that eliminate plants before they have a chance to produce seeds may result in the loss of seed reproducers over time, however, infrequent fire regimes may also result in loss of seed reproducers, as they are not as long lived as re-sprouting species and require fire to prepare the seed bed for reproduction (Carmichael and others 1978).

Given the tenacity of chaparral vegetation, and shrub live oak in particular, the use of repeated treatments, or a combination of treatments was experimented with in order to convert chaparral to grassland. For example, chaparral conversion on the Tonto National Forest utilized several approaches. On the Three Bar Experimental watershed, aerial application of granular karbutilate herbicide was followed by non-native seeding, while other studies used plant desiccating chemicals, followed by prescribe fire, then non-native seeding, followed by 3 years of annual herbicide application to try and keep shrub cover low and perennial grass cover high (Baldwin 1968; Pase and Ingebo 1965).

While these conversion attempts did not lead to long lasting "chaparral grasslands" in Arizona, they did result in studies of these temporary changes that generated valuable information regarding changes in water and sediment yield. Specifically, attempted conversions on the Tonto National Forest showed increased water yields ranging from 1 ½ area inches per year up to 6 area inches per year with increases in perennial flow of streams in treated watersheds (Baldwin 1969; Davis 1989; Hibbert and others 1974; Pase and Ingebo 1965). A detailed analysis of chaparral conversion on three hydrograph components for the Natural Drainages experimental watershed in central Arizona, showed increases in quick flow (30 %), peak flow (26 %), and delayed flow(32 %) (Alberhasky 1983). Additionally, in the Three Bar watershed, nitrate levels on treated watersheds fluctuated with rainfall events increasing from normal levels of 0.2 p/m to 24 p/m and 36 p/m, following storm events of 2.1 and 3.3 inches (Hibbert and others 1974). Davis (1989) found 10 times greater nitrate concentrations in streams associated with a 13 year old herbicide treated watersheds, however, prescribed fire did not cause further increases in nitrate nor did it increase sulfate, bicarbonate, or chloride anions nor calcium, magnesium, sodium, or potassium cations. The increase in nitrate was attributed to the decomposition and mineralization of a huge quantity of dead biomass followed by precipitation driven leaching (Davis 1989).

Multiple studies also showed that sediment loads also increased following shrub removal. Specifically, Pase and Ingebo (1965) found sediment transport to be 0.02 acre feet before a fire and between 5.9 and 13.74 acre feet following a fire in a conversion watershed, with sediment transport returning to pre-treatment levels within 4 years. Heede and others (1988) also found that large amounts of sediment are moved into stream channels within chaparral watersheds following fire, due to the complete denuding of vegetation within a steeply sloping environment. However, rapid recovery of vegetation created buffer strips was found to greatly reduce sediment loss from slopes. In particular, they found that following a 1959 fire erosion pavements had the highest sediment delivery (average of 1470 kg/ha/yr) to stream channels while buffer strips experienced low sediment delivery (average 5 kg/ha/yr) (Heede and others 1988). Sediment movement to the channels caused an aggradation event followed by degradation within the channels that continued through 1985. Ultimately, Heede and others (1988) suggested that to avoid severe erosion following fires that work should focus on establishing vegetation buffers along channel banks.

More recent studies have looked at the effects of fire on water repellency and nutrient cycling within chaparral. In regards to water repellency, Brock and DeBano (1988) found that water repellency in chaparral soils varies both horizontally and vertically within the soil profile and exists both before and after burning; however, fire greatly increases overall water repellency in the soil. Changes in repellency can result in increased soil erosion and can prevent the wetting of microsites which are necessary for seed germination (Brock and DeBano 1988). Prescribed fires were also found to have an effect on nutrient cycling within chaparral soils. Overby and Perry (1996) found that nitrogen and phosphorus concentrations increased following prescribed fires, with birchleaf mountain mahogany dominated sites having greater increases over shrub live oak dominated sites due to higher liter accumulation and higher nutrient concentrations within birchleaf mountain mahogany tissues. Specifically, exchangeable NH4+-N increased from 5.39 to 71.62 mg/kg on birchleaf mountain mahogany sites and from 5.23 to 36.10 mg/kg on shrub live oak sites; extractable P increased from 5.8 to 22.62 mg/kg on birchleaf mountain mahogany sites and from 5.46 to 14.58 mg/kg on shrub live oak sites. This release of nutrients increases soil fertility for seedlings, re-sprouting species, and soil micro-organisms.

In an effort to find a more “natural” solution to the use of chemicals to convert chaparral to grassland, multiple studies were conducted on the effectiveness of goats as shrub cover decreasers. While studies by Severson and DeBano (1991) and Knipe (1983) both found goats to be effective at decreasing shrub cover, they also noted some detrimental impacts of goat browsing. Specifically, Knipe (1983) noted that due to the penning of goats, overuse of browse in what he referred to as “sacrifice zones” was high and impacted forage most palatable to wildlife (mountain mahogany and Wright’s silktassel). Similarly, Severson and DeBano (1991) showed that forage most heavily grazed by goats was also the most palatable deer forage and noted that this pattern of use could result in the loss of these species which would “reduced forage diversity and [cause] nutritional stress” to livestock and wildlife. Additionally, they noticed that litter levels were statistically decreased under desert ceanothus plants due to goat browsing and this, in combination with trampling of the nitrogen fixing plants by goats, decreased nutrients under the shrub canopy which they hypothesized could have long term impacts on nutrient cycling within chaparral (Severson and DeBano 1991).

Fragmentation – We found no studies that documented the effects of fragmentation on the interior chaparral vegetation type.

Mining - We found no studies that documented the effects of mining in the interior chaparral vegetation type.

Fire Management – Given the relatively less frequent and broad fire return interval (20 to 100) that chaparral shrublands are adapted to, the last 120 years of fire suppression has had less effect on chaparral than frequent fire regime adapted vegetation types. In general, structural changes, such as changes from a grass dominated structure to a shrub dominated one, have not occurred within chaparral. The only change that has been documented within this PNVT is an increase in shrub cover densities within already existing chaparral stands (Huebner and others 1999; Huebner and Vankat 2003). For a detailed discussion of their studies see section 4.5, **Overstory.**

Exotic Introductions (Plant & Animal) - We found no studies that documented the effects of exotic introductions on the interior chaparral vegetation type. However, there is documentation of the seeding of non-native perennial grasses, such as Lehmann and Boer’s lovegrasses, in an effort to convert chaparral to grassland (Hibbert and others 1974). While seeding was effective in some areas, grasses only remained until shrub cover had returned to pre-disturbance levels, hence while non-native perennial grasses may be present, they do not dominate areas or effectively change chaparral vegetation (Hibbert and others 1974).

4.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory - On a landscape scale, there has been little change in chaparral shrublands since 1880, even with the unsuccessful 1950's to 1980's era attempts to convert these shrublands to grasslands. Studies by Huebner and Vankat (2003) and Huebner and others (1999) in central Arizona have investigated vegetation change within the chaparral shrublands and its associated grasslands and found that little change has occurred within this vegetation type. Huebner and others (1999) study investigated changes within 3 chaparral classes (chap 75 = ≥ 75 % cover, chap 50 = ≥ 50 % and < 75% cover, and chap 25 = ≥ 25 % and < 50 % cover), chaparral associated grasslands (≥ 75 % cover) and 2 juniper woodland classes (jun 50 = ≥ 50 % and < 75% cover, and jun 25 = ≥ 25 % and < 50 % cover). The results for the chaparral and chaparral associated grasslands showed that 93% of dense chaparral (≥ 75 % cover) and 75 % of chaparral associated grasslands (≥ 75 % cover) were unchanged between the years 1940 and 1989 (Huebner and others 1999). They also noted a change in the density of chaparral with a decline in the least dense class, chap 25, and an increase in the densest chaparral, chap 75, with moderately dense, chap 50, showing no change (Huebner and others 1999). Additionally, in Huebner and Vankat's (2003) investigation of environmental and disturbance factors associated with vegetation change in chaparral, they show environmental factors to be the most important determinants in creating chaparral, chaparral associated grassland, juniper woodland, and juniper woodland grassland with disturbances such as fire and livestock grazing not playing a role in differentiating the chaparral types. Additionally, they note that the changes seen in chaparral associated grasslands are the result of non-chaparral trees (junipers, mesquite, and acacia) suggesting that chaparral is a stable vegetation type (Huebner and Vankat 2003).

On a species level, it is quite possible that changes have occurred within chaparral shrublands. A repeat photography study, by Pond (1971), of individual chaparral plants between 1920 and 1967 showed that longevity of individual chaparral species varied. Shrub live oak, skunkbush (*Rhus trilobata*), manzanita (*Archtostaphylos pungens*), and wait-a-bit bush (*Mimosa biuncifera*) all survived the 47 year study, through vegetative growth; however, sacahuista (*Nolina microcarpa*) and desert ceanothus (*Ceanothus greggii*) died sometime between 1935 and 1967 (Pond 1971). It is important to note that other sacahuista and desert ceanothus plants were found near the dead photographed plants and were presumed to be the offspring of original plants (Pond 1971). Given that all of the plants, except manzanita, that survived the 47 year study are sprouters following disturbance and the two plants that did not survive reproduced via seed, it seems likely that species composition and cover change little over time without disturbance. Cable (1975) documented a change in chaparral species composition from a pre-burn dense manzanita community, with minor amounts of shrub live oak and desert ceanothus, to a post-burn community consisting of narrow leaf yerbasanta, Pringle manzanita, desert ceanothus, deerbrush, true mountain mahogany, yellowleaf silktassel, emory oak, and shrub live oak seedlings. This suggests that while chaparral vegetation is relatively stable, disturbance events can change the species composition of an area.

The lack of structural changes in chaparral vegetation is likely due to three main factors: 1) Chaparral vegetation quickly regenerates shrub cover following disturbances (fire, clipping, and herbicide), hence conversion attempts have proven to be unsuccessful at eliminating these shrublands; 2) Chaparral is little effected by livestock grazing as the density of chaparral and steep slopes make the majority of this vegetation type unusable by livestock; 3) Due to its adaptation to a less frequent and broad fire return interval (20 to 100), 120 years of fire suppression has not greatly changed its historic fire regime.

Understory – We found no studies, in addition to those cited in the **Overstory** and **Density** sections, that documented changes within the interior chaparral understory.

Herbaceous Layer - We found no studies that documented changes within the interior chaparral herbaceous layer.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure - We found no studies, in addition to those cited in the **Overstory** section, that documented canopy cover changes within the interior chaparral vegetation type.

Structure Class (Size Class) - We found no studies that documented changes in tree size classes within the interior chaparral vegetation type.

Life Form - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in life form within the interior chaparral vegetation type.

Density – We found no studies, in addition to those cited in the **Overstory** section, that documented changes in tree density within the interior chaparral vegetation type.

Age Structure – We found no studies, in addition to those cited in the **Overstory** section, that documented changes in age structure within the interior chaparral vegetation type.

Patch Dispersion – We found no studies, in addition to those cited in the **Overstory** section, that documented changes in patch dispersion within the interior chaparral vegetation type.

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Chapter 5 – Madrean Pine Oak Woodlands

5.1 General Description

Madrean Pine-Oak woodland occurs throughout southeastern Arizona, southwestern New Mexico, northeastern Sonora, and northwestern Chihuahua at elevations ranging from 4,600 to 7,500 ft (McPherson 1992). These pine-oak woodlands are bounded by oak encinal and semi-desert grasslands at the lowest elevations and montane forests at the higher elevations. Oak species vary throughout the range of pine-oak woodlands. In Arizona, Emory oak (*Quercus emoryi*), Arizona white oak (*Quercus arizonica*), Mexican blue oak (*Quercus oblongifolia*), silverleaf oak (*Quercus hypoleucoides*), netleaf oak (*Quercus rugosa*), Gambel oak (*Quercus gambelii*), and Toumey oak (*Quercus toumeyi*) are the common species. In New Mexico, gray oak (*Quercus grisea*) and Emory oak are common while encino roble (*Quercus coccolobifolia*), encino blanco (*Quercus laeta*), and encino prieto (*Quercus sideroxyla*) are the common species in Mexico (Abbott 1998; Fulé and Covington 1996, 1998; Fulé and others 2005). Conifer species also vary, with Douglas fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), Chihuahuan pine (*Pinus leiophylla* var. *chihuahuana*), apache pine (*Pinus engelmannii*), and border pine (*Pinus discolor*) as the common conifers of Arizona and New Mexico while dominant species in northern Mexico include pino blanco (*Pinus ayacahuite*), Durango pine (*Pinus durangensis*), Chihuahua pine, pino triste (*Pinus lumholtzi*), Douglas fir, and *Pinus herrerai* (Fulé and Covington 1996, 1997, 1998; Fulé and others 2005; Kaib and others 1996; Swetnam and others 1992).

Additionally, pine and oak species vary along an elevational gradient. In Arizona, oaks sampled along a gradient from 3800 to 7600 ft showed Mexican blue oaks and Emory oaks dominate the lower elevations (below 6560 ft) whereas silverleaf oak and Arizona white oak are more common above 4590 ft (Abbott 1998). For pines, border pinyon pine is found above 5250 ft and Apache and ponderosa pine are common above 5900 ft; the elevational sequence of pines from low to higher elevation is: border pinyon, Chihuahua, Apache, and ponderosa pine (Abbott 1998; Whittaker and Niering 1964, 1965, 1968). Chaparral species (such as point-leaf manzanita, Wright silktassel, ceanothus, skunkbush sumac, Mearn sumac, catclaw acacia, pringle manzanita, mountain mahogany, cliffrose, Toumey oak, birchleaf buck thorn, holly-leaf buckthorn, and rosewood), juniper species, and perennial bunch grasses (such as Texas blue stem (*Andropogon cirratus*) threeawns, sideoats grama, blue grama, hairy grama (*Bouteloua hirsuta*), cane beardgrass (*Bothriochloa barbinodis*), plains lovegrass (*Eragrostis intermedia*), tanglehead (*Heteropogon contortus*), green sprangle top (*Leptochloa dubia*), wolftail (*Lychurus pheloides*), bull grass (*Muhlenbergia emersleyi*), and crinkleawn (*Trachypogon montufari*) are also common species found throughout these pine-oak woodlands (Abbott 1998).

5.2 Historic Range of Variation of Ecological Processes

Vegetation Dynamics – Madrean pine oak woodlands have been shown to trend away from woodlands with an open pine dominated overstory and oak understory (Figure 1) to dense woodlands with oaks and other sprouting tree species dominating the overstory (Figure 2) (Barton 2002; Barton and other 2001; Fulé and Covington 1998). This change in density and species composition is the result of switching from centuries of predominantly low intensity frequent surface fires to decades without fire followed by stand replacing fires (Barton 2002; Fulé and Covington 1998). Frequent low intensity surface fires allow fire resistant pines to dominate a site and maintain an open stand structure. Additionally, short fire-free periods, between 20 and 30 years in length, are necessary to allow for periodic pine regeneration (Barton and others 2001). The recent shift in fire regimes has been more conducive to the reduction of pine species and regeneration of oak and other sprouting species.

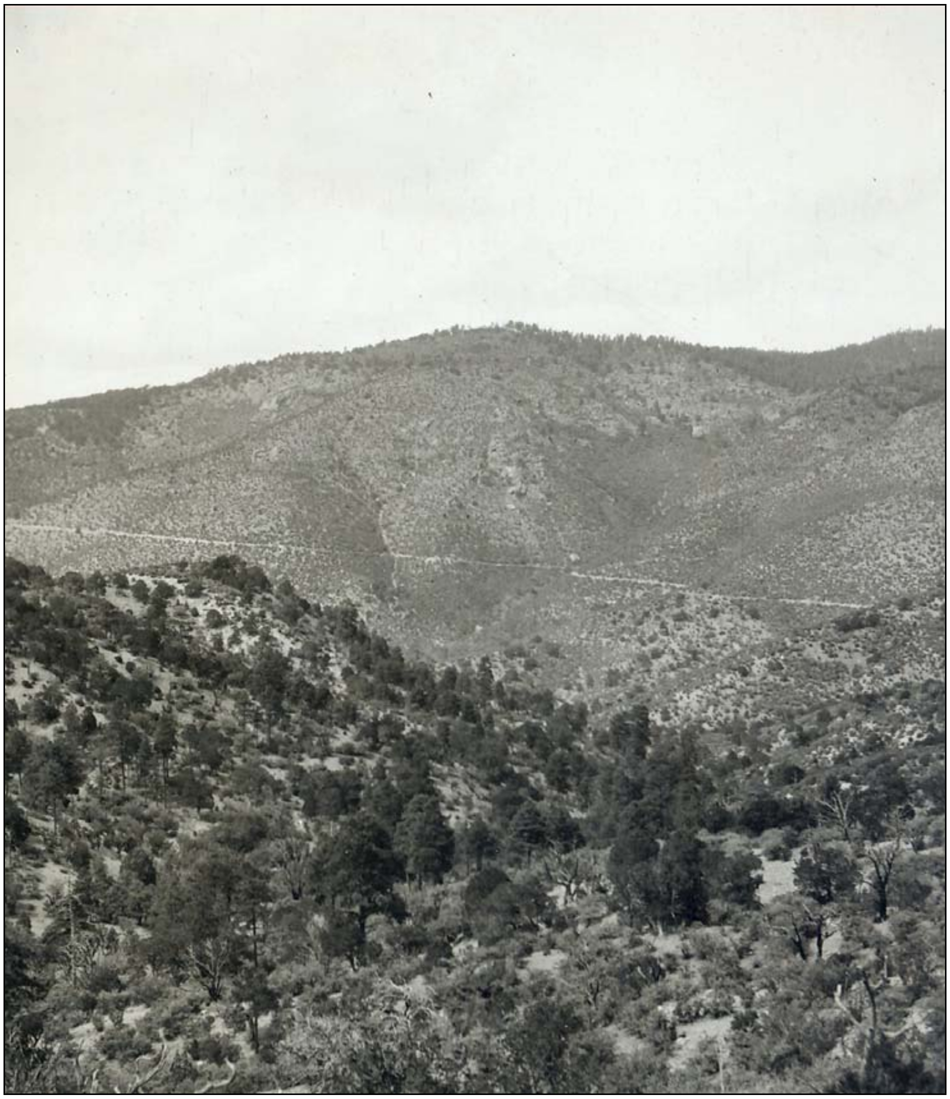

Figure 5-1. Photograph of open pine oak woodland taken in 1913 on the south end of the Black range in the Gila National Forest, New Mexico. Photograph courtesy of USFS Region 3.

Figure 5-2. Photograph of Cathedral Rock in the Chiricahua Mountains in southeastern Arizona. Photograph on the left taken in 1907, photograph on the right (taken slightly above 1907 spot in order to see Cathedral Rock) 2006. Repeat series shows an increase in the oak understory and Pine overstory such that the view of Cathedral rock is obscured. 1907 photograph courtesy of Coronado National Forest, 2006 photograph courtesy of Paul Hirt.

Disturbance Processes and Regimes

Below is a discussion of the frequency, intensity, severity, seasonality, and spatial and temporal scale of disturbances that occur within the Madrean pine oak woodland vegetation type.

Climate – See Chapter 1, climate analysis section.

Fire - It is well documented through fire scar data that prior to about 1880, low intensity surface fires frequently swept through the Madrean pine-oak woodlands between early spring and summer, primarily as a result of lightning ignitions (Fulé and others 2005; Kaib and other 1996; Swetnam and Baisan 1996; Swetnam and others 1992; Swetnam and others, in press). The continuity of fire scars between lower and higher elevations suggests that fire sizes were large enough to span the entire elevational distribution of the pine-oak woodlands and that fire ignitions coming from semi-desert grasslands and/or montane conifer systems were not uncommon (Fulé and Covington 1997, 1998; Fulé and others 2005; Kaib and other 1996; Swetnam and others 1992). Mean fire return intervals (MFRI) for all fires at 17 sites located in Arizona and northern Mexico ranged between 3 and 9 years, while the MFRI for fires that scarred 25% of the trees ranged between 6 and 14.5 years. The minimum and maximum number of years between any two consecutive fires was between 1 and 38 years (Fulé and Covington 1996, 1997, 1998, 1999; Fulé and others 2005; Kaib and other 1996; Swetnam and Baisan 1996; Swetnam and others 1992; Swetnam and others, in press).

At six pine-oak sites in Durango, northern Mexico, Fulé and Covington (1996, 1997, 1999) reported that 60 to 80% of wildfires before the mid-20th century burned in the spring and the remainder burned in the summer, similar to patterns in the southwestern United States (Swetnam and Baisin 1996). A possible connection between climate and fire occurrence was suggested at 3 of the sites by the fact that 7 of 10 regional fire years corresponded to (dry) positive Southern Oscillation Index (SOI) extremes while no regional fire years occurred during (wet) negative SOI extremes (Fulé and Covington 1996). Similarly, at the other 3 sites, negative SOI extremes were consistently associated with relatively low-fire years (i.e., no regional fires), however, the association between positive SOI extremes and high-fire years was weak (Fulé and Covington 1999). Positive SOI extremes have been correlated with major regional fire years in the southwestern United States (Swetnam and Betancourt 1990).

While knowing the mean fire return interval for a PNVT is important, understanding the variability around the mean is equally important. Research by Barton and others (2001) found that in Madrean pine-oak woodlands, fire-free periods of between 20 and 30 years in length were necessary to allow Arizona pines to establish, while subsequent years of frequent fire were responsible for allowing mature pines to remain dominant. In addition, Barton (1993) explored the factors affecting the distribution of three elevationally stratified pine species (Apache pine – 1500 to 2000 m, Chihuahua pine – 1600 to 2100 m, and border pinyon – 1900 to 2200m) in southeast Arizona. He found that the lower elevational limits of all three were controlled by dry season water stress while the upper elevational limits were controlled by light limitations (Chihuahua pine), less frequent fire than at higher elevations (Apache pine), and equivocal results for border pinyon although this species was shown to expand upslope in response to the lack of fire (Barton 1993). It is important to keep in mind factors such as fire, fire return and its interactions with climate and topography affecting species distributions as changes can have drastic effects on species composition and structure. Changes in fire patterns have already resulted in shifts in dominant vegetation in the Madrean pine-oak woodlands of Arizona and Mexico (Barton 1999; Fulé and Covington 1998). For a detailed discussion of vegetation change, see section 5.5, **Overstory**.

Hydrology – We found no studies that documented hydrological processes, such as flooding, as important ecological determinants for the Madrean pine oak woodland vegetation type.

Herbivory - Coues white-tailed deer and rodents such as yellow nosed cotton rat, white-throated wood rat, southern pocket gopher, Apache squirrel, Bailey's pocket mouse, and eastern cotton tail are common in the Madrean pine-oak woodlands (Abbott 1998; Evans 1984 in Ockenfels 1995; Findley 1975 in Ockenfels 1995). Rodents, in particular, have been shown to have large impacts on the structure and composition of vegetation in grassland and desert ecosystems in the southwest (Brown and Heske 1990; Finch 2004; Miller 1994; Weltzin and others 1997; Van Devender 1995), however, there is no documentation of vegetation change or manipulation by rodents within the Madrean pine-oak woodlands.

On the other hand, information regarding Coues white-tailed deer diet, level of resource competition with livestock, and their impacts on the Madrean evergreen vegetation (Madrean encinal and Madrean pine-oak woodland PNVTs) is well documented. Results of diet studies suggest that Coues white-tailed deer primarily browse on shrubs, eating forbs when abundant, utilize grass infrequently and prefer rugged brushy terrain (Day 1964 in Ockenfels 1995; Findley 1975 in Ockenfels 1995; Gallina and others 1981 in Ockenfels 1995; Gallina 1984 in Ockenfels 1995; Hoffmeister 1986 in Ockenfels 1995). Information such as this collected by Gallina (1984 in Ockenfels 1995) at La Michilia Biosphere Reserve, Mexico, led the researcher to conclude that there was little resource competition between Coues white-tailed deer and livestock. However, other researchers suggested resource competition did exist. Specifically, Galindo-Leal and others (1993), also doing studies in La Michilia Biosphere Reserve, Mexico showed that Coues white-tailed deer avoided areas heavily grazed by livestock, while Knipe (1977 in Ockenfels 1995) and Evans (1984 in Ockenfels 1995) noted an overlap between livestock and white-tailed deer diets (primarily forbs) during times of drought. Knipe (1977 in Ockenfels 1995) also noted white-tailed deer's need for high grass cover during the fawning season. In addition, in the 1930's and 1950's, Arizona biologists thought that white-tailed deer numbers were too high and attributed range degradation and overuse of key browse species (including mountain mahogany (*Cercocarpus breviflorus*) and deer ceanothus (*Ceanothus greggii*)) to their high population numbers (Day 1964 in Ockenfels 1995; Day and Gallizioli 1967 in Ockenfels 1995).

While the level of resource competition between Coues white-tailed deer and livestock is unclear, it is clear that Coues white-tailed deer did/do have an impact on browse species and would have been responsible for decreasing shrub cover, to some extent, in Madrean pine-oak woodlands. In contrast, livestock decrease grass cover and so their impact on the Madrean pine-oak woodland would be different than that of white-tailed deer, the dominant native herbivore.

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and/or introductions as important ecological determinants for the Madrean pine-oak woodland vegetation type.

Insects and Pathogens - We found no studies that implicated insects and/or pathogens as important ecological determinants for the Madrean pine oak woodland vegetation type.

Nutrient Cycling - We found no studies that documented nutrient cycling within the Madrean pine oak woodland vegetation type.

Windthrow - We found no studies that implicated windthrow as an important ecological determinant for the Madrean pine oak woodland vegetation type.

Avalanche - We found no studies that implicated avalanche as an important ecological determinant for the Madrean pine oak woodland vegetation type.

Erosion **–** We found no studies that documented erosion within the Madrean pine oak woodland vegetation type.

5.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation **–** We found no historical photographs or documentation that identified historic conditions in the Madrean pine oak vegetation type.

Overstory **–** We found no studies, in addition to those cited in the *Patch or Stand Structure of Vegetation* section, that documented the historic overstory conditions within the Madrean pine oak woodland vegetation type.

Understory – We found no studies that documented the historic understory conditions within the Madrean pine oak woodland vegetation type.

Herbaceous Layer **–** We found no studies that documented the historic herbaceous layer conditions within the Madrean pine oak woodland vegetation type.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure - We found no studies, in addition to those cited in the **Density** section, that documented historic canopy cover within the Madrean pine oak woodland vegetation type.

Structure Class (Size Class) **–** We found no studies, in addition to those cited in the **Density** and **Age Structure** sections, that documented the historic structure class of trees within the Madrean pine oak woodland vegetation type.

Life Form **–** We found no studies, in addition to those cited in the **Density** section, that documented the historic life forms within the Madrean pine oak woodland vegetation type.

Density - We identified one study that documented tree density at sites in northern Mexico with undisturbed (frequent fire) fire regimes (Fulé and Covington 1996, 1997, 1998). While this study looks at current tree density, given that the historic fire regime is intact, results of this study document tree densities under a historic fire regime and hence are valuable in understanding historic conditions. Results showed that live tree density (trees/ha) and basal area (m^2 / ha), for trees over and under 1.3 m as well as for dead trees over 1.3 m, varied by species with pines accounting for the most trees with the greatest basal area, followed by oak species and then by junipers, alders, and madrones, Table 5-1 (Fulé and Covington 1996, 1997, 1998). Regeneration of young pines and oaks was approximately equal and comprised 89% of the 8,725 seedlings/ha recorded. This high regeneration density compared with the low overstory density is consistent with the thinning effect of fires which appear to reduce the density of trees surviving to the overstory. Information on density of trees by quadratic mean diameter classes shows between roughly 75 and 225 trees/ha in the 20 to 50 cm class and a diameter distribution that, though positively skewed, appeared more normally distributed than fire-excluded sites (Fulé and Covington 1997, 1998). Additionally, their study documented a uniform distribution of live trees (>1.3 m) at scales between 2 and 12 m (Fulé and Covington 1998). Dead woody biomass was distributed between diameter classes ranging from 0.6 cm to > 7.6 cm for a total of 10.6 metric tons/ha. Herbaceous cover averaged 20% (Fulé and Covington 1997).

Table 5-1. Tree densities from stem mapped plots at a frequent fire site near Durango Mexico, data taken from Fulé and Covington's (1998) evaluation of spatial structure in Madrean pine-oak forests with varying fire regimes. See Fulé and Covington (1996, 1997) for other summaries of these data.

Size Class	Species Group	Density (trees/ha)	S.E.M	Basal Area (m^2 / ha)	S.E.M
Live > 1.3 m	Pine	404	13.7	42.5	3.2
Live > 1.3 m	Oak	200	24.0	6.1	0.9
Live > 1.3 m	Juniper, Alder, Madrone	28	17.7	0.5	0.3
Live > 1.3 m	**Total**	**632**	**28.1**	**49.2**	**3.3**
Live < 1.3 m	Pine	15080	2289.6	Not reported	Not reported
Live < 1.3 m	Oak	2240	611.9	Not reported	Not reported
Live < 1.3 m	Alder	156	83.5	Not reported	Not reported
Live < 1.3 m	Juniper, Madrone	116	27.2	Not reported	Not reported
Live < 1.3 m	**Total**	**17,592**	**2514.6**	**Not reported**	**Not reported**
Dead > 1.3 m	Pine	132	56.7	8.5	5.5
Dead > 1.3 m	Oak	36	13.7	0.9	0.7
Dead > 1.3 m	Alder	4	6.5	0.0	0.0

Dead > 1.3 m	Juniper, Madrone	16	4.0	0.2	0.1
Dead > 1.3 m	**Total**	**188**	**59.3**	**9.6**	**5.3**

Age Structure - There were three studies that document historic age structure in southeastern Arizona pine oak woodlands (Barton 1999; Barton and others 2001; Danzer and others 1996). Results of the studies showed tree establishment to be tied to release from frequent fire. Specifically, Barton and others (2001) documented three recruitment pulses (identified by the number of Arizona pine stems germinating per decade from 124 samples in middle Rhyolite Canyon, and 105 samples from lower Rhyolite Canyon, Chiricahua Mountains, Arizona), one between 1610 and 1640 (for middle Rhyolite Canyon), the second between 1810 and 1830 (for middle and lower Rhyolite Canyon), and the third between 1870 and 1900 (for middle and lower Rhyolite Canyon). Each recruitment pulse occurred during a 20 to 30 year low fire frequency period preceded by multiple decades of frequent fire. Stem numbers for the 1600 and early 1800 recruitment pulses ranged between 3 and 10 stems for both middle and lower Rhyolite Canyon while stem numbers ranged between 19 and 30 plus for the late 1800 pulse with non-pulse stem numbers ranging between 0 and 2 stems (Barton and others 2001).

Barton (1999) presents historic age structure data for Chihuahua and Apache pine and silverleaf oak in lower Rhyolite Canyon. Pines established throughout the 1800's except between 1800 and 1810 and survived the high fire frequency period of 1851-1872. In contrast, only one silverleaf oak individual established prior to 1850 (1847) and only a few individuals date from before the last fire during the period of high fire frequency, a pattern that is consistent with the different responses of pines vs. oak to fires.

Similarly, Danzer and others (1996) also documented the connection between tree establishment and fire events on six 100-meter pine-oak transect plots from Garden canyon near Fort Huachuca, Arizona. This study shows a large increase in the number of ponderosa pines following a stand replacing fire in 1899, with number of trees increasing from roughly 1 to 20 individuals between 1750 and 1890 to between 20 and 60 individuals by 1930 and just under 80 individuals by 1990 (Danzer and other 1996). Additionally, Danzer and others (1996) also noted the establishment of Douglas fir, southwestern white pine and Gambel oak between 1910 and 1950 due to the cessation of fire. Douglas fir, southwestern white pine and Gambel oak numbers were low between 1750 and 1910 with numbers of trees for each species ranging roughly between 0 and 5 individuals, increasing to between roughly 10 and 20 trees for the 1910 to 1990 time span (Danzer and others 1996).

Both of these studies show the episodic recruitment of trees during low fire frequency times, showing unusually large increases of trees after the cessation of fire circa 1900.

In addition, one study documents the historic age structure of pines over 6 cm dbh (Fulé and Covington 1997). The oldest tree at the site recruited during the early 1700's and the majority of trees were less than 100 years old in the 1990's.

Patch Dispersion – We found no studies, in addition to those cited in the **Density** section, that documented the historic patch dispersion of trees within the Madrean pine oak woodland vegetation type.

Reference Sites Used

Limitations – Historic information regarding the Madrean pine oak woodlands comes from many (11) dendrochronology studies that range in location from southeastern Arizona to northern Mexico (Barton 1993; Barton 1999; Barton and others 2001; Danzer and others 1996; Fulé and Covington 1996, 1997, 1998; Fulé and others 2005; Kaib and other 1996; Swetnam and Baisan 1996; Swetnam and others 1992; Swetnam and others, press). In regards to the discussion of fire return intervals, there is a good deal of information from many locations throughout the Madrean pine oak woodland range. In contrast, few of these studies investigated stand structure and or age structure, hence information on the HRV of these characteristics comes from only 3 studies with two of the studies utilizing dendrochronology information while the third study uses current information from a reference site with an intact historic fire regime (Barton and others 2001; Danzer and others 1996; Fulé and Covington 1997, 1998). While information regarding some aspects of the HRV is well documented, other information comes from a narrower set of studies and may not capture the range of variation historically seen.

Characteristics of Applicable Sites – With the use of dendrochronology, the need for studies to be conducted at reference sites is less important as information from historic trees can still be gathered. However, information from reference sites with intact historic processes, such as fire and herbivory regimes, are in valuable as they yield information on vegetation characteristics not available from tree rings, such as information on non-pine species. Increasing the number of reference sites that have intact historic fire regimes in the United States (if possible) and Mexico would greatly increase our understanding of this vegetation type.

5.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory - There are no studies that look at the direct effects of livestock grazing in the Madrean pine-oak woodlands. However, the cessation of widespread fires throughout the southwest, and in the Madrean pine-oak woodlands in particular, in the late 1880’s has been attributed to the lack of fine fuels available due to heavy livestock grazing (Bahre 1991; Fulé and others 2005; Swetnam and others 1992; Swetnam and Baisan 1996); Swetnam and Betancourt 1998). For a discussion on the effects of fire suppression on vegetation see *Vegetation Change Over Time* section.

Silviculture - Fuel wood cutting for mining and domestic use in Madrean pine-oak woodlands was common in southeastern Arizona until the late 1800’s, and is still common in northern Mexico today (Bahre 1991, Webster and Bahre 2001). In southeastern Arizona, mesquite, oak, and juniper at low elevations, along with pines from higher elevations were the dominant species cut for fuel wood and timber for mining and domestic uses. Intense harvesting in and around mining towns in southeast Arizona between 1870 and the 1890’s (peak years) had a devastating effect on trees in the surrounding watershed (Bahre 1991). For example, Bahre (1991) estimated that the total amount of fuel wood cut from the Tombstone watershed between 1878 and 1940 was “more than the total cordage presently reported by the Forest Service for the evergreen woodlands of the Tombstone woodshed”. While it is certain that this short duration, high intensity utilization of woodlands had a devastating effect on pine-oak woodlands, the disturbance was highly variable and predominantly affected watersheds near mining towns (Bahre 1991: Turner and others 2003). In addition, based on repeat photography Turner and others (2003) concluded that while fuel wood harvesting had a dramatic effect, its "influence was local and transient".

Fragmentation – We found no studies that documented the effects of fragmentation on the Madrean pine oak woodland vegetation type.

Mining - We found no studies, except those cited in the S*ilviculture* section, that documented the effects of mining in the Madrean pine oak woodland vegetation type.

Fire Management – Passive fire suppression through livestock grazing in the late 1800's through today, as well as active suppression, which has been increasing over the last 100 years in the southwestern United States, has resulted in drastically reduced fire return intervals and increased fire severity in Madrean pine-oak woodlands (Barton 1999; Swetnam and Baisan 1996). Specifically, the last widespread fire that occurred in 4 study sites in the United States was between 1876 and 1914 (Swetnam and Baisan 1996). In Mexico, intense livestock grazing as a result of ejido land grants resulted in reduced fires in some areas between 1930 and 1960. Livestock grazing is still on-going today, but occurs only patchily throughout the landscape (Fulé and others 2005). Mexico's active fire suppression efforts are also patchy as they lack a coordinated and well funded fire suppression program similar to that seen in the United States (Fulé and others 2005; Swetnam and Baisan 1996). As a result of these factors, the last year of widespread fire identified at 11 northern Mexico sites ranged from 1945 to 1995. (Fulé and Covington 1996, 1998, 1999; Fulé and others, 2005). This difference in fire suppression activities between the United States and northern Mexico has led to differences in the structure of Madrean pine-oak woodland vegetation. For a detailed description of these differences and vegetation change over all, see section 5.5, **Overstory**.

Exotic Introductions (Plant & Animal) - We found no studies that documented the effects of exotic introductions on the Madrean pine oak woodland vegetation type.

5.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory - Pre-European settlement (before 1880), the dominant disturbance process in the Madrean pine-oak woodland was low intensity surface fire, hence much of what is known regarding vegetation change over time is the result of scientific studies and survey comments that compare vegetation structure and composition for areas with differing fire regimes. The results of these studies and survey accounts give a pre-1880 picture of Madrean pine-oak woodlands with widely spaced pines and oaks, with pines dominating the overstory and abundant perennial bunch grasses covering the ground (Barton and others 2001; Fulé and Covington 1998; Swetnam and others 1992). The changes that have occurred as a result of fire suppression are an increase in the dominance of oak and other sprouting species, such as alders, as well as the reduction of perennial bunch grasses, resulting in structurally denser vegetation with higher fuel loads (Barton 1999; Fulé and Covington 1998). Specifically, Fulé and Covington's (1996, 1977, 1998) study of spatial patterns and structure of Madrean pine-oak woodland under differing fire regimes near Durango, Mexico found that mean forest density varied from 647 trees ha $^{-1}$ at a site with a frequent fire regime (FF) to 2733 trees ha $^{-1}$ at a site excluded from fire (FE) from 1945 to 1994. They also found sprouting alders present in the FE site and in a site where fire was excluded for 29 years followed by fire return (FR); alders comprised 12 % and 20 % of total tree density respectively at these sites. In contrast, the site with a frequent fire (FF) regime was alder free (Fulé and Covington 1998). In addition, understory species differed between the sites, with the FF regime site dominated by pines and non-oak species (86 % of total) while the FE and FR sites were dominated by sprouting species (68 % and 86 % respectively of total tree density). Although regeneration density at FE, FR, and FF sites was similar, the density of oak seedlings was proportionately greater at the FE and FR sites (Fulé and Covington 1996, 1977, 1998).

The compositional shift in pine oak woodland following fire suppression may be related to the different recruitment dynamics of oaks (sprouting species) and pines. Barton (1995) found that increasing litter depth and canopy cover, both consequences of fire suppression, reduced establishment of Apache and Chihuahua pine but not silverleaf oak such that, with increasing time since fire, oak establishment outstrips that of pine establishment.

Similarly, Barton (1999) showed the displacement of pines by oaks following moderate intensity fires, occurring between 1982 and 1989, in the Chiricahua and Animas Mountains of Arizona and New Mexico. He also noted that most oaks established between 1860 and 1920, following the cessation of widespread fire in the southwest, and hence were re-sprouts from the last fire event and not long-term top-survivors of frequent fire (Barton 1999). Barton (1995, 1999) identified differing fire survival strategies for pines (fire resistance through thick bark) and oaks (fire endurance via re-sprouting) and suggested that this differences may result in the dominance of oaks following infrequent moderate to high severity burns that have become common for these pine-oak woodlands in the United States.

Fulé and others (2000) report similar results for an intense wildfire in a Madrean oak-pine forest in northern Mexico following a fire free period of 62 years. Oaks which comprised over 90% of the preburn density and basal area, suffered nearly 92% mortality or top killing compared to only 45% for pines. However, oaks and manzanita were strong resprouters such that an increase in the relative proportion of resprouting species' (compared to pines) was expected. Herbaceous production increased significantly after 1 growing season and fine fuels (grass, forbs) were sufficient to carry subsequent fire (>1600 kg/ha) suggesting that a frequent, low intensity fire regime could be re-established at the site.

Given the structural and compositional changes that have occurred in these Madrean pine-oak woodlands since the cessation and/or re-introduction of fire, it is hypothesized that fires occurring within this changed vegetation in the future will be of moderate to high intensity and result in even further compositional and structural changes (Barton 1999; Barton and others 2001; Fulé and Covington 1996, 1998). Specifically, Madrean pine-oak woodlands will shift away from pines which cannot tolerate the higher fire intensities and take longer to regenerate from seed, towards oak species which sprout prolifically from underground buds immediately following fire, creating oak shrublands with dense fuel loads (Barton 1995, 1999; Barton and others 2001; Barton 2002; Fulé and Covington 1996, 1998).

Photographic monitoring of Madrean pine oak woodland plot, dominated by Arizona white oak, in Saguaro National Park, Arizona, shows a variety of post fire vegetation responses to a fall fire (Figures 5-3 to 5-5). Responses ranged from increasing cover of re-sprouting shrubs (Figure 5-4) to decreasing overstory species and opening up the woodland (Figures 5-3 and 5-5).

Figure 5-3. Repeat photographs of one Madrean pine oak woodland plot taken in Saguaro National Park's Rincon Mountains following a 1998 fall fire (top) and 5 years

later (bottom) in 2003. Note the patchy boundary of the fire as well as the herbaceous canopy recovery.

Figure 5-4. Repeat photographs of one Madrean pine oak woodland plot taken in Saguaro National Park's Rincon Mountains following a 1998 fall fire (top) and 5 years later (bottom) in 2003. Note the shift from mixed story woodland to a predominantly scrubby canopy.

Figure 5-5. Repeat photographs of one Madrean pine oak woodland plot taken in Saguaro National Park's Rincon Mountains following a 1998 fall fire (top) and 5 years later (bottom) in 2003. Note the survival of one oak tree and the resulting open woodland aspect.

Understory **–** We found no studies, in addition to those cited in the **Overstory** and **Density** sections, that documented changes within the Madrean pine oak understory.

Herbaceous Layer - We found no studies that documented changes within the Madrean pine oak herbaceous layer.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure - We found no studies, in addition to those cited in the **Overstory** and **Density** sections, that documented canopy cover changes.

Structure Class (Size Class) - In addition to the studies cited in the **Overstory** and **Age Structure** sections, Fulé and Covington (1997) document changes in the overstory tree diameter distribution at a fire-excluded (FE) site relative to a frequent-fire (FF) site. The FE site showed a distinct reverse-J shaped distribution curve compared to a more normally distributed diameter distribution for the FF site; pines, oaks and other species generally shared the same distribution pattern at each site.

Barton (1999) documents changes in population size structure for Chihuahua and Apache pine and several species of oaks before and after management burns or wildfires at five sites in the Chiricahua Mountains, Arizona. Fire modified size structure and had differential effects among species. For all species the relative number of smaller to larger stems decreased from pre- to post-fire. This effect was more dramatic in two oak species, silverleaf and Arizona white oak than in the two pine species. In less than 10 years, the oak species "were well on their way to restoring their populations of small stems" and the ratio of small to large oaks had increased. In contrast, this ratio in pine populations had decreased.

Fulé and Covington (1994) report lower stem radial growth rates of pines at the FE site compared to the FF site. Average stem radial growth from pith to 1992, both for all pines and for Durango pine, the sites' dominant pine, was 0.3 cm/yr at the FE site, significantly lower than the 0.5 cm/yr estimated for the FF site. However, there was no difference in average growth rate found for pines over 100 years old. Since sites were closely matched in slope, aspect, elevation, slope position, soil parent material, and plant composition, the authors attributed the growth rate differences to their differing fire regimes; fire has been shown to positively influence growth through thinning of trees and release of nutrients (Covington and Sackett 1992).

Life Form - We found no studies, in addition to those cited in the **Overstory** section, that documented changes in life form.

Density – In addition to information cited in the Overstory section and in section 5.3, Fulé and Covington's (1998) comparison of pine oak woodlands under differing fire regimes details out differences in tree density for 2 reduced frequency fire regimes, Tables 5-2 and 5-3 summarize their results.

Table 5-2. Tree densities from stem mapped plots at a fire excluded (from 1945 to 1994) site near Durango Mexico, data taken from Fulé and Covington's (1998) evaluation of spatial structure in Madrean pine-oak forests with varying fire regimes. See Fulé and Covington (1996, 1997) for other summaries of these data.

Size Class	Species Group	Density (trees/ha)	S.E.M	Basal Area (m^2 / ha)	S.E.M

Live > 1.3 m	Pine	2,780	339.5	29.7	5.4
Live > 1.3 m	Oak	1,136	239.7	9.4	1.0
Live > 1.3 m	Alder	572	124.7	1.4	0.2
Live > 1.3 m	Juniper, Madrone	260	62.1	1.1	0.2
Live > 1.3 m	**Total**	**4,748**	**483.1**	**41.6**	**4.6**
Live < 1.3 m	Pine	4,420	752.8	Not Reported	Not Reported
Live < 1.3 m	Oak	8,916	1,436.6	Not Reported	Not Reported
Live < 1.3 m	Alder	1,624	421.0	Not Reported	Not Reported
Live < 1.3 m	Juniper, Madrone	792	348.3	Not Reported	Not Reported
Live < 1.3 m	**Total**	**15,572**	**1,861.7**	Not Reported	Not Reported
Dead > 1.3 m	Pine	328	126.9	8.9	3.1
Dead > 1.3 m	Oak	32	11.3	1.3	0.9
Dead > 1.3 m	Alder	92	7.7	0.2	0.1
Dead > 1.3 m	Juniper, Madrone	12	39.4	0.1	0.1
Dead > 1.3 m	**Total**	**464**	**129.8**	**10.5**	**3.9**

Table 5-3. Tree densities from stem mapped plots at a fire returned (fire excluded for 29 years and then returned) site near Durango Mexico, data taken from Fulé and Covington’s (1998) evaluation of spatial structure in Madrean pine-oak forests with varying fire regimes. See Fulé and Covington (1996, 1997) for other summaries of these data.

Size Class	**Species Group**	**Density (trees/ha)**	**S.E.M**	**Basal Area (m^2 / ha)**	**S.E.M**
Live > 1.3 m	Pine	296	200.0	29.9	9.7
Live > 1.3 m	Oak	216	56.0	5.3	0.1
Live > 1.3 m	Alder	160	0	1.2	1.2
Live > 1.3 m	Juniper, Madrone	136	24.0	1.9	0
Live > 1.3 m	**Total**	**808**	**280.0**	**38.2**	**11.0**
Live < 1.3 m	Pine	1,096	840.0	96	32.0

Live < 1.3 m	Oak	12,248	2,184.0	3,560	792.0
Live < 1.3 m	Alder	8,760	1,080.0	2,512	1,088.0
Live < 1.3 m	Juniper, Madrone	2,448	1,488.0	448	144.0
Live < 1.3 m	**Total**	**24,552**	**2,432.0**	**6616**	**1704.0**
Dead > 1.3 m	Pine	432	256.0	13.4	6.7
Dead > 1.3 m	Oak	160	112.0	3.2	3.1
Dead > 1.3 m	Alder	344	136.0	0.3	0.1
Dead > 1.3 m	Juniper, Madrone	192	192.0	0.2	0.2
Dead > 1.3 m	**Total**	**1,128**	**696.0**	**17.1**	**9.9**

Additionally, a study by Barton (2002) of pine oak woodland response to the Rattlesnake (1994) and Methodist (1983) fires shows that the density of pines drops while oak density remains the same. Specifically, pine and oak abundance were both greater than 1000 and 500 trees/ha for the Rattlesnake and Methodist fire plots before the fires. Post-fire, pine abundance dropped below 250 trees/ha for both locations while oak abundance remained the same as pre-burn levels (Barton 2002).

Age Structure - In addition to information cited in the **Overstory** section and section 5.3, Fulé and Covington's (1994, 1997, 1998) comparison of pine oak woodlands under differing fire regimes shows differences in age and size structure as a function of fire regime. Sites with frequent fire (FF) had the majority (between roughly 75 and 225 living trees/ha) of trees in the 20 to 50 cm mean quadratic diameter range, whereas areas with Fire excluded (FE) or Fire returned (FR) regimes had the majority (between roughly 2,600 and 4,800 trees/ha and 250 and 600 trees/ha respectively) of trees in the 10 cm mean quadratic diameter class (Fulé and Covington 1997, 1998). Conversely, FF sites had only roughly between 0 and 10 trees/ha in the 10 cm diameter class (Fulé and Covington 1997, 1998). This study suggests that larger, hence older, trees are dominant under historic fire regimes, and that increases in the number of smaller diameter trees are the result of low fire frequency.

Age structure information is consistent with this interpretation (Fulé and Covington 1994, 1997, 1998). The majority of trees at the FE site established after the last widespread fire in 1945 whereas most trees at the FF site date from before this; this is also true of the FR site where the majority of trees established prior to the last widespread fires there in 1955 and 1983. The age distributions were more heterogeneous at the FF and FR (burned sites) because the proportion of younger to older trees was higher at the FE site. The density of trees older than 50 years (the approximate period of fire exclusion at FE) was similar at the 3 sites (ca. 97 trees/ha) but by 1993, tree densities increased substantially at the FE site, altering the forest age structure toward greater dominance by younger trees. The ratio of trees < 50 years old to those > 50 years was 2.7 at the FE site, in contrast to 0.2 and 0.6 at the FF and FR sites, respectively.

Barton (1999) presents current age structure information for silverleaf oak, Apache pine, and Chihuahua pine in lower Rhyolite Canyon. Most individuals established between 1860 and 1920 at the interface between a period of high fire frequency and the onset of fire exclusion in the late 1800's. The range of establishment years in silverleaf oak was more restricted than in the pines which established in low number in all decades following 1920.

Patch Dispersion – In addition to information cited in the **Overstory** section and section 5.3, Fulé and Covington's (1998) comparison of pine oak woodlands under differing fire regimes shows differences in patch dispersion of overstory species. Sites with FF showed a uniformly to randomly distributed live tree (> 1.3 m) overstory at scales of between 2 and 12 meters, whereas FE and FR sites showed a predominantly clumped with some random pattern distribution at scales between 2 and 12 meters (Fulé and Covington 1998). These results suggest that the distribution of overstory species is dependent of frequent fire and that, at least initially, the return of fire does not restore the overstory spatial distribution of trees.

5.6 Madrean Pine Oak Woodland References

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Chapter 6 - Mixed Conifer Forest

6.1 General Description

Mixed conifer (MC) forest occurs in very small patches in the higher elevation areas of Arizona and New Mexico, comprising about 3% and 4% respectively, of total land cover for the two states (Moir and Ludwig 1979). This forest type occurs across a broad range of elevations, spanning 7,100 ft to 11,900 ft above sea level, depending upon latitude, aspect, and slope. The dominant tree species for mixed conifer forests is the interior or blue variety of Douglas-fir (*Pseudotsuga menziesii* var. *glauca),* which ranges from the Rocky Mountains in Canada along a 2700 mile belt into the mountains of central Mexico (Hermann and Lavender 1990), although in some areas the dominant or codominant tree species may be Engelmann spruce *(Picea engelmanii),* white fir (*Abies concolor),* big tooth maple (*Acer grandidentatum),* southwestern white pine *(Pinus strobiformis),* limber pine (*Pinus flexilis)* ponderosa pine (*Pinus ponderosa*), and rarely, blue spruce (*Picea pungens)*. Although not a conifer, quaking aspen (*Populus tremuloides)* is often an important seral (successional) component of this forest type. At upper elevations, mixed conifer forests intergrade with spruce-fir forests, and at lower elevations, mixed conifer forests gradually cede dominance to ponderosa pine and several oak species. Climatological data indicate that MC forest occurs where mean annual precipitation exceeds 30 in, and mean snowfall depth exceeds 100 in (2540 mm) (Pearson 1931).

Moir and Ludwig (1979) proposed a classification system for MC forests throughout Arizona and New Mexico that differentiates three series into 11 habitat types based upon the presence of tree species and understory vegetation composition. Other authors since then have proposed classification systems for portions of the two-state region (e.g. Muldavin and others 1996, USFS 1997).

The first of the three series is *Picea pungens* Series, in which there are five Habitat Types (HT):

1. The *Picea pungens-Picea engelmanii/Senecio cardamine* HT is found primarily in the Hannagan Creek and Thomas Creek drainages of the White Mountains of the Apache-Sitgreaves National Forest, and has a diverse and well developed understory of bittercress ragwort (*S. cardamine)*, Canadian violet (*Viola Canadensis)*, sneezeweed (*Dugaldia hoopsii),* Richardson's geranium *(Geranium richardsonii),* wild strawberry (*Fragaria virginiana),* fringed brome (*Bromus ciliatus), and* sedges *(Carex spp).*
2. The *Picea pungens-Picea engelmanii/Erigeron superbus* HT occurs around Big Lake again in the Apache-Sitgreaves NF, and has a well developed understory comprised of splendid daisy (*E. superbus),* dry sedge (*Carex foena), F. virginiana,* Arizona peavine (*Lathyrus lanswertii* var. *arizonica),* Arizona fescue (*Festuca arizonica),*screwleaf muhly (*Muhlenbergia virescens),*Kentucky bluegrass *(Poa pratensis),* and *B. ciliatus.*
3. The *Picea pungens/Poa pratensis* Habitat Type occurs in the Sangre de Cristo, San Juan, Sacramento, Mogollon, and San Mateo mountains, has willow (*Salix)* and alder (*Alnus spp).*, serviceberry (*Amelanchier alnifolia),* chokecherry *(Prunus virginiana),* and Rocky Mountain maple (*Acer glabrum*) in the shrubby midstory,

and has an extremely rich and diverse herbaceous understory represented by *P. pratensis, Fragaria spp., E. superbus, G. richardsonii,* horsetail (*Equisetum spp).,* northern bog violet (*Viola nephrophylla), Schizanche pupurescens,* and cow parsnip (*Heracleum lanatum).* The *Picea pungens/Carex foena* Habitat Type occurs in the White Mountains, North Kaibab Plateau, and in the Mogollon Mountains of New Mexico.

4. The fourth type is the *Picea pungens/Carex foena* Habitat Type, the understory of which is dominated by *C foena* and the grasses *Festuca arizonica, Muhlenbergia montana,* and *Bromus ciliatus.* Important forbs for this Habitat Type include *F. virginiana, Antennaria spp., Achillea lanulosa, Lathyrus arizonica,* and *Erigeron spp.*, and it is found in the White Mountains and North Kaibab Plateau of Arizona, and in the Mogollon Mountains in New Mexico.
5. The last Habitat Type for this series is the *Picea pungens/Pseudotsuga menziesii* HT. For this type, both trees are codominant, occurs on sideslopes rather than alluvial terraces or valley bottoms. Four different phases are recognized for this HT, determined by the shrub and understory composition, which varies from *Arctostaphylos uva-ursi,* to *Linnaea borealis*, *Quercus gambelii, Amelanchier alnifolia, Salix scouleriana, Jamesia Americana, Pachistima myrsinetes, Berberis repens, Juniperus communis, Rosa woodsii, Symphoricarpus oreophilus,* and *Rubus parviflorus.* Understory vegetation varies widely as well, and may include *Valeriana acutiloba, Oryzopsis asperifolia, Geranium spp., Lithospermum multiflorum, Achillea lanulosa, Pedicularis canadensis, Fragaria virginiana, F. vesca, Bromus ciliatus, Poa fendleriana, Aquilegia spp.,* and *Cystopteris fragilis.* These phases occur in Sangre de Cristo and San Juan, Sacramento, Mogollon and White mountains.

The second series delineated by Moir and Ludwig (1979) is the *Abies concolor* Series, which also has five different Habitat Types (HT):

1. The *Abies concolor-Pseudotsuga menziesii/Acer glabrum HT* with a Oregon grape (*Berberis repens)* understory occurs in the mountains of northern New Mexico, while the same HT with a *Holodiscus dumosus* understory occurs in the Sacramento, Mogollon, Chiricahua, and Pinaleno mountains.
2. The *Abies concolor-Pseudotsuga menziesii/Quercus gambelii HT* has both *Pinus ponderosa and P. strobiformis* as seral trees, while *Q. gambelii* and *Robinia neomexicana* dominate the shrub layer and *Acer* is absent. Dominant graminoids include *Bromus ciliatus, Poa fendleriana, Carex rossii,* and *Muhlenbergia virescens*, as well as *Stipa pringlei, Elymus elymoides,and minor amounts of Festuca arizonica, P. fendleriana, P. interior,* and *Koeleria cristata*. Important forbs include *Pteridium aquilinium, Thermopsis pinetorum, G. caespitosum, Erigeron platyphyllus, Artemisia ludoviciana,* and *Vicia pulchella.* This HT is common in AZ and NM, occurring on Bill Williams Mountain, the Sierra Anchas (Pase and Johnson 1968), the Mogollon Rim, White Mountains, Chiricahuas, Mogollon Mountains, Sacramento Mountains (Hanks and Dick-Peddie 1974), Capitan, San Juan, and Sangre de Cristo mountains.
3. The *Abies concolor-Psedotsuga menziesii* HT with sparse understory occurs in two phases, either having *Berberis repens* or *Juniperus communis* or *Pachistima myrsinites* as the evergreen shrub layer, or with deciduous shrubs such as *R. neomexicana, S. oreophilus, S. scouleriana, and Q. gambelii.* This HT is widespread, with the *B. repens* phase occurring on the North Kaibab Plateau, White Mountains, San Juan and Sangre de Cristo mountains, while the *R.*

neomexicana phase is in the Sacramento, Pinaleno, Chiricahua, Mogollon, and White Mountains.

4. The *Abies concolor/Acer grandidentatum* HT has a minor component of *P. menziesii*, and occurs both on top of, and along drainages of the Mogollon Plateau, and in the Pinaleno and Santa Catalina Mountains (Whittaker and Niering 1965).
5. The final HT in this series is the *Abies concolor/Festuca arizonica* HT, which includes *Pseudotsuga menziesii* and *Pinus ponderosa*, but for which shrubs are a minor component. There are conspicuous patch openings that are inhabited by a rich understory composed of the grasses *F. arizonica, M. montana, M. virescens, P. fendleriana, K. cristata, E. elymoides,* and *S. pringlei.* There are also forbs associated with the grasses, including *L. multiflorum, Antennaria spp., L. arizonicus, Thalictrum fendleri, A. lanulosa,* and *Erigeron spp.* This HT occurs on the San Francisco Peaks, Mogollon Plateau, White Mountains, and San Juan Mountains.

The last series is the *Pseudotsuga menziesii* Series, and it is represented by the *Pseudotsuga menziesii/Pinus strobiformis/Muhlenbergia virescens HT.* This HT contains *P. ponderosa*, shrubs are minor, and the understory is dominated by *M. virescens.* This HT is found in the Chiricahua, Mogollon, Pinaleno, and Santa Catalina Mountains (Whittaker and Niering 1965), and is the hottest and driest of MC Forests (See Figure 6-1).

Figure 6-1 Early photograph of mixed conifer forest (Douglas-fir and possibly southwestern white pine) in the Sierra Ancha Range of Tonto National Forest, 1917. Note mixed age of stand, with large trees in foreground (with fire scars) and younger trees in background. Photograph courtesy of the USFS Regional Office collection #164752.

6.2 Historical Range of Variation of Ecological Processes

Vegetation Dynamics - Outside of fire histories and climate studies using tree rings, little research has been focused on MC forest stand dynamics. This is despite the fact that MC forests have been used heavily for grazing, timber harvest, recreation, and hunting since around 1700. One study in the Sacramento Mountains (Hanks 1966) articulated the approximate seral stages of succession following fire. Hanks (1966) determined that stand replacing fires occurred in 1886, 1939, 1945, 1950, and 1963. Following the fire, stands were dominated by herbaceous species for 1 to 3 years, followed by increased growth of *Q. gambeli* and *R. neomexicana* until these attained tree size, at which point conifers gradually began to dominate. Another study by Hanks and Dick-Peddie (1974) found that after a stand-replacing fire, a forb (herbaceous) stage lasted for 1 to 2 years before resprouting oaks assumed dominance. This oak stage dominated until conifers began to colonize and overtop the oaks. Conifers that could colonize within the oak thickets included *P. ponderosa* and *P. strobiformis,* as well as *A. concolor* and *P. menziesii.* Low-intensity surface fires killed seedling conifers and even sapling and juvenile *Abies* and *Pseudotsuga*, creating open, park-like savannas with scattered groves of oak (Cooper 1961, Weaver 1968, Hanks and Dick-Peddie 1974). Hanks (1966) suggested that the oakbrush stage was never a climax in this sere, and would eventually be replaced by conifers, albeit slowly. In many other MC forests, *Populus tremuloides* is the primary initial colonizer, although several coniferous species quickly establish themselves as well (Moir and Ludwig 1979).

Disturbance Processes and Regimes-

Climate- Please see Climate Analysis in Introductory chapter.

Fire- In a comparison of fire regimes of ponderosa pine and mixed conifer forests in the Jemez Mountains, Touchan and others (1996) found that MC had less frequent surface fires, but also experienced patchy crown fires that were not in evidence for ponderosa pine. They also found that precipitation was reduced in the winter to spring period immediately prior to the fire occurrence. They reported a pre-1900 Weibull Median Probability Index (WMPI) for major fires (fires that scar more than 10% of trees in a study area) as 9.7 to 14 years, with a maximum fire interval of 18 to 32 years, and a minimum of 4 to 6 years. Similarly, Swetnam and Baisan (1996a) equated severe droughts with large fire years (total area burned/yr), and wetter periods with smaller fire years in a dendrochronological study comparing ponderosa and mixed conifer tree rings that date back to 1700. They also found a *general* pattern of longer intervals between low intensity surface fires, but higher variability around means that indicates that elevation and forest type were poor determinants of fire frequency. They postulated that fire frequency was more likely determined by site characteristics and land use history. They also found that, in contrast to the lag time for ponderosa pine, there was no lag between wet and dry years and large-scale fires in MC forests.

In another study, Swetnam and Baisan (1996b) determined the seasonality of fires in MC forests of the Madrean Province occurring prior to 1900. More than 40% of fires occurred between early May and early June, 30% occurred in June, 20% occurred before early May, and approximately 9% occurred late June to mid-July, and about 1% occurred between July and September. This timing corresponds to the arid 'foresummer' and lightning-caused fire season. However, they point out, there were slight differences in seasonal timing during specific years, over different time periods, and in different sites. Swetnam and Baisan (1996b) also indicate that while there probably are ecological implications of fire interval distributions and phenological effects of fire seasons, no studies have definitively linked ecological patterns and processes for southwestern systems over periods of centuries.

Grissino-Meyer and others (1995) studied fire scars of mixed conifer forest trees from two sites in the Pinaleño Mountains, and found a WMPI of 4 to 6 years for low intensity surface fires prior to 1880. They also determined from the age structure of the residual spruce-fir forest (based on tree ring data reconstructions) that it established in 1685 after one of the most widespread and intense stand replacing fires. They also determined that pre-1880 fires were initiated in the early part of the season (May to June).

In a study of charcoal from bogs going back 9,000 years, and from dendrochronologically dated fire scar collections from over 600 trees at 42 localities with over 4,000 pre-1900 fire scar dates extending back to 1422, Allen and others (2002 abstract only) differentiated between the scale and intensity of high- and low-elevation MC forest fire regimes. Their data suggest that prior to 1900, extensive (>100 ha) crown fires did occur in higher elevation mixed conifer and spruce-fir forests, but lower elevation mixed conifer forests burned primarily as surface fires, and stand-replacing events probably occurred at smaller scale (<100 ha).

Hydrology - We found no studies that documented hydrological processes such as flooding as important historical ecological determinants for the mixed conifer forest.

Herbivory - We found no studies that documented herbivory as an important historical ecological determinant for the mixed conifer forest.

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and introductions as important historical ecological determinants for the mixed conifer forest.

Insects and Pathogens – Swetnam and Lynch (1989) found that there have been 8 or 9 outbreaks of western spruce budworm (*Choristoneura occidentalis*) since 1700, with average return intervals of 30 to 40 years. Western spruce budworm populations periodically increase to outbreak proportions, and cause extensive defoliation, tree mortality and altered succession in several mixed conifer species. Lynch and Swetnam (1992) studied several old growth mixed conifer sites in New Mexico and found evidence of multiple outbreaks of western spruce budworm, but found that outbreaks were not focused on old growth stands. Several other species of insects as well as fungi currently use mixed conifer tree species (more information forthcoming in Insect Analysis).

Nutrient Cycling - We found no studies that documented nutrient cycling as an important historical ecological determinant for the mixed conifer forest, although several authors have conducted soil nutrient cycling research in mixed conifer forests (Covington and Sackett 1986, White 1994, 1996). Mixed conifer forests typically have slower rates of mineralization, although rates are variable, possibly due to overstory composition, season of year, or time since last fire (Covington and Sackett 1986, White 1996).

Windthrow - We found no studies that documented windthrow as an important historical ecological determinant for the mixed conifer forest.

Avalanche - We found no studies that documented avalanche as an important historical ecological determinant for the mixed conifer forest.

Erosion - We found no studies that documented erosion as an important historical ecological determinant for the mixed conifer forest.

Synthesis - Little is known about pre-settlement processes in mixed conifer forests, except regarding fire, drought, insects, and their interaction. In pre-settlement times, the fire regime of mixed conifer forests was a mixture of infrequent, small patch size, high intensity crown fires interspersed with more frequent, widespread and low intensity surface fires (Touchan and others 1996). There is no published information differentiating disturbance regimes between or among different types of vegetation or moisture regimes of mixed conifer forests, although two theses are in preparation to document these differences (Allen *pers. comm*).

6.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation - We found no studies that documented historical patch composition of mixed conifer forests.

Overstory - Fule and others (2003) reconstructed forest structure from 1880 for mixed conifer forests at Grand Canyon National Park's north rim, and Cocke and others (2005) reconstructed forest structure from 1876 for mixed conifer forests on the San Francisco Peaks. Table 6-1 displays reported values for the following mixed conifer forest structure data by trees per acre, basal area, and percentage of basal area by tree species or group of species:

GCNP	**ABCO**	**ABLA**	**PIEN**	**PIPO**	**POTR**	**PSME**	**RONE**	**Total**
Trees/ac	24.0	1.14	4.4	26.3	24.0	18.5	N/A	98.3
BA(ft^2/ac)	23.9	0.4	2.6	23.5	3.0	23.5	0	76.7
% BA	31.3	0.6	3.4	30.7	3.9	30.7	0	100.0
SFPA	**ABIES**	**PIAR**	**PIEN**	**PIPO**	**POTR**	**PSME**	**PIFL**	**Total**
Trees/ac	0.97	0.9	0.2	7.3	17.5	20.5	17.7	65.1
BA(ft^2/ac)	1.0	1.1	0.0	8.9	2.8	43.7	20.2	77.9
% BA	1.3	1.5	0.0	11.5	3.6	56.1	25.9	100.0

Table 6-1. Historic forest structure reconstructed for two sites (GCNP=Grand Canyon National Park in 1880, SFPA=San Francisco Peaks in 1876) in Arizona. Basal area (BA) is expressed both in square ft. per acre (ft^2/ac) and as a percent of total. Species or groups across column labels are as follows: ABCO=white fir (Abies concolor), ABLA=corkbark fir (Abies bifolia formerly A. lasiocarpa), PIEN=Engelmann spruce (Picea engelmanii)+blue spruce (Picea pungens), PIPO=ponderosa pine (Pinus ponderosa), POTR=aspen (Populus tremuloides), PSME=Douglas-fir (Pseudotsuga menziesii), RONE=New Mexican locust (Robinia neomexicana), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (Pinus aristata), PIFL=limber pine (Pinus flexilis).

Understory - We found no studies that documented the historical understory composition of mixed conifer forests.

Herbaceous Layer - We found no studies that documented the historical herbaceous layer composition of mixed conifer forests.

Patch or Stand Structure of Vegetation – We found no studies that documented the historical stand structure of mixed conifer forests.

Canopy Cover Class (%) or Canopy Closure - We found no studies that documented the historical canopy closure of mixed conifer forests.

Structure Class (Size Class) - Historically, there was a larger proportion of older, larger trees and a smaller proportion of younger smaller trees compared to contemporary forests (Covington and Moore 1994, Dieterich 1983, Fule and others 1997).

Life Form - We found no studies that documented the historical life form composition of mixed conifer forests.

Density - We found no studies that documented historical density of mixed conifer forests. Several historic photographs suggest (e.g., see Figure 6-2), and some authors have postulated that historic forests were lower density than contemporary forests, due to fire suppression and in-filling by white fir and Douglas-fir (Swetnam and Baisan 1996a, Muldavin and Tonne 2003).

Age Structure - We found no studies that documented the historical age structure of mixed conifer forests.

Patch Dispersion - We found no studies that documented the historical patch dispersion of mixed conifer forests, although Touchan and others (1996) allude to patchiness as a result of a mixed fire regime (see *Synthesis*, below).

Recruitment Dynamics - We found no studies that documented the historical recruitment dynamics of mixed conifer forests.

Reference Sites Used – None at this time, although two studies are forthcoming from the Valles Caldera and Jemez Mountains that may identify these areas as useful reference sites for mixed conifer forests.

Synthesis – Very little is known about the historical condition of mixed conifer forests, except that in general, forests had a more open structure, with a larger proportion of older, larger trees, and a smaller proportion of younger, smaller trees. Historically these forests were less dense, although there were small patches of trees in several age classes, and in areas that experienced frequent fire, there were fewer fire sensitive species such as white fir, and a mixture of age classes. Areas that experienced less frequent and more severe fires probably had even aged stands of trees, although these patches were smaller than those areas that experienced more frequent fire. At the landscape scale, these forests were probably very patchy or heterogeneous, with dispersion of high and low frequency fire patches controlled by some combination of topography, soils, and vegetation (Touchan and others 1996, Muldavin and Tonne 2003).

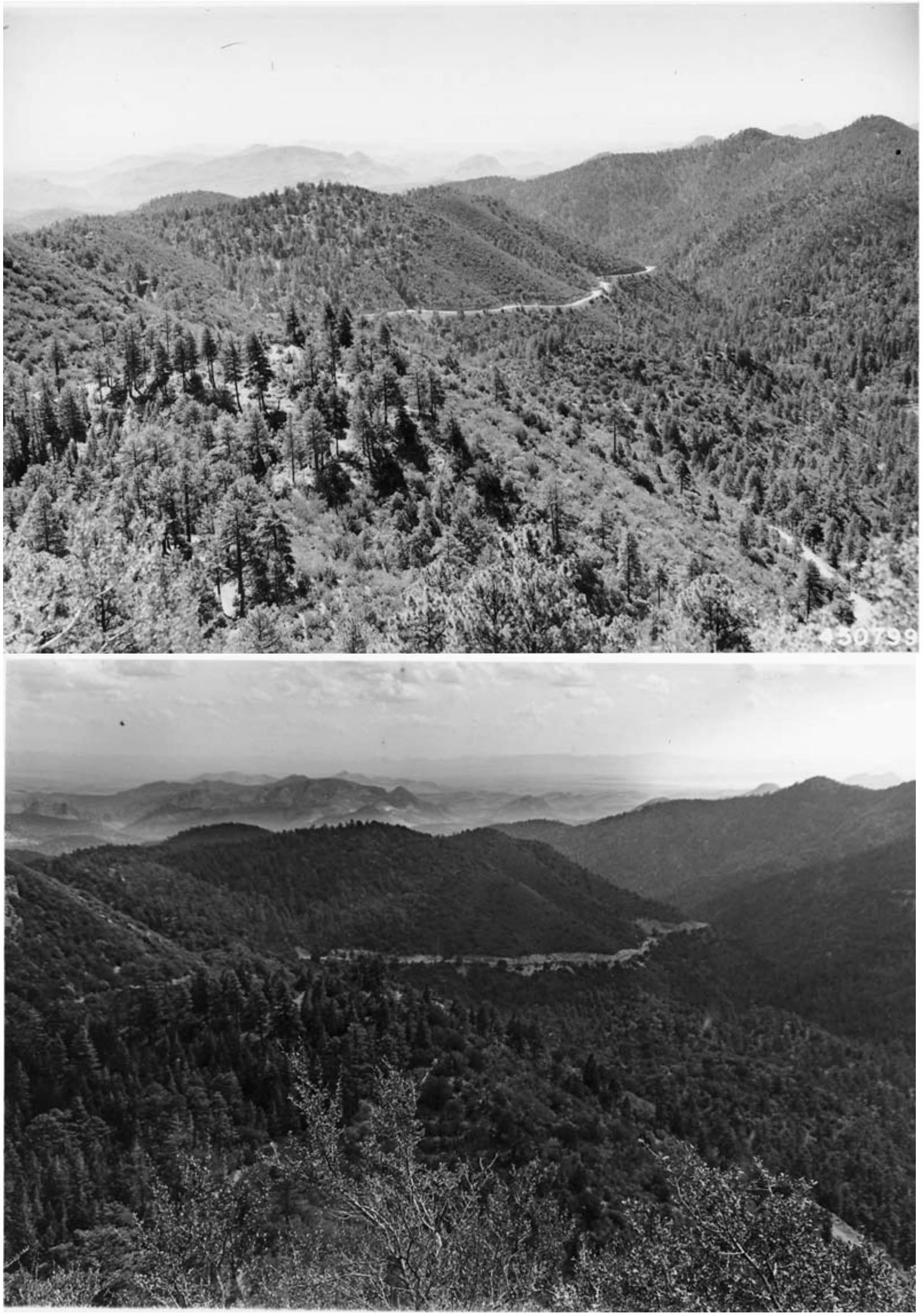

Figure 6-2. Photographic comparison from the Gila National Forest, top photo from 1948, and bottom photo from 1984. These images depict an increase in tree density, especially in the understory over 36 years. Photos courtesy USDA Forest Service Region 3 Office.[Top image15a, #450799. Looking east from top of divide on Black Range Road. By E.L. Perry. 7/29/48 (Gila) and bottom image15b, NM Museum of Natural History Field Number 183. Retaken by B. Sallach. 10/1/84 (Gila).]

6.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory - It has been suggested that the extinction of large carnivores such as grey wolf and grizzly bear has affected at least one component of the mixed conifer forest, quaking aspen, which experiences increased mortality due to high levels of herbivory by native and introduced (elk) ungulates. Predator control has allowed herbivores to increase in numbers, and to exert longer duration and higher intensity grazing and browsing effects on select vegetation types, especially aspen (Shepperd and Fairweather 1994, Romme and others 1995, Kay 1997, Ripple and others 2001).

Silviculture – Selective logging practices emphasized the removal of the largest, most merchantable tree species (Douglas-fir, ponderosa pine) from mixed conifer forests, leaving many of the stands dominated by smaller, more fire susceptible, and less desirable (for timber) white fir and limber pine trees (Muldavin and Tonne 2003).

Fragmentation – In some areas, the construction of a high density road network in support of logging operations has led to fragmentation of mixed conifer forests, altering the local hydrology, and increasing soil erosion (Muldavin and Tonne 2003).

Mining – We found no studies that documented mining as an important ecological determinant for the mixed conifer forest.

Fire Management – The disruption of historic fire regimes by introduced grazing animals has been well documented in southwestern ecosystems, and high elevation mixed conifer forests were well utilized as summer range for large numbers of sheep and cattle (Carlson 1969, Allen 1989, Covington and Moore 1994, Swetnam and others 1999). In the early 1900s, active fire suppression through the construction of fire lines and roads, and later, more organized efforts with fire brigades and air tankers, began to function as the primary mechanisms for excluding fire from Southwestern forests. Baisan and Swetnam (1997) analyzed fire scars along a “proximity-to-humans” gradient near Albuquerque in ponderosa pine and mixed conifer forests over three centuries of human expansion running from pre-Pueblo Revolt period (1680) into the contemporary forest management and fire suppression period (up to 1992). They found large reductions in fire frequency for both vegetation types, corresponding to a doubling of the Mean Fire Interval (MFI) over the first two centuries, followed by the ‘extirpation’ of fire over the third century in mixed conifer systems, with the greatest perturbation occurring in forests closer to human settlements (Baisan and Swetnam 1997). However, Grissino-Meyer and Swetnam (2000) indicated that large scale climatic patterns also affect fire regimes, showing that after an extended dry period with frequent fires (1400 to 1790), annual precipitation has increased, fire frequency has diminished, and the season of fire has shifted from mid-summer to late spring.

Exotic Introductions (Plant and Animal) – We found no studies that documented exotic introductions as important ecological determinants for the mixed conifer forest, with the exception of Rocky Mountain elk (see herbivory, above).

Synthesis – Mixed conifer forests have been affected primarily by grazing, silviculture and fire management practices that have favored conditions more conducive to infrequent, stand replacing fires, and the decline of aspen. Some areas also had a high density road network established for the removal of timber.

6.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory - In some mixed conifer forests on the North Kaibab Plateau and other areas, lack of fire has led to dominance of former ponderosa pine ecosystems by less fire-tolerant mixed conifer species (e.g. *P. menziesii*, *A. concolor*). This has led to the mistaken classification of former ponderosa pine forests as mixed conifer forests, when in fact they are ponderosa pine forests that have become dominated by mixed conifer species (Fule *pers. comm.*, Allen and others 2002). Fire suppression has probably led to an increase in density of young white fir and Douglas-fir trees (Muldavin and Tonne 2003). Fule and others (2003) described current forest structure for mixed conifer forests at Grand Canyon National Park's north rim, and Cocke and others (2005) described current forest structure for mixed conifer forests on the San Francisco Peaks. Table 6-2 displays reported values for the following mixed conifer forest structure data by trees per acre, basal area, and percentage of basal area by tree species or group of species:

GCNP	**ABCO**	**ABLA**	**PIEN**	**PIPO**	**POTR**	**PSME**	**RONE**	**Total**
Trees/ac	87.5	64.7	77.2	18.9	68.9	30.9	5.5	353.6
Regeneration	782.6	472.1	109.3	0.0	1549.8	109.3	222.7	3245.7
BA(ft^2/ac)	62.3	9.1	23.5	20.0	13.9	40.1	0.1	169.0
% BA	36.9	5.4	13.9	11.9	8.3	23.7	0.1	100
SFPA	**ABIES**	**PIAR**	**PIEN**	**PIPO**	**POTR**	**PSME**	**PIFL**	**Total**
Trees/ac	3.7	3.9	0.9	11.4	71.5	130.7	110.3	332.4
Regeneration	0.0	3.0	0.0	11.2	579.4	73.1	88.1	754.7
BA(ft^2/ac)	4.8	1.8	0.7	8.2	37.9	75.6	68.3	197.1
% BA	2.4	0.9	0.3	4.2	19.2	38.4	34.6	100

Table 6-2. Current forest structure determined for two sites (GCNP=Grand Canyon National Park, SFPA=San Francisco Peaks) in Arizona. Basal area (BA) is expressed both in square ft. per acre (ft^2/ac) and as a percent of total. Species or groups across column labels are as follows: ABCO=white fir (Abies concolor), ABLA=corkbark fir (Abies bifolia formerly A. lasiocarpa), PIEN=Engelmann spruce (Picea engelmanii)+blue spruce (Picea pungens), PIPO=ponderosa pine (Pinus ponderosa), POTR=aspen (Populus tremuloides), PSME=Douglas-fir (Pseudotsuga menziesii), RONE=New Mexican locust (Robinia neomexicana), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (Pinus aristata), PIFL=limber pine (Pinus flexilis). Trees are defined as stems having dbh > 1 inch, and regeneration as stems having dbh <= 1 inch.

Fule and others (2003) censused mixed conifer forests at Grand Canyon National Park's north rim, and determined a current-era basal area of 170 ft^2/acre (standard error=12), and a tree density of 354 trees/acre (s.e.=36) for trees >1 inch (dbh), and a tree density of 3246 trees/acre for trees < 1 inch dbh. Cocke and others (2005) censused mixed conifer forests on the San Francisco Peaks, and determined current-era basal area of 197 ft^2/acre (s.e.=18.3), and a tree density of 332 trees/acre (s.e.=31) for trees >1 inch (dbh), and a tree density of 755 trees/acre (s.e.=202) for trees < 1 inch dbh. (Table 2-3).

Understory- We found no studies that documented the effects of human disturbance on the understory composition of mixed conifer forests.

Herbaceous Layer – We found no studies that documented the effects of human disturbance on the herbaceous layer composition of mixed conifer forests.

Patch or Stand Structure of Vegetation - Many mixed conifer forests have experienced logging activity for economical purposes, although the extent of this disturbance and its impacts on stand dynamics or other ecological effects has not been well documented (Bahre 1991). Muldavin and Tonne (2003) noted that fire suppression has led to greater homogeneity of forest stands, and a decrease in homogeneity of structure among stands through time. Fule and others (2003) reported a 260% increase in total trees per acre over historic conditions on the north Rim of the Kaibab National Forest, and Cocke and others reported a 410% increase in total trees per acre over historic conditions for the San Francisco Peaks.

Canopy Cover Class (%) or Canopy Closure - We found no studies that documented the effects of human disturbance on the canopy cover of mixed conifer forests.

Structure Class (Size Class) - We found no studies that documented the effects of human disturbance on the structure class of mixed conifer forests.

Life Form – We found no studies that documented the effects of human disturbance on the life form of mixed conifer forests.

Density - We found no studies that documented the effects of human disturbance on the density of mixed conifer forests, although see discussion in *Overstory*, above.

Age Structure - We found no studies that documented the effects of human disturbance on the age structure of mixed conifer forests.

Patch Dispersion - We found no studies that quantified the effects of human disturbance on the patch dispersion of mixed conifer forests, although one paper speculated that fire suppression has led to an increase in homogeneity of forest patch structure (Muldavin and Tonne 2003). They further postulated that the concomitant increase in large crown fires has led to a positive feedback loop of an increase in uniformity of stand structure, and a decrease in the small patch mosaic that once dominated mixed conifer forests.

Recruitment Dynamics - We found no studies that documented the effects of human disturbance on the recruitment dynamics of mixed conifer forests.

Synthesis - Unfortunately, little has been quantified on the effects of anthropogenic disturbance on mixed conifer forests. However, a combination of anthropogenic factors, namely grazing management, fire management, and silvicultural practices have had several profound effects on general trends in mixed conifer forest form and function. The historical mixed severity fire regime of small patches of infrequent, high severity fire within a broader matrix of low severity, frequent fires has been replaced with a more uniform and large scale, low frequency and high severity fire regime. This change in disturbance, combined with selective logging techniques that preferentially removed the larger, older Douglas-fir trees has resulted in more homogenous mixed conifer forests, with greater numbers of smaller, fire susceptible trees.

6.6 Mixed Conifer References

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Chapter 7 - Ponderosa Pine Forest & Woodland

7.1 General Description

Southwestern ponderosa pine (*Pinus ponderosa* Dougl. Ex Laws. *Var. scopulorum*) occurs in large contiguous patches throughout Arizona and New Mexico, at elevations ranging from 5500 feet to 8500 feet. These relatively warm and dry forests are dominated by ponderosa pine, pinyon pine (*Pinus edulis*, *P. discolor*), junipers, (*Juniperus spp.*), and several oaks (*Quercus spp.*). Ponderosa pine forest occupies 9,247,700 acres (or 6.12%) of the total combined land area of Arizona and New Mexico, of which 5,835,400 acres (or 63.10%) are under US Forest Service management on 11 National Forests, which comprises 28.64% of Region 3 National Forest land area (USGS 2004). The number of acres of ponderosa pine forest on each National Forest is found in Table 7-1:

Apache-Sitgreaves	**Carson**	**Cibola**	**Coconino**	**Coronado**	**Gila**
926,400	408,000	508,900	814, 600	65,400	1,754,600
Kaibab	**Lincoln**	**Prescott**	**Santa Fe**	**Tonto**	**Total**
555,100	68,500	98,400	505,400	130,100	5,835,400

Table 7-1. Approximate area (in acres) of potential natural vegetation types (PNVTs) across 11 Region 3 National Forests in Arizona and New Mexico. Region 3 National Grasslands in New Mexico, Oklahoma and Texas were not included in this analysis. Data used to generate this table included The Southwest Regional Gap Analysis Program (SWReGAP) and the landownership GIS-based layer. Note that SWReGAP data have not been tested for accuracy and are derived from remote sensing; therefore, analyses at the individual National Forest scale may be inaccurate.

Ponderosa pine forest is typically bounded at the upper elevation by mixed conifer forest, and at the lower elevation by grassland, pinyon-juniper forest, or chaparral, although extensive intergrading of species may occur at ecotone boundaries along gradients of slope, elevation, aspect, and moisture (Moir 1993). Climatological data indicate that ponderosa pine forests occupy a wide moisture and temperature gradient, with annual precipitation ranging from 20 to 35 inches, and mean annual air temperatures ranging from 41 to 52 °F, which allow for a growing season of approximately 180 days (Moir 1993). Figure 7-1 is a 1906 photograph of ponderosa pine forest in the Coconino National Forest, and Figure 7-2 is a 1903 photograph from the Kaibab National Forest showing a typical ponderosa pine-bunchgrass community.

Figure 7-1. 1906 Photograph of ponderosa pine forest in the Coconino National Forest. Note mixed sizes/ages - large trees in the foreground, and smaller trees including regeneration in the background. Photograph from Northern Arizona University's Ecological Restoration Institute.

Figure 7-2. 1903 photograph of ponderosa pine bunchgrass forest (or savannah) near Tusayan, AZ on the Kaibab National Forest. Note the wide expanse of grass between clumps of large trees, but also a mixture of age and size classes is present. Photograph from Northern Arizona University's Ecological Restoration Institute.

The Southwest Region of the United States Forest Service (1997) delineates 19 plant associations or habitat types for the ponderosa pine series. Below we provide the relevant, verbatim excerpted names of those habitat types, their “key criteria” for identification, and their location within Region 3. Other details of plant association composition, structure, TES climate class, fire ecology, reforestation and revegetation considerations, adjacent plant associations, and references can be found in the original text.

1. Ponderosa pine/rockland. KEY CRITERIA: Exposed rock outcroppings or very shallow (<4" deep) soil comprises 50 to 90% of the surface. Understory species are scarce and trees include ponderosa pine, pinyon and/or junipers. LOCATION: Scattered throughout the southwest where large rock outcroppings occur in the ponderosa pine region. Elevation: 7,500' - 8,500'.
2. Ponderosa pine/Indian ricegrass (*Oryzopsis hymenoides*). KEY CRITERIA: Must have hoary rosemarymint (*Poliomintha incana*), which is a shrub. An open forest that occurs on sandy soils. LOCATION: Very local in northern New Mexico (on stabilized sand dunes near Espanola) and southern Colorado (on the eastern edge of the Great Sand Dunes). Occurs on deep, sandy soils. Elevation: 5,900' to 6,300'.
3. Ponderosa pine/blue grama (Bouteloua gracilis). KEY CRITERIA: Blue grama is well represented, or if poorly represented, alligator juniper or pinyon are common. Arizona fescue is scarce or absent. Ponderosa pine overstory is often open; shrubs are poorly represented although big sagebrush may be well represented on some sites. LOCATION: Widespread in New Mexico, Arizona, Colorado and Utah. Lower elevations (6,250' to 8,550') of the ponderosa pine series.
4. Ponderosa pine/Arizona walnut. KEY CRITERIA: Arizona walnut or canyon grape are common, but riparian trees like Arizona alder (*Alnus oblongifolia*), boxelder (*Acer negundo*), sycamore (*Platanus wrightii*), or species of cottonwood (*Populus* spp.) are limited to microsites. Twoneedle pinyon (*Pinus edulis*), Arizona pinyon (*Pinus fallax*) are absent or accidental. LOCATION: Occasionally found on alluvial terraces of intermittent washes or stream sides south of the Mogollon Rim and in southwestern New Mexico. Elevation: 5,500 to 6,400'.
5. Ponderosa pine/Gambel oak. KEY CRITERIA: Must have at least 5% canopy cover of Gambel oak. Overstory regeneration is ponderosa pine. Douglas-fir (*Pseudotsuga menziesii*) is absent or accidental. Widespread and common throughout New Mexico, Arizona, Colorado, and Utah. Elevations range from 6,000’ to 9,200’ on a wide variety of slopes, landforms, and soils. Mean annual precipitation is 20-22” per year.
6. Ponderosa pine/pointleaf manazanita (*Arctostaphylos pungens*). KEY CRITERIA: Pointleaf manzanita, a large shrub, is abundant (> 25% canopy cover). This community type is interpreted (Muldavin and others 1996) as a fire-derived expression of various ponderosa pine/evergreen oak habitat types. Douglas-fir and silverleaf oak are absent or accidental. LOCATION: Central Arizona, south of the Mogollon Rim, particularly on the Tonto National Forest and San Carlos Reservation, and north of the rim in the Pinedale/Showlow area and vicinity. On steep upper slopes, ridgetops, or elevated plains. Elevation: 5,600' to 6,800'.

7. Ponderosa pine/Silverleaf oak. KEY CRITERIA: Silverleaf oak (*Quercus hypoleucoides*) is well represented (>5% canopy cover). Ponderosa pine is the dominant overstory species, and Douglas-fir and white fir (*Abies concolor*) are absent or accidental. LOCATION: A major plant association in southeastem Arizona, including the following geographic areas- Chiricahua, Pinaleno, Santa Rita, Santa Catalina, Huachuca and the Galiuro Mountains with outliers to San Carlos and Ft. Apache Reservation, and in New Mexico, to the Brushy Mountains on the Glenwood Ranger District (Gila NF). Elevations generally 5,700' - 8,000', but can be outside this range on special topographic sites.
8. Ponderosa pine/Emory oak (*Quercus emoryi*). KEY CRITERIA: Emory oak is well represented (>5% canopy cover). Located primarily in drainages and lower slopes. Gambel oak and silverleaf oak are absent or poorly represented. Rocky Mountain juniper (*J. scopulorum*) is absent or accidental. LOCATION: This plant association occurs south of the Mogollon Rim, in southwestern New Mexico, and southern Arizona. It is more likely found in the Central Highlands, along the base of the Mogollon Rim and the Nantanes Plateau, and it is uncommon in the basin ranges and plateau regions of southern Arizona. Found most commonly on mid to lower slopes and ravine bottoms, PIPO/QUEM is most differentiated along drainages with granitic soils (Udic Ustochrepts). Elevation: 5300' -6900'. Mean annual precipitation is 20-22" per year.
9. Ponderosa pine/netleaf oak (*Quercus rugosa*). KEY CRITERIA: Netleaf oak is well represented, or if poorly represented, oaks are well represented and netleaf oak is the dominant species of oak. This association deviates from the typical silverleaf oak series rule as Douglas-fir and southwestern white pine (*Pinus strobiformis*) can be minor in the stand at climax. Ponderosa pine is still the dominant overstory species. White fir is absent or accidental. LOCATION: Mostly found in southeastern Arizona and southwestern New Mexico (Animas Mountains, with outliers in the Mogollon Mountains near Glenwood). This is the highest elevational type of the ponderosa pine plant associations with an evergreen oak understory. Generally [found] on steep, upper slopes or ridge-tops with shallow rocky soils with rocky outcrops. Elevation: 5,200' - 8,800'.
10. Ponderosa pine/mountain muhly (*Muhlenbergia montana*). KEY CRITERIA: Although mountain muhly is often present to well represented, it does not have to be present to be called this plant association. Gambel oak, if present, is <5% canopy cover. Douglas-fir and quaking aspen (*Populus tremuloides*) are absent or accidental and then usually in microsites, or if present, may represent an ecotone between PIPO/MUMO and an adjacent plant association. LOCATION: Southwest and central New Mexico up through southern Colorado, southern Arizona to southern Utah. Elevation: 7100' to 9400' on south facing slopes. Elevated and valley plains, piedmont hillslopes, mountain slopes, mesas and benches. Soils are varied. Mean annual precipitation is 20-25" per year.
11. Ponderosa pine/Arizona white oak (*Quercus arizonica*). KEY CRITERIA: Arizona white oak is well represented (>5% canopy cover). This is one of the warmest, driest ponderosa pine environments. Gambel oak, silverleaf oak, and Emory oak are poorly represented or absent. Douglas-fir, white fir, southwestern white pine and Rocky Mountain juniper are absent or accidental. LOCATION: South of the Mogollon Rim, in southwestern New Mexico and southern Arizona. More likely found in the Central Highlands, along the base of the Mogollon Rim and the Nantanes Plateau. Uncommon in the basin ranges and plateau regions of

southern Arizona. On a wide range of soils and parent materials. Elevation: 5,380' to 7,750'.

12. Ponderosa pine/Gray oak (*Quercus grisea*). KEY CRITERIA: Oaks must be well represented (>5% canopy cover). Must have at least 1% canopy cover of gray oak, but gray oak must be the dominant oak. Gambel oak, if present, is clearly minor in abundance to gray oak. Southwestern white pine and Douglas-fir are absent or accidental. These grasses are absent or accidental: Arizona fescue, pine muhly, bullgrass, and pinyon ricegrass. LOCATION: In central New Mexico and and east-central Arizona: on Apache-Sitgreaves National Forests. (Clifton and Alpine ranger districts)-Big Blue, Blue Mtns., AZ; on the Gila National Forest (Luna, Reserve, Mimbres, and Quemado ranger districts)-Blue Mtns., NM, Saliz Mtns., San Francisco Mtns., Mogollon Mtns., Black Range, NM, Tularosa Mtns., NM; on the Cibola National Forest (Magdalena and Mountainair ranger districts) - San Mateo, Gallinas Mtns., NM, also in Organ Mtns., NM. Frequently on slopes and ridgetops, often on shallow soils and rocky outcrops. Also found on deep soils of alluvial terraces and valley plains. Elevation: 6,100-8,800', at upper elevations on south or west slopes). Mean annual precipitation is 19-21" per year.
13. Ponderosa pine/wavyleaf oak (*Quercus x pauciloba*). KEY CRITERIA: This ponderosa pine dominated plant association must have at least 5% canopy cover of oak, with at least 1% cover of wavyleaf oak. Gray oak is scarce or absent, and Gambel oak, if present, is less dominant than wavyleaf oak. LOCATION: Widespread in southern (Sacramento Mtns., Lincoln NF and Mescalero Apache Reservation) and central to northeastern New Mexico. Found locally in other reaches of northern New Mexico. Elevation: 6,500'-8,200' on hot, dry sites. Surface rock cover can be high [averaging 27% in one study (DeVelice, 1986)]. Mean annual precipitation is 20-21" per year, with hot and dry weather in May and June.
14. Ponderosa pine/screwleaf muhly (*Muhlenbergia virescens*). KEY CRITERIA: A mesic ponderosa pine site, must have screwleaf muhly. This is the wettest type in the ponderosa pine series in northern Arizona. Douglas-fir, white fir, and pinyon are absent or accidental. LOCATION: Southwest and central New Mexico, southern Arizona to central Arizona (up to San Francisco Peaks area). Elevation: 6,700' to 8,800' to 9,400' (2,879 m) on south-facing slopes. Mean annual precipitation is 23"-25" per year. Found on many slopes and aspects.
15. Ponderosa pine/screwleaf muhly-Arizona fescue (*Festuca arizonica*). KEY CRITERIA: A mesic ponderosa pine site, must have screwleaf muhly and Arizona fescue. Douglas-fir, southwestern white pine, and Utah juniper (*Juniperus osteosperma*) are absent or accidental. LOCATION: Central Arizona (generally north of the Mogollon Rim up to the San Francisco Peaks area), and southwestern New Mexico (Gila NF). Elevation: 6,900'-9,200'. Found on many slopes and aspects.
16. Ponderosa pine/Arizona fescue. KEY CRITERIA: Must have Arizona fescue, or if grazing history includes persistent use which can explain the absence of Arizona fescue, then Kentucky bluegrass (*Poa pratensis*) is present. Douglas-fir, white fir, pinyon, and juniper are generally absent or accidental. The DAPA2 phase may contain Douglas-fir, usually in microsites, and the BOGR2 phase may contain some pinyon and junipers. LOCATION: Widespread in New Mexico, central Arizona, and southern Colorado, infrequent south of the Mogollon Rim.

Elevated and valley plains, piedmont hillslopes and mountain slopes, Elevation: 6,800' - 8,800' to 9,400' on south-facing slopes. Mean annual precipitation is 20-25" per year.

17. Ponderosa pine/kinnikinnik (*Arctostaphylos uva-ursi*). KEY CRITERIA: Kinnikinnic (*Arctostaphylos uva-ursi*), a low-growing shrub, ranges from 25-70% canopy cover and generally is the sole dominant understory species. Douglas-fir, twoneedle pinyon, and white fir are absent or accidental. LOCATION: Northern New Mexico (Jemez, Sangre de Cristos, and San Juan mountains) and southern Colorado. Elevation: 7,700' to 9,200'. Shallow soils of ridgetops, stony or excessively well drained soils on other slopes.
18. Ponderosa pine/Stansbury cliffrose (*Purshia stansburiana*). KEY CRITERIA: Must have Stansbury cliffrose in the understory. White fir is absent or accidental, but occasional Douglas-fir may be present in the late successional overstory. LOCATION: Central and northern Arizona, local in central and northern New Mexico (Zuni Mtns., Jicarilla Apache Reservation); also in Utah, Colorado, Wyoming, and Idaho. Rough, rocky topography at warmer limits of ponderosa pine forests. Elevations: 6,700' to 7,400'. Usually on soils with sandstone-limestone parent materials. Mean annual precipitation is19-20" per year.
19. Ponderosa pine/black sagebrush (*Artemisia nova*). KEY CRITERIA: Must have black sagebrush, which is a short sagebrush shrub found from northwestern New Mexico and northern Arizona (Kaibab Plateau and Grand Canyon) north to Oregon and Idaho. The open stand structure of ponderosa pine combines with a denser structure in woodland species, and greater than 5% canopy cover of shrubs. LOCATION: Local in northern New Mexico, northern Arizona, and southern Colorado. Widespread in southern Utah. This plant association has a minor distribution in New Mexico and Arizona. Elevation: around 8,200'. Occurs primarily on flat, basaltic mesa tops and elevated plains. In southern Utah, often occurs on flat benches. Rooting depths are often shallow due to restrictive subsurface horizons. Youngblood and Mauk (1985) state that in southern Utah, "sites may potentially have seasonal high water tables and even ponding."

Moir and others (1997) have combined these 19 plant associations of the Southwest into three major groups, based on similarities in structure, composition, and fire response.

> "The **fringe pine forest types** are at dry, warm, lower elevations where ponderosa pine occurs with woody species that are common in the adjoining pinyon/juniper and pinyon/oak/juniper woodlands. Depending on geographic location, typical associated species are *P. edulis*, *P. discolor*, *P. californiarum*, *Juniperus* spp., *Quercus grisea*, *Q. arizonica*, *Q. emoryi*, *Arctostaphylos pungens*, *Artemisia tridentata*, and *Chrysothamnus nauseosus*. Associated trees form a mid-level canopy layer below the ponderosa pine overstory (Marshall 1957). These additional species provide resources for a wide variety of animals. Blue grama (*Bouteloua gracilis*) is a diagnostic species, and ponderosa pine/blue grama has widespread forest association throughout the Southwest (USFS 1986).
>
> Where precipitation is greater than about 480 mm [20 inches], blue grama is absent or minor and ponderosa pine occurs with **understory bunchgrass** species, mainly *Festuca arizonica*, *Muhlenbergia montana*, and/or *M. virescens*. There may be a mid-level canopy of shrubs, copses of oaks, or even an occasional oak

> tree (Kruse 1992), but these are minor vegetation components. Fires, either lightning- or human-caused, are frequent in these dry forests. Southwestern pine forests can be grouped with ponderosa pine forests in other areas in the Western United States that share a similar fire ecology. Southwestern ponderosa pine/bunchgrass forests are similar to warm, dry forests in Idaho, Montana, and Utah (Davis et al. 1980; Crane and Fischer 1986; Fischer and Bradley 1987; Bradley et al. 1992). Numerous descriptions of presettlement forests in the Southwest (Woolsey 1911; reviews Cooper 1960; Covington and Moore 1994; Moir and Dieterich 1988) apply to this group of forests.
>
> The third group has **understories dominated by shrubs and midlevel trees**. Bunchgrasses may still be abundant, especially as patches in open areas. Common woody associates include *Quercus gambelii*, *Q. undulata*, *Robinia neomexicana*, *Cercocarpus montana*, and *Symphoricarpos oreophilus*. These forests are similar in structure and fire responses to the warm, moist ponderosa forests of central Idaho and Utah (Crane and Fischer 1986; Bradley and others 1992)."

Another grouping of the nineteen habitat types could be delineated as follows, based on the similarities of surface fire regimes (USFS 1997):

1) Bunchgrass types, including 3, 10, 14, 15, 16, and possibly 5;

2) Shrub understory types, including 6, 7, 8, 9, 11, 12, 13, and possibly 5;

3) Fringe types, which includes 1, 2, 4, 17, 18, and 19. These include mesic areas or rocky areas where fire is either less frequent or less important in maintaining the vegetation type.

7.2 Historical Range of Variation of Ecological Processes

Vegetation Dynamics – Ponderosa pine regeneration by seed has been suggested to be episodic, occurring only with the unique combination of heavy seed production, very moist springs and summers, and a period of several years without fire (Pearson 1950, Savage and others 1996). The results of a series of successful, historical regeneration events have been quantified from the Gus Pearson Natural Area prior to 1876 to be approximately 0.4 to 1.57 trees per acre per decade for the past 300 years (Mast and others 1999). Figure 7-3 shows a 1933 photograph of ponderosa pine regeneration and the notes by early forester Gus A. Pearson on the photograph, indicating that there are three age classes, and the approximate years of regeneration events: pre-1890, 1890, 1910, 1914, 1919, and 1929.

F288262 Fort Valley Exp. Forest, Ariz. Oct. 1933 G. A.
Complete restocking on 40-year-old cutover area of ponderosa
pine. There are 3 age classes of seedlings, besides fair
thickets of saplings and poles. The main age-classes represented are: 1929, 1919, 1914, 1910, about 1890, and older.

Figure 7-3. 1933 photograph and annotation by Gus A. Pearson of regeneration in ponderosa pine from the Ft Valley Experimental Forest on the Coconino National Forest. Photograph from Northern Arizona University's Ecological Restoration Institute.

Six vegetative structural stages (VSS) have been identified for ponderosa pine (Moir and Dieterich 1988), VSS1-VSS6, corresponding to different ages, densities, and sizes of trees within a forest. Ponderosa pine forest dynamics are affected by fire, insects, diseases, and windfall (Moir and others 1997). Canopy cover classes for each stage are delimited as A (0 to 40%), B (40 to 60%), and C (>60%). The VSSs and their descriptions are as follows:

1. VSS1 -- Openings occur following major disturbance such as fire or gap processes. These openings may be maintained as open parks or meadows in pine savannahs where frequent surface fires occur and may also include a snag stage following a stand replacing fire.
2. VSS2 – This forest is composed of seedlings and saplings with dbh<5 in growing in an herbaceous or shrubby environment. As seedlings grow into saplings the canopy begins to close.
3. VSS3 – Young stands are composed of small trees (5 in<dbh<11.9 in), and usually clumped and dense, with canopy cover often exceeding 70% without recurring fire. Stands have a sparse herbaceous understory, few snags, and single-storied structure.
4. VSS4 -- Mid-aged stands have larger trees (14 in<dbh<17.9 in) that are sexually mature, tend to be multi-storied, and contain some snags.
5. VSS5 – Mature stands have large trees (dbh>17.9 in) in single or multi-storied stands, with more litter and dead and downed debris in fire-suppressed stands.

These stands may contain larger snags than mid-aged stands, and produce a good seed crop.

6. VSS6 – Old-growth forests are single and multi-storied, have many mature trees and dense canopies (>40% canopy cover) in stands that have not experienced ground fires in their VSS1 and VSS2 stages. Prior to 1890, these forests were extensive and more open (0 to 40% canopy cover), had very little coarse woody debris on the forest floor, but required at least 300 years beyond the VSS1 or burned snag stage to attain old-growth characteristics.

Ponderosa pine stand dynamics have been estimated for a generalized southwestern ponderosa pine forest in the "Management Recommendations for the Northern Goshawk in the Southwestern United States" (Reynolds and others 1992). The authors utilized Vegetative Structural Stage (VSS) classes in this document, and based their findings on published data for the time required for trees to become established following disturbance and on data for tree diameter growth rates for several species, and they defined the pathological age of trees as the age beyond which growth slows, decay develops significantly, and mortality becomes high (Reynolds and others 1992). The authors acknowledged that a successful seedling establishment period in time depends upon a combination factors determined by ponderosa pine characteristics, individual tree genetics, and climate:

- ◊ Cone crop frequencies
- ◊ Successful cone development
- ◊ Seed production and development
- ◊ Proper germination conditions in the soil
- ◊ Root system establishment
- ◊ Climatic conditions.

They also identified factors that influence tree diameter growth rates:

- ◊ Initial diameter
- ◊ Site productivity
- ◊ Climatic conditions
- ◊ Level of management

Reynolds and others (1992) provided estimates of diameter growth rates, age in VSS, accumulated age, and proportion of landscape in each VSS for a typical ponderosa pine forest with Basal Area (BA) of 60 ft^2/ac and a site index of 70 (trees at this site attain height of 70 feet in 100 years), shown here in Table 7-2:

Vegetative Structural Stage and dbh (inches)	VSS1 0-1	VSS2 1-5	VSS3 5-12	VSS4 12-18	VSS5 18-24	VSS6 24+
MINIMAL MANAGEMENT						
Diameter growth/decade (inches)	0	1.33	1.52	1.48	1.3	1
Years (acc-yrs)	20 (20)	30 (50)	46 (96)	41 (137)	46 (183)	50 (233)

Landscape in SS (%)	9	13	20	17	20	21
MODERATE MANAGEMENT						
Diameter growth/decade (in)	0	1.91	1.76	1.64	1.4	1.10
Years (acc-yrs)	20 (20)	21 (41)	40 (81)	37 (117)	43 (160)	45 (204)
Landscape in SS (%)	10	10	19	17	20	24
INTENSIVE MANAGEMENT						
Diameter growth/decade (inches)	0	2.5	2	1.8	1.5	1.2
Years (acc-yrs)*	20 (20)	16 (36)	35 (71)	33 (104)	40 (144)	50 (194)
Landscape in SS (%)	10	8	18	17	21	26

Table 7-2. Estimates of diameter growth rates, age in Vegetative Structural Stage (VSS), accumulated age, and proportion of landscape in each VSS for a typical ponderosa pine forest with Basal Area (BA) of 60 ft^2/ac and a site index of 70. *Years (acc-yrs) = Number of years in that VSS (and total accumulated years in tree age). Data are from Reynolds and other (1992).

This information is generalized for all types of ponderosa pine forests, and no other information on stand dynamics could be located. With the variety of plant associations or habitat types that exist for ponderosa pine in the Southwest, one could expect variations in response to disturbance by different habitat types as well as variations in the frequency of disturbance.

Disturbance Processes and Regimes

Climate- See Introduction Chapter on Climate. Swetnam and Baisan (1996) determined that fuel production and fuel moisture, both dependent upon climate, were important factors in ponderosa pine fire regimes. Fire years are tightly correlated with drought, especially when preceded by one to three years of high precipitation which builds fine fuels in grasses and herbaceous cover. Years with few fires are correlated with high precipitation. Swetnam and Betancourt (1990, 1998) described climate-fire linkages.

Fire- Ponderosa pine forests throughout the Southwest formerly experienced widespread, low-intensity surface fires of frequent return intervals (Weaver 1951, 1952, Cooper 1960, Dieterich 1980, Covington and Moore 1994, Swetnam and Baisan 1996). Analysis of a comprehensive network of fire scar sites and their fire chronologies indicates that for 53 sites in Arizona and New Mexico where ponderosa pine dominates or co-dominates, mean fire return intervals were 2 to 17 years for fires scarring one or more trees, and 4 to 36 years for fires scarring between 10% and 25% of trees between the years of 1700 and 1900 (Swetnam and Baisan 1996). For the same network of sites, Swetnam and Baisan (1996) reported a range of Weibull Median Probability Interval (WMPI) values of 1.74 to 13.83 years. For a smaller subset of 31 pure ponderosa pine sites, the FRI ranged from

5.4 to 36.3 years for fires scarring more than 25% of trees, with an average of 15.6 years. With such a wide range of fire return intervals, it would be instructive to parse the data from the 53 sites into the three broad categories of xerophytic ponderosa pine forest delineated by Moir and others (1997): **fringe ponderosa pine forest**, **ponderosa pine-bunchgrass**, and **ponderosa pine-shrub**. Due to the presence of multiple moisture, temperature, and soil gradients, it is unclear at this time how to group similar forest communities within similar disturbance groups, although Touchan and others (1994) identified the major factors controlling differences in fire intervals among sites as topography, grazing history, and climatic variability, and not habitat type.

Other fire regime studies include Sneed and others (2002) finding of a WMPI of 1.79 to 3.93 years, with a minimum fire interval (FI) of 1 year and a maximum of 24 years for the period of 1615 to 1996 for ponderosa pine forest in the Prescott National Forest in Arizona. They also found through analysis of early, middle, and late- early-wood that 8.1% of fires occurred in May or early June, 24.4% of fires occurred in mid-June, and peaked at 63.9% in late June and July with the onset of "monsoon" rains and lightning storms, and 3.8% of fires occurred in August and early September as fuels became wetter. They reported no evidence of fires occurring from late September to early May (Sneed and others 2002). Fule and others (2003) also analyzed rings for seasonality at Grand Canyon National Park, and noted that they were able to discern differences on 54% of samples. They found that 12% of fires occurred during the dormant season, spring fires accounted for 54% of fires, and 46% of fires occurred during the summer. They also noted that because narrow rings were the most common reason for not being able to detect fire season, their seasonal distribution may be biased toward wetter years, which might not reflect the entire range of variation for ponderosa pine forests (Fule and others 2003).

Fule and others (2003) analyzed fire scars from ponderosa pine forests on the north and south rims of Grand Canyon National Park, and compared "mainland" sites further from the rim of the canyon with "island" sites that were closer to the rim on plateaus or points. They found that prior to 1880, fires burned most frequently on lower elevation "islands" at a WMPI of 3.0 to 3.9 for all fires, 6.3 to 8.6 for large fires scarring 25% or more of the sampled trees. Fires on the higher elevation "mainland" site on the interior of the North Rim were less frequent, with a WMPI of 5.1 years for all fires and 8.7 years for large fires. The "mainland" site on the South Rim had the least frequent fire, with WMPI of 6.5 years for all fires, and 8.9 years for large fires (Fule and others 2003). In this same study, Fule and others (2003) were able to compare their tree-ring fire scar analysis with known extent of recent historical fires, to calibrate the accuracy of the chronology. This was in response to a criticism raised by Baker and Ehle (2001) that fire scar analysis provided estimates that were sufficiently uncertain as to require bracketing of mean fire interval estimates by as much as 1100 to 1200%. The analysis by Fule and others (2003) showed that for known fire events recorded by direct observation over the period 1924 to 1999, the tree-ring fire scar analysis correctly identified 100% of those fires down to a 20-acre size. Fule and others (2003) did offer that rather than bracket their fire return intervals, they needed to be explicit with the assumptions that the fire-scar analysis techniques probably missed some additional, smaller fires, and that all fires studied, including historic as well as current, "burn with a mosaic of intensities and include unburned areas within the overall fire perimeter." Figure 7-4 is a 1909 photograph from near Grand Canyon indicating patches of different ages and sizes, with a grassy understory.

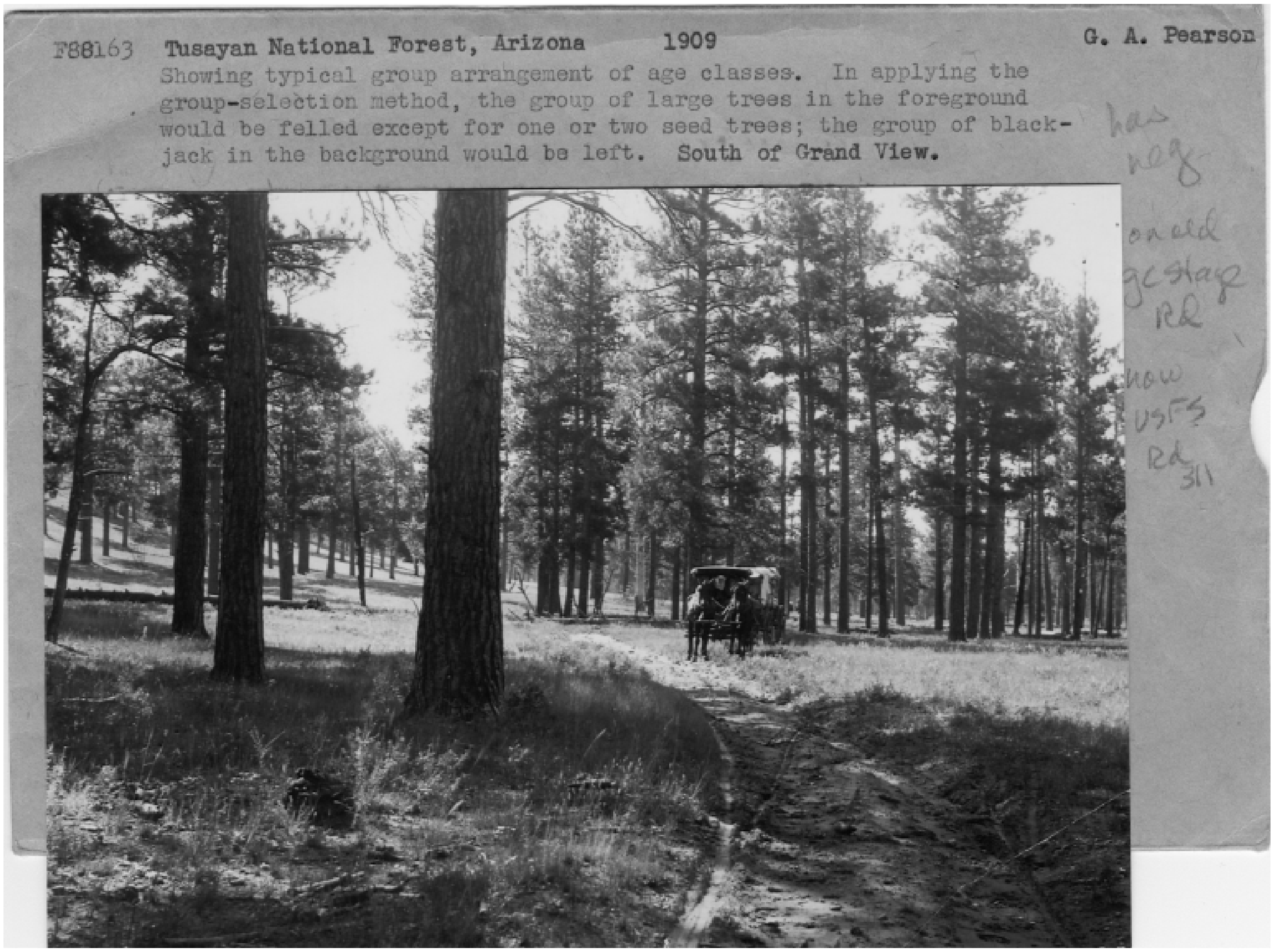


Figure 7-4. Photograph from 1909 from the Kaibab (formerly Tusayan) National Forest near Grand Canyon's s South Rim showing interspersion of patches of different ages and presumably, fire intensities. Photograph by Gus Pearson. Photograph from Northern Arizona University's Ecological Restoration Institute.

The size of historic fires has been estimated from several studies (Bahre 1985, Swetnam and Dieterich 1985, Swetnam 1990, Swetnam and Baisan 1996, Fule and others 2003), but has not been definitively determined for all years nor all areas. Bahre (1985) refers to early historic accounts of millions of acres burning during certain years in the mountains of southeastern Arizona, and Fule and others' (2003) data from Grand Canyon National Park indicate that in certain years bracketed by their study (1873, 1879), fires likely would have covered most of the study area, if not more (22 square miles = 14,000 acres). Swetnam and Dieterich (1985) analyzed fire scars and reported an average fire size of approximately 3,000 acres. Swetnam and Baisan (1996) indicate that large fires in ponderosa pine were synchronized in certain years, and may have covered large areas burning for months, but indicated that further spatial analysis of their regional datasets were underway to give more precise estimates of the spatial extent of historic fires.

Hydrology- We found no studies that documented hydrological processes such as flooding as important historical ecological determinants for the ponderosa pine vegetation type.

Herbivory- We found no studies that documented herbivory as an important historical ecological determinant for the ponderosa pine vegetation type.

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and introductions as important historical ecological determinants for the ponderosa pine vegetation type.

Insects and Pathogens – For thousands of years, Southwestern forest trees have been host to several species of insects, pathogenic fungi, and parasitic plants (Dahms and Geils 1997). Unfortunately, the earliest reports of bark beetles on ponderosa pine date from the early 1900s, after settlement (Dahms and Geils 1997), so there are no accounts of historic insect outbreak periodicity.

Nutrient Cycling – We found no studies that documented historic nutrient cycling processes or rates for the ponderosa pine vegetation type, although several studies have shown short-term increases in litter decomposition and nutrient cycling rates as a result of restoration efforts (thinning and burning) compared to un-restored controls (Covington and Sackett 1984, Covington and Sackett 1986, Kaye and Hart 1998, review in Selmants and others 2003, Kaye and others 2005).

Windthrow - We found no studies that documented windthrow as an important historical ecological determinant for the ponderosa pine vegetation type.

Avalanche - We found no studies that documented avalanche as an important historical ecological determinant for the ponderosa pine vegetation type.

Erosion - We found no studies that documented erosion as an important historical ecological determinant for the ponderosa pine vegetation type.

Synthesis –Many of the studies of stand dynamics of ponderosa pine forests have focused on ponderosa pine-bunchgrass communities, with general trends in size and age of stands inferred from existing stands, and remnants of past stands. Ponderosa pine forests in the Southwest generally experienced a high frequency, low intensity surface fire regime, although on a small scale, individual trees occasionally may have torched via fuel ladders carrying surface fire into the crowns over small areas (Swetnam and Baisan 1996, Vankat 2006). Beyond fire studies, little is known about historic disturbance factors that shaped ponderosa pine forests in historic times, because settlement and disturbance disruption occurred simultaneously.

7.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation - We found no studies that documented historical patch composition of ponderosa pine forests.

Overstory – Several studies have documented age and size structure, as well as tree density and basal area for pre-settlement ponderosa pine forests (Covington and Moore 1994, Fule and others 1997, Mast and others 1999, Moore and others 1999, Huffman and others 2001, Fule and others 2002, Sneed and others 2002, Fule and others 2003, Gildar and others 2004, Moore and others 2004, Cocke and others 2005). Fule and others (2003) reconstructed 1880 forest structure for ponderosa pine forests at Grand Canyon National Park's north rim, and Cocke and others (2005) reconstructed 1876 forest structure for ponderosa pine forests on the San Francisco Peaks. Table 5-3 displays reported values for the following ponderosa pine forest structure data by trees per acre, basal area, and percentage of basal area by tree species or group of species:

GCNP	**ABCO**	**ABLA**	**PIEN**	**PIPO**	**POTR**	**PSME**	**RONE**	**Total**
Trees/ac	12.3	0	0	64.4	57.5	2.0	N/A	136.2
BA(ft^2/ac)	8.3	0	0	41.8	6.1	1.7	0	57.9
% BA	14.3	0	0	72.2	10.5	3.0	0	100.0
SFPA	**ABIES**	**PIAR**	**PIEN**	**PIPO**	**POTR**	**PSME**	**PIFL**	**Total**
Trees/ac	0	0	0	23.4	0.9	0.4	0	24.9
BA(ft^2/ac)	0	0	0	31.4	0.3	1.4	0	33.1
% BA	0	0	0	95.1	0.8	4.1	0	100.0

Table 7-3. Historic forest structure reconstructed for two sites (GCNP=Grand Canyon National Park in 1880, SFPA=San Francisco Peaks in 1876) in Arizona. Basal area (BA) is expressed both in square feet per acre (ft^2/ac) and as percent of total. Species or groups across column labels are as follows: ABCO=white fir (*Abies concolor*), ABLA=corkbark fir (*Abies bifolia* formerly *A. lasiocarpa*), PIEN=Engelmann spruce (*Picea engelmanii*)+blue spruce (*Picea pungens*), PIPO=ponderosa pine (*Pinus ponderosa*), POTR=aspen (*Populus tremuloides*), PSME=Douglas-fir (*Pseudotsuga menziesii*), RONE=New Mexican locust (*Robinia neomexicana*), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (*Pinus aristata*), PIFL=limber pine (*Pinus flexilis*). Data are from Fule and others (2003) and Cocke and others (2005).

In a similar study, Fule and others (1997) determined 1883 forest stand density and basal area for 62 plots at Camp Navajo near Flagstaff, AZ. They reported a mean tree density of 59.9 trees/acre (s.d.=45.6), and mean total basal area of 56.2 ft^2/acre (s.d.=6.1) for ponderosa pine forest as pre-settlement conditions.

In another study that compared historic data from early forest inventories, Moore and others (2004) reported values from the "Woolsey plots" that were established in high timber productivity sites across Arizona and New Mexico in ponderosa pine forests over the period 1909 to 1913 to track potential productivity of SW forests through time through periodic resampling. They reported the following values for tree density and basal area.

Tree Species								Total	
		PIPO		PSME		PIST/PIFL		BA	Trees/ac
Plot	Location	BA	TPA	BA	TPA	BA	TPA	BA	TPA
CIBS1A	Cibola NF	19.2	32.8	3.0	2.4	4.8	7.3	27.0	42.5
CIBS2A	Cibola NF	38.8	50.6	0.00	0.00	0.00	0.00	38.8	50.6
GILAS1A	Gila NF	14.8	19.4	6.5	4.1	0.3	0.4	21.6	23.9
JEMS2A	Jemez	38.8	36.0	0.00	0.00	0.00	0.00	38.8	36.0
JEMS3A	Jemez	60.6	88.7	1.3	1.2	0.00	0.00	61.9	89.9

Table 7-4. Basal area (BA in ft^2/ac) and trees per acre (TPA) for trees with dbh>3.6 inches for five sites in New Mexico by trees species within ponderosa pine forest. Tree codes are PIPO=ponderosa pine, PSME=Douglas-fir, PIST/PIFL=southwestern white pine (*Pinus strobiformis)*/limber pine. Data are from Moore and others (2004).

Understory - We found no studies that documented the historical understory composition of ponderosa pine forests.

Herbaceous Layer- We found no studies that documented the historical herbaceous layer composition of ponderosa pine forests. Before European settlement, ponderosa pine forests were generally open stands with well-developed herbaceous understories (Cooper 1960).

Patch or Stand Structure of Vegetation – Cooper (1960) used a contiguous quadrat analysis of two study areas in the White Mountains of eastern Arizona to determine that trees were clumped in distribution, with clumps ranging in size from 0.16 to 0.32 acre (Covington and Moore 1994). White (1985) used the nearest neighbor technique to determine clumpiness in the Gus Pearson Natural Area, and reported that most clumps were composed of 3 to 44 trees, and each group varied in size from 0.05 to 0.70 acre in size. Moore and others (1993) studied the same area and determined clumps to average around 0.16 acre, with a range of 0.08 to 0.64 acre. Biondi and others (1994) used spatial statistics to describe spatial patterning at GPNA, and reported that stem size was spatially auto-correlated over patches with diameter of 98.43 feet, or 0.17 acre.

Canopy Cover Class (%) or Canopy Closure –White (1985) determined a pre-settlement canopy cover value of 22%, while Covington and Sackett (1986) determined a value of 17%. Covington and Moore (1994) cite a description by Pearson (1923) of how, "rarely does ponderosa pine crown cover reach more than 30%, and usually not over 25%." These values were determined for the ponderosa pine-bunchgrass habitat type on basaltic soils (Covington and Moore 1994).

Structure Class (Size Class)- Historically, there was a larger proportion of older, larger trees and a smaller proportion of younger smaller trees compared to contemporary forests (Dieterich 1983, Covington and Moore 1994, Fule and others 1997). Moore and others (2004) compared historical versus current size class structure of ponderosa pine forests from the 15 Woolsey plots in Arizona and New Mexico, and found that forest surveys conducted from 1909 to 1913 had stand density (of trees >= 3.6 inches DBH) of 34.9 ft^2/ac. They also determined that the Quadratic Mean Diameter, a measure of central tendency in tree diameter within a stand weighted by number of trees was 15.2 inches over the same time period, and that on average, each 1.1 hectare (2.8 ac) plot had 61.5 "young" trees, and 13.3 "old" trees [old and young based on morphology of tree bark] (Moore and others 2004).

Life Form - We found no studies that documented the historical life form composition of ponderosa pine forests.

Density – See *Overstory*, above, for reported values of tree density by species expressed in trees/acre (TPA) and basal area (BA – ft^2/acre).

Age Structure – White (1985) determined that pre-settlement stands were of uneven aged or mixed age composition (Covington and Moore 1994). Moore and others (2004) determined that the Quadratic Mean Diameter, a measure of central tendency in tree diameter within a stand weighted by number of trees was 15.2 inches over the 1909 to 1913 time period, and that on average, each 1.1 hectare (2.8 ac) plot had 61.5 "young" trees, and 13.3 "old" trees [old and young based on morphology of tree bark].

Patch Dispersion – We found no studies that documented the historical patch dispersion of ponderosa pine forests, although the Woolsey plots (Woolsey 1911, Moore and others 2004) could be analyzed for spatial distribution of patches.

Recruitment Dynamics - Ponderosa pine regeneration by seed has been suggested to be episodic (Cooper 1960, Covington and Moore 1994, Savage 1996), and has been quantified from the Gus Pearson Natural Area prior to 1876 to be approximately 0.4 to 1.57 trees per acre per decade for the past 300 years (Mast and others 1999).

Reference Sites Used – Reference sites are the Gus Pearson Natural Area and Camp Navajo (Moore and others 1999), the Woolsey Plots (Moore and others 2004), Kaibab National Forest (Fule and others 2002), Grand Canyon National Park (2003), and the San Francisco Peaks (Cocke and others 2005).

Synthesis –Historic forest structure, patch size, basal area and density of trees are well quantified for portions of northern Arizona, but may not reflect the entire distribution of ponderosa pine forests in Arizona and New Mexico, and may not be sufficient to reflect the historic range of variation for the two-state region. Other measures of forest structure such as patchiness, canopy cover, and age or size-class distribution are not as well quantified, and the values reported are not representative of the entire geographic distribution of ponderosa pine in the Southwest.

7.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory – The disruption of historic fire regimes by introduced grazing animals has been well documented in southwestern ecosystems, and ponderosa pine forests were utilized as summer range for large numbers of sheep and cattle (Allen 1989, Covington & Moore 1994, Swetnam et al 1999). Herbivory by cattle also reduces competition by grasses with conifer seedlings, allowing them to expand or encroach into grasslands (Dahm and Geils 1997).

Silviculture – The practice of silviculture has a long history in the Southwest, and ponderosa pine was the most economically important species because of its use in many applications such as fuel wood, mining timbers, home building, and other commercial uses for lumber. Initially, most hand felling was very selective, as only the most desirable, large trees were removed. Around 1880, with the influx of more Anglo-Americans and others to the region, along with railroads, large-scale logging became more feasible (Schubert 1974). A variety of harvest techniques including clearcutting, selective logging, and group selection were employed, which have left harvested areas with a mosaic of even-aged and uneven-aged stands, a deficit in the oldest trees (Dahms and Geils 1997), as well as some bare areas more prone to erosion (Bahre 1998).

Fragmentation – Construction of logging roads has occurred in many ponderosa pine forests of the Southwest, although we could find no studies that documented the impacts of fragmentation on ponderosa pine forests.

Mining – We found no studies that documented mining as an important ecological determinant for the ponderosa pine vegetation type.

Fire Management –

The disruption of historic fire regimes by introduced grazing animals has been well documented in southwestern ecosystems, and ponderosa pine forests were well utilized as summer range for large numbers of sheep and cattle (Allen 1989, Covington & Moore 1994, Swetnam et al 1999). In a study of ponderosa pine fire scars in the Chuska Mountains bordering northern Arizona and New Mexico, Savage and Swetnam (1990) reported a MFI of 2.8 years for the period 1660 to 1830, when early introduction of sheep disrupted this frequent fire regime. Subsequently, from 1830 to 1930, the MFI increased to 6.1 years, and for the period 1930 to 1986, when fire suppression activities were more organized and effective, the MFI jumped to 13 years, meaning fires were almost five times less frequent in modern times than in pre-settlement times. In the early 1900s, active fire suppression through the construction of fire lines and roads, and later concerted efforts with fire brigades and air tankers, began to function as the primary mechanism for excluding fire from Southwestern forests (Covington and Moore 1994, Swetnam and Baisan 1996). Fire exclusion was very successful initially, but subsequent accumulation of fuels, through litter-fall and logging debris accumulation, and development of fuel “ladders” of live and dead trees that are capable of conveying surface fires in to the crowns and canopies of forests (Covington and others 1994) made fire suppression more difficult. As the number and size of fires has increased over the last century (Dahms and Geils 1997), the emphasis on use of prescribed or “fire-use” fire has increased within land management agencies, with varying levels of success due to complex social, economic, and climatic factors (Zimmerman 2003).

There is an increase in fire intensity and severity in recent years, with large, stand replacing fires such as the La Mesa fire of 1977, Cerro Grande of 2000, and Rodeo-Chediski of 2002 serving as examples. Figure 7-5 portrays the dense growth of ponderosa pine along the Mogollon Rim (AZ) in 1923 following about 40 years of fire suppression.

Figure 7-5. 1923 photograph showing large number of ponderosa pine stems of small size along the Mogollon Rim, presumably as a result of fire suppression. Original photo caption: Remarkable growth of Ponderosa pine. Much of this growth was since the military abandoned its use of the Verde Rim Road. Photo by E. W. Kelley. FS #175776

Exotic Introductions (Plant & Animal) – See Herbivory, above. Several exotic plant species are beginning to colonize areas that have experienced high intensity wildfires (Crawford and others 2001).

Synthesis –

The most important anthropogenic disturbances for ponderosa pine forests include grazing, which removed fine fuels needed for carrying frequent, low intensity surface fires; silvicultural practices, which have changed forest age class distribution, composition, density, and cover values, and the cessation of frequent fire regimes, which prior to about 1900 ranged in frequency between 2 and 17 years. There are interactions between human-caused disturbance and climate, which may intensify or confound their effects.

7.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory – Fule and others (2003) described current forest structure for ponderosa pine at Grand Canyon National Park's north rim, and Cocke and others (2005) described current forest structure for ponderosa pine forests on the San Francisco Peaks. Table 7-5 displays reported values for the following ponderosa pine forest structure data by trees per acre, basal area, and percentage of basal area by tree species or group of species:

GCNP	ABCO	ABLA	PIEN	PIPO	POTR	PSME	RONE	Total
Trees/ac	80.7	44.0	1.1	92.4	90.1	8.3	0	316.7
Regeneration	672.9	20.2	0	429.9	932.8	20.2	0	2076.1
BA(ft^2/ac)	35.7	6.5	0.3	82.3	26.1	2.6	0	153.8
% BA	23.2	4.3	0.2	53.5	17.0	1.7	0	100
SFPA	**ABIES**	**PIAR**	**PIEN**	**PIPO**	**POTR**	**PSME**	**PIFL**	**Total**
Trees/ac	0	0.1	0.1	253.6	7.3	19.9	12.1	293.2
Regeneration	0	0	0	216.1	89.7	16.5	13.6	335.9
BA(ft^2/ac)	0	0	0.1	132.4	2.3	8.7	6.6	150.0
% BA	0	0	0	88.3	1.5	5.8	4.4	100

Table 7-5. Current forest structure determined for two ponderosa pine sites (GCNP=Grand Canyon National Park in 1880, SFPA=San Francisco Peaks in 1876) in Arizona. Basal area (BA) is expressed both in square feet per acre (ft^2/ac) and as percent of total. Species or groups across column labels are as follows: ABCO=white fir (*Abies concolor*), ABLA=corkbark fir (*Abies bifolia* formerly *A. lasiocarpa*), PIEN=Engelmann spruce (*Picea engelmanii*)+blue spruce (*Picea pungens*), PIPO=ponderosa pine (*Pinus ponderosa*), POTR=aspen (*Populus tremuloides*), PSME=Douglas-fir (*Pseudotsuga menziesii*), RONE=New Mexican locust (*Robinia neomexicana*), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (*Pinus aristata*), PIFL=limber pine (*Pinus flexilis*). Trees are defined as stems having dbh > 1 inch, and regeneration as stems having dbh <= 1 inch. Data for GCNP are from Fule and others (2003), and for SFPA are from Cocke and others (2005).

For the GCNP north rim site, the ponderosa pine forest has increased from 136.2 (se=20.1) trees/acre to 316.7 (se=47.5) trees/acre, not counting current regeneration, which is a 132% increase in tree density from pre-settlement times to present. Similarly, the basal area increased from 57.9 (se=23.1) ft^2/ac to 153.8 (se=20.0) ft^2/ac, representing a 165% increase in tree density (Fule and others 2003). For the San Francisco Peaks study, Cocke and others (2005) reported an increase in tree density from 24.9 (se=2.6) trees/ac to 293.1 (se=47.2) trees/ac, for a 1,079% increase, and basal area increased from 33.0 (se=4.9) ft^2/ac to 150.0 (se=13.1) ft^2/ac, for a 355% increase in current forest density over pre-settlement ponderosa pine forest density.

Understory- See *Herbaceous Layer*, below.

Herbaceous Layer – The effects of fire suppression and livestock grazing on understory and herbaceous vegetation is reviewed by Korb and Springer (2003), and effects of tree thinning and prescribed burning by Abella (2004). Generally, as the overstory density has increased, both total cover and species richness have decreased (Korb and Springer 2004). Also, thinning and burning generally increase ground flora (understory and herbaceous) biomass, but composition and populations processes (such as recruitment) have been little studied in ponderosa pine forests (Abella 2004).

Patch or Stand Structure of Vegetation – We found no studies that documented the effects of human disturbance on the patch or stand structure of ponderosa pine forests.

Canopy Cover Class (%) or Canopy Closure – Fule and others (2003) reported a value of 53% +/- 10.8% (se) for ponderosa pine at Grand Canyon National Park's North Rim.

Structure Class (Size Class) - Moore and others (2004) compared historical (1909-1913) versus current (1997-1999) size class structure of ponderosa pine forests from the 15 Woolsey plots in Arizona and New Mexico. They reported average current forest stand density (of trees >= 3.6 inches DBH) of 124.2 ft^2/ac. They also determined that the Quadratic Mean Diameter, a measure of central tendency in tree diameter within a stand weighted by number of trees, was 11.3 inches over the same time period, and that on average, each 1.1 hectare (2.8 ac) plot had 416 "young" trees, and 57.2 "old" trees [old and young based on morphology of tree bark] (Moore and others 2004).

Life Form – We found no studies that documented the effects of human disturbance on the current life form of ponderosa pine forests.

Density – The density of Southwest ponderosa pine forests has been reported to be 124.2 ft^2/ac for the Woolsey plots throughout Arizona and New Mexico (Moore and others 2004). More site specific studies have reported values of 153.8 ft^2/ac for the Grand Canyon National Park (GCNP) (Fule and others 2003), and 150.0 ft^2/ac for the San Francisco Peaks (Cocke and others 2005). Density expressed in trees per acre has been reported as 316.7 for stems > 1 inch DBH and 2076 for stems < 1 inch for the GCNP, for a total of 2392.7 trees/acre (Fule and others 2003). For the SFPA, Cocke and others (2005) reported 293.2 trees/acre for stems > 1 inch DBH and 335.9 for stems < 1 inch, for a total of 629.1 trees/acre.

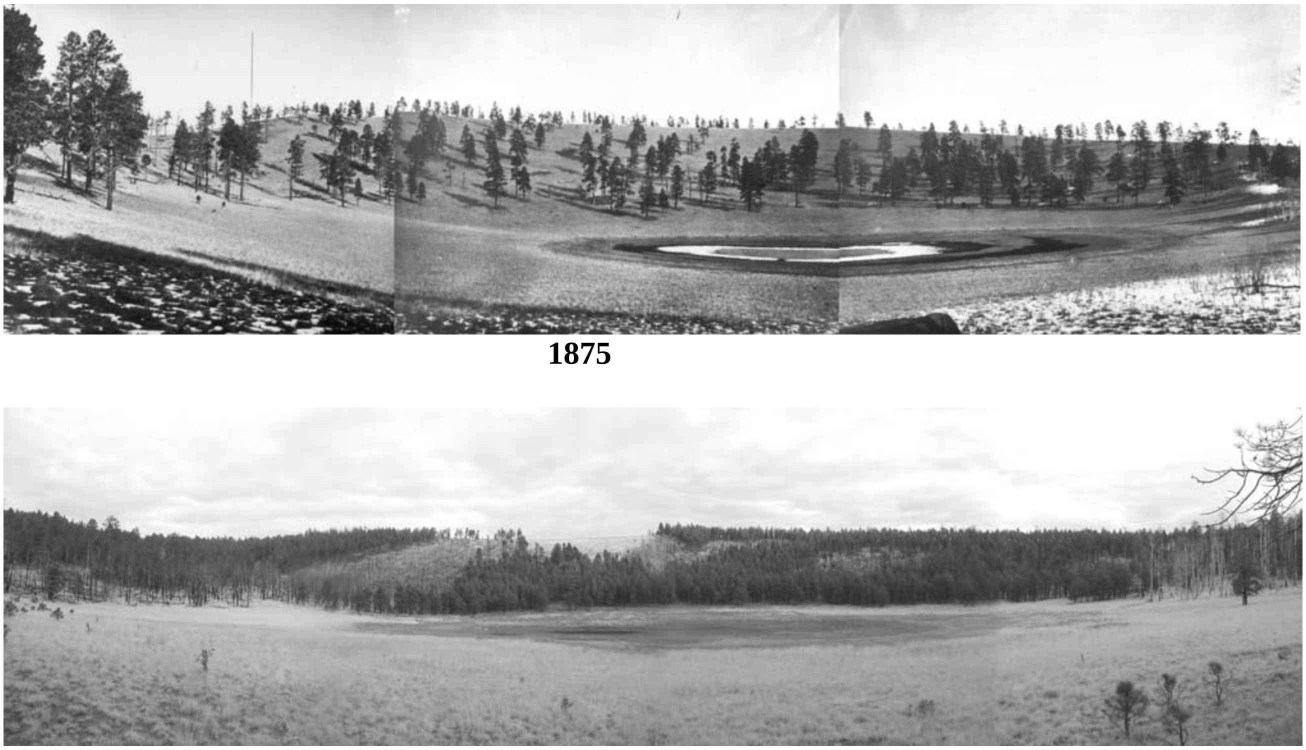
Age Structure – Generally, ponderosa pine forests have been shown to have more young trees in the understory (Fule and others 2003, Moore and others 2004, Cocke and others 2005). Moore and others (2004) determined that the Quadratic Mean Diameter, a measure of central tendency in tree diameter within a stand weighted by number of trees was 11.3 inches over the 1997 to 1999 time period, and that on average, each 1.1 hectare (2.8 ac) plot had 416 "young" trees, and 57.2 "old" trees [old and young based on morphology of tree bark].

Patch Dispersion - We found no studies that quantified the effects of human disturbance on the patch dispersion of ponderosa pine forests.

Recruitment Dynamics – Several authors have reported regeneration rates for the post-settlement period to be on the order of tens to hundreds of stems per acre (Covington and Moore 1994, Savage and others 1996, Moore and others 1999).

Synthesis – Anthropogenic disturbance has led to major changes in ponderosa pine forest structure and function. With the introduction of grazing animals at various times during the 19th century, low intensity and frequent surface fires have been replaced with high intensity and infrequent crown fires. Although the effects of these large, stand-replacing fires are variable, several fires have led to long-term changes from forested systems to grasslands, shrublands, and areas of dense pine regeneration (Savage and Mast unpublished data, Dahms and Geils 1997). Areas that have not burned yet have higher density of trees, especially of the smaller size class and younger age class, changing the quality of habitat for wildlife and humans (Covington and Moore 1994, Dahms and Geils

1997, Allen and others 2002). While good information exists for some areas that have the benefit of intensive study, many areas remain unstudied, and many data gaps remain that will help to ascertain reference conditions (Moore and others 1999, Allen and others 2002). Figure 7-6 is a photographic comparison 128 years apart in the Coconino National Forest, showing the change in density, spacing or clumpiness, and age class distribution.



1875

2003

Figure 7-6. Paired photographs from Walker Lake (Coconino NF) between 1875 and 2003 after approximately 128 years of fire suppression. Note the number, size and spacing of ponderosa pine in the upper photo, and the density increase by smaller trees in the lower photograph. Photos courtesy of Northern Arizona University's Ecological Restoration Institute.

7.6 Ponderosa Pine References

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Chapter 8 - Spruce-Fir Forest

8.1 General Description

The spruce-fir or subalpine coniferous forest type is found at high elevation, generally above 10,000 ft., but ranges widely, from 8,500 ft. on the Kaibab Plateau and Mogollon Mountains to 11,900 ft. in the Sangre de Cristo Mountains. This vegetation type covers approximately 2% of New Mexico, and less than 0.5% of Arizona, and is important ecologically for the high elevation plant, animal, and fungal communities it supports, and also economically as snow catchments, watershed areas, and for winter and summer recreational use (Moir and Ludwig 1979). According to southwest ReGAP data, spruce-fir forests cover approximately 611,000 acres in Arizona and New Mexico, of which 355,200 acres (or 58.1%) is on USFS lands on nine of the eleven national forests as portrayed in Table 8-1 (USGS 2003):

Apache-Sitgreaves	**Carson**	**Cibola**	**Coconino**	**Coronado**	**Gila**
18,500	174,900	10,700	7,200	*0	17,900
Kaibab	**Lincoln**	**Prescott**	**Santa Fe**	**Tonto**	**Total**
20,800	17,600	0	87,400	0	355,200

Table 8-1. Approximate area (in acres) of spruce-fir forest potential natural vegetation type (PNVT) across eleven Region 3 National Forests in Arizona and New Mexico. Region 3 National Grasslands in New Mexico, Oklahoma and Texas were not included in this analysis. Data used to generate this table included The Southwest Regional Gap Analysis Program (SWReGAP) and the landownership GIS-based layer. *Note that SWReGAP data have not been tested for accuracy and are derived from remote sensing; therefore, analyses at the individual National Forest scale may be inaccurate. For example, spruce-fir forests occur on the Coronado National Forest in the Pinaleno Mountains, however SWReGAP did not detect this vegetation type.

The dominant tree species in spruce-fir forest include Engelmann spruce (*Picea engelmanii* and *P.e ssp mexicana*), and sub-alpine fir (*Abies lasiocarpa*), and corkbark fir (*A. lasiocarpa* var. *arizonica*). Quaking aspen (*Populus tremuloides*) often serves as an important early seral species in spruce-fir forests, and occasionally Douglas-fir (*Pseudotsuga menziesii*) and/or white fir (*Abies concolor*) is present. The upper boundary in elevation is characterized by a shift to tundra species (i.e. forming tree line), although in some locations *krummholz* (German for ‘crooked tree’) habit Engelmann spruce may intergrade with bristlecone pine (*Pinus aristata*). Spruce-fir forest intergrades with mixed conifer forest at the lower boundary in elevation. There are few climatic data available for spruce-fir forest, but Alexander (1987) described the climate as cool, with a mean annual temperature of 35° F, ranging from 15 to 20°F in January and 50 to 60°F in July, for a frost-free period of 30 to 60 days. Mean annual precipitation ranges from 24-35+ inches with snowfall averaging 200+ inches (Alexander 1987). Along with short growing seasons, the spruce-fir forest typically has heavy snow accumulation, and soils with a cryic temperature regime (mean annual soil temperature at 20 inches depth is between 32 and 46°F) (Moir 1993).

Moir and Ludwig (1979) proposed a classification system for spruce-fir forests throughout Arizona and New Mexico that differentiates two series and eight habitat types (HT) based on the dominant tree type, the presence of reproduction in the understory, and shrub and herbaceous vegetation composition. Muldavin and others (1996) described a habitat typing system for a portion of Arizona for spruce-fir forests that delineates two series and ten habitat types, and the USFS (1997) proposed another system that includes Engelmann spruce, corkbark fir and bristlecone pine as one series, and describes 29 plant associations that fall within that series. For purposes of this study, we provide here the habitat types of Moir and Ludwig (1979).

The first of the two series is the *Picea engelmannii* Series, in which there are two habitat types:

1. The *Picea engelmannii/Vaccinium scoparium/Polemonium delicatum* HT is found primarily in the Sangre de Cristo Mountains, and has moderate to heavy regeneration (750 to 1000 stems/acre) of Engelmann spruce, and light to heavy regeneration (250-1000 stems/acre) of corkbark fir. Crown dominance is usually by spruce, and less commonly fir is co-dominant. The understory dominant is grouse whortleberry (*Vaccinium scoparium*), and the well developed herbaceous layer may be dominated by any of several species, such as Jacob's ladder (*Polemonium delicatum*), alpine groundsel (*Senecio amplectens*), or burnet ragwort (*Packera sanguisorboides*, formerly *Senecio sanguisorboides.*). Other characteristic species found in this HT include those found in tundra or tundra-forest ecotones: tufted hairgrass (*Deschampsia caespitosa*), nodding bluegrass (*Poa reflexa*), creeping sibbaldia (*Sibbaldia procumbens*), alpine clover (*Trifolium dasyphyllum*), and Eastwood's podistera (*Podistera eastwoodae*).
2. The *P. engelmannii/Moss* HT occurs in the Sangre de Cristo and San Juan Mountains, the Pinaleño Mountains, and San Francisco Peaks, and is characterized by a mixture of all sizes of Engelmann spruce, moderate or heavy regeneration of corkbark fir, but sometimes fewer mature fir than spruce. The understory is notable for a lack of shrubs and herbs and the dominance of mosses and lichens.

The second Series delineated by Moir and Ludwig (1979) is the *Abies bifolia* Series, in which there are six Habitat Types (HT):

1. The *A. bifolia/V. scoparium* HT occurs in the Sangre de Cristo, San Juan, Mogollon, and Pinaleño Mountains, and on Mount Baldy in Arizona. This HT usually has moderate to heavy regeneration of both spruce and fir, and sometimes mature Douglas-fir are common. Herbs are typically of low canopy cover.
2. The *A. bifolia/V. scoparium-Linnaea borealis* HT also occurs in the Sangre de Cristo Mountains, and characteristically has moderate to heavy regeneration by either or both Engelmann spruce and corkbark fir. Both Douglas-fir and white fir (*Abies concolor*) are seral species in this HT, and Douglas-fir is mostly dominant or co-dominant in the canopy overstory with Engelmann spruce. The understory is co-dominated by grouse whortleberry and twinflower (*Linnaea borealis*), and possibly falsebox (*Pachistima myrsinites*). This HT has a rich assortment of herbaceous species, including western red columbine (*Aquilegia elegantula*), rock clematis (*Clematis pseudoalpina*), splendid daisy (*Erigeron superbus*), Rocky Mountain strawberry (*Fragaria ovalis*), Parry's goldenweed (*Oreochrysum parryi* formerly *Haplopappus parryi*), roughleaf ricegrass (*Oryzopsis asperifolia*),

fringed brome (*Bromus ciliatus*), and common pink wintergreen (*Pyrola asarifolia*).

3. The *A. bifolia/Rubus parviflorus* HT occurs in the Mogollon Mountains of New Mexico, and is dominated by both *A. bifolia* and *P. engelmanii* with moderate to heavy regeneration of *Abies* and light to moderate regeneration of *Picea*. *P. menziesii* is seral in this HT, and thimbleberry (*Rubus parviflorus*) provides a well developed dominant in the shrub layer. There are many herbaceous species, including Richardson's geranium (*Geranium richardsonii*), Parry's goldenweed, osha (*Ligusticum porteri*), splendid daisy, or bittercress ragwort (*Packera cardamine*, formerly *Senecio cardamine*).
4. The *A. bifolia/Erigeron superbus* HT is widespread, occurring in the San Juan, Sangre de Cristo, Mogollon and White Mountains, along with Escudilla Mountain and the San Francisco Peaks. Corkbark fir has moderate to heavy regeneration in this HT, while Engelmann spruce has light to moderate regeneration, or less frequently, regeneration is either absent or heavy. Douglas-fir and white fir may have light to moderate regeneration, but their combined density is less than the combined density or *P. engelmanii* and *A. bifolia*. *Vaccinium* species are usually absent, and the understory is typically comprised mainly of herbaceous species, including splendid daisy, Parry's goldenweed, Richardson's geranium, fringed brome, aspen peavine (*Lathyrus lanszwertii leucanthus* formerly *L. arizonicus*), Canadian violet (*Viola canadensis*), bluntseed sweetroot (*Osmorhiza depauperata* formerly *O. obtusa*), Rocky Mountain strawberry, and ragweed sagebrush (*Artemisia franseroides*).
5. The *A. bifolia/Juniperus communis* HT occurs on the north Kaibab Plateau and the mountains of northern New Mexico, and both *A. bifolia* and *P. engelmanii* dominate the regeneration. Both Douglas-fir and white fir are seral for this HT, and understory shrubs and herbs are sparse, with the most constant species being common juniper (*Juniperus communis*) and one-sided wintergreen (*P. secunda*).
6. The *A. bifolia/Packera sanguisorboides* (formerly *Senecio sanguisorboides*) HT occurs in the Sacramento Mountains on Sierra Blanca peak, and is characterized by an abundance of *A. bifolia* in all sizes and classes, with moderate to heavy regeneration. *P. engelmanii* has moderate to low regeneration, and while *A. bifolia* usually dominates the canopy, *P. engelmanii* sometimes co-dominates. *A. concolor* is absent, and *P. menziesii* is seral only at lower elevations. The shrub layer is dominated by wolf's currant (*Ribes wolfii*) and gooseberry currant (*R. montigenum*). The herbaceous layer is rich and well developed, dominated by burnet ragwort, with other common species including osha, bluntseed sweetroot, red baneberry (*Actaea rubra*, formerly *A. arguta*), fringed brome, spike trisetum (*Trisetum spicatum*, formerly *T. montanum*), *Festuca scroria,* mountain parsley (*Cymopterus montanus*, formerly *Pseudocymopteris montana*), and splendid daisy.

8.2 Historical Range of Variation of Ecological Processes

Vegetation Dynamics – The spruce-fir forest has been studied very little in the Southwest, probably due to its limited distribution and lack of commercial timber value. Spruce-fir forests are much more extensive in the Rocky Mountains of Colorado, Montana, and Wyoming, and there is a substantial literature base for the more northern occurrences of this forest type. However, most of the central and northern Rocky Mountain spruce-fir forests probably have a very different evolutionary history including disturbance regimes due to major differences in patch size, soils, latitude, climate, and forest community composition, especially with the inclusion of lodgepole pine (*Pinus contorta*), which acts as an early seral species for the central and northern Rocky Mountain forests, but does not occur in the Southwest. Thus while some of the northern Rocky Mountain studies are included in this report for comparison, generally those studies from outside the Southwest are not considered to be applicable to Southwest spruce-fir forests.

The severe and cold environment found in high elevations generally reduces forest productivity, and slows succession. Most areas require hundreds of years to move from early successional stages to later, more mature stands (Moir 1993). In the central Rocky Mountains of Colorado, vegetation dynamics are more influenced by the type of disturbance than the spatial scale of the disturbance (Veblen 1986). As was noted in the Habitat Type descriptions, early seral species that establish after major disturbances such as fire, windthrow, avalanche, or insect outbreak are variable, and include aspen, Douglas-fir, bristlecone pine, and white fir, as well as the dominant Engelmann spruce and corkbark fir. Disturbance does not recur for a period of 70 to 100 years (or more) due to lack of sufficient fuels (for fire), or biomass (for insects, windthrow or avalanche) (Veblen and other 1994, Vankat 2006). Aspen is an important component of some of the Habitat Types for 50 to 150 years but declines rapidly in density and canopy dominance as the coniferous canopy cover increases (Moir 1993). Without disturbance, in the Rocky Mountains of Colorado, Engelmann spruce slowly increases in dominance in the canopy or overstory, while corkbark fir increases in abundance in the understory (Aplet and others 1988). The spruce-fir forest continues to grow and develop, but is increasingly susceptible to disturbance events. The average lifespan for tree species in the Southwest and southern to central Rocky Mountains was reported to be 300-350 years for corkbark fir, and 500 to 600 years for Engelmann spruce (Alexander 1987, Moir 1992). Currently, many spruce-fir forests on the north rim of Grand Canyon National Park (Vankat 2006) and the Pinaleno Mountains of southeastern Arizona (Koprowski and others 2005) are experiencing high mortality in older canopy trees due to a combination of drought and insects, especially of Engelmann spruce.

Disturbance Processes and Regimes

Climate- See Introduction section of the HRV document.

Fire- Both Engelmann spruce and corkbark fir are fire sensitive due to thin bark at all ages, and hence are unlikely to survive even low intensity fires and provide fire scars for analysis (Veblen and others 1994). Thus, most of the fire regime research has been accomplished using current tree and stand age or stand structure analysis (White and Vankat 1993, Fulé and others 2003), as well as fire scar analysis of adjacent forest types, which for spruce-fir is primarily mixed conifer forests at lower elevation (Grissino-Mayer and others 1995, Baisan 1995, Swetnam and others 2005). For the central Rocky Mountains of Colorado, spruce-fir forests burn as crown fires and at return intervals of centuries (Schoennagle and others 2004), although at lower elevation, some areas have experienced mixed severity and surface fires (Baker and Veblen 1990). Some have suggested that spruce-fir forests in the Southwest experience crown fire, with insufficient time having passed since the last crown fire for these forests to have experienced crown fire in contemporary time, at least for the small patch of spruce-fir forest on Mt. Graham in the Pinaleno Mountains (Swetnam and others 2005). Grissino-Mayer and others' (1995) analysis of stand structure in the Pinalenos led them to conclude that the fire return interval (FRI) for spruce-fir forest was 300-400 years. However, there is ample evidence to suggest that some spruce-fir forests in Arizona and New Mexico have a mixed severity fire regime that burned with a return interval on the order of decades rather than centuries (Dieterich 1983, Moir 1993, Fulé and others 2003, Vankat 2006).

At Grand Canyon National Park on the North Rim, Fulé and others (2003) reported that tree densities and groups of trees that were determined to be either fire-initiated or non-fire-initiated were very patchily distributed, and thus stands created by fire could not be discerned from analysis of satellite imagery. This diverse forest structure suggests a combination of a surface fire regime as well as crown-fire initiated groups of trees. They also reported that the mean fire interval (MFI) for spruce-fir was 8.8 years, 8.0 for 10% trees scarred, and 31.0 for 25% scarring (larger fires) for the period 1700 to 1879 (Fulé and others 2003). In addition, they reported a median fire return interval of 7 years, with minimum of 2 years and maximum of 32 years; a standard deviation of 7.7 years, and a Weibull median probability interval (WMPI) of 7.2 years. They found that upper elevation fire dates often coincided with fire dates for lower elevation forest types, suggesting that pre-1880 fires may have been very large (Fulé and others 2003). Most of the fires they studied occurred in summer, and large fire dates occurred in dry years that followed several wet years (Fulé and others 2003). Vankat (2006) asserted that the mixed severity fire regime of the North Rim spruce-fir forest might be a function of the relatively low elevation, but might also be a product of the Southwest's climate.

Shoennagel and others (2004) hypothesized that the main variables controlling fires in the central Rocky Mountains of Colorado shift from climate at higher elevation, crown-fire prone mesic sites to a combination of climate and fuel-related variables at drier mid-elevation, mixed fire regime sites. Fulé and others (2003) did caution that the mixed surface and crown fire regimes "appeared not to be stable over the temporal and spatial scales of this study." This suggests that the return interval itself may be dynamic, and the 279-year period for which they collected data may not have been sufficient to bracket the historical range of variability for fire return intervals for that site.

In a study of spruce-fir forests on Mt. Graham in the Pinaleno Mountains of SE Arizona, Swetnam and others (2005) studied age structure of existing stands, cross-dated with fire-scar dates from downslope mixed conifer trees, and determined that a widespread and severe crown fire occurred in 1685. Engelmann spruce and corkbark fir have been co-dominants since 1685, and there was a large pulse of corkbark fir recruitment in the mid-1800s (Grissino-Mayer and others 1995, Swetnam and others 2005). They also found that the growth rates of Douglas-fir trees that survived the 1685 fire at the spruce-fir/mixed conifer ecotonal boundary were slow growing after the fire, indicating that they had been damaged by a severe crown fire. They also determined from relative ring width that Engelmann spruce and corkbark fir that established after the fire (i.e., during the 1690s and early 1700s) had rapid growth rates, probably indicating an open stand condition. Tree-age structure and fire-scar evidence also pointed to varying periods of tree recruitment after the 1685 burn, possibly indicating trees' response to a combination of climatic variation and fire events (Swetnam and others 2005). They concluded that the dominant FRI for spruce-fir forest on Mt Graham is 150+ years, with large, high severity crown fire behavior. But, they conceded that "occasional surface and ground fires crept into portions of this forest from adjacent mixed-conifer, and high severity, small-patch size (individual trees or groups) events probably also occurred in this zone" (Swetnam and others 2005).

Vankat (2006) concluded that the mixed severity fire regime of spruce-fir forests in the Southwest may follow a moisture and/or elevation gradient similar to the difference between southern and central Rocky Mountain fire regimes. Swetnam and others (2005) also postulated the possibility that, historically, the long term stability of the spruce-fir forest was somewhat protected by the more frequent fire regime in the adjacent and down-slope mixed conifer forests. Frequent fires in the mixed conifer forests maintained more open stands, with low woody fuel accumulations, grassy understories, and higher crown base heights, which may have combined to produce lower intensity surface fires that would be less likely to transition into crown fires upon spread into the spruce-fir forests upslope. Swetnam and others (2005) have observed such fire behavior, and assert that the historic fire regime may have been responsible for maintaining a more patchy vegetation and fuel mosaic that was less conducive to crown fire.

Hydrology- We found no studies that documented hydrological processes such as flooding as important historical ecological determinants for the spruce-fir vegetation type.

Herbivory- We found no studies that documented herbivory as an important historical ecological determinant for the spruce-fir vegetation type.

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and introductions as important historical ecological determinants for the spruce-fir vegetation type.

Insects and Pathogens –Historically, spruce beetles (*Dendroctonus rufipennis*) have had the greatest impact on spruce-fir forests of the central and southern Rocky Mountains, and in the Southwest (Alexander 1987). Historic photographs and tree-ring analysis indicate that there have been six major outbreaks since the middle 1800s, and between 1850 and 1880 a very large-scale outbreak affected forests from northern New Mexico to northern Colorado (Baker and Veblen 1990). These outbreaks are considered part of the natural variability of spruce-fir forests, and spruce beetles likely persist in small windthrow areas (Veblen and others 1991) and in live trees (Veblen and others 1994). Small to large predators such as nematodes to woodpeckers maintain populations at low levels, until conditions favor an outbreak (Alexander 1987). Factors influencing outbreaks include large diameter Engelmann spruce or canopy dominance, slowed growth, mild winters, and well-drained creek bottom sites (Veblen and others 1994, Bebi and others 2003). Several root and stem decay fungi currently affect spruce-fir forests, and along with dwarf mistletoe species they are assumed to have been present in historic times, although no studies substantiate this (Dahms and Geils 1997).

Nutrient Cycling – We found no studies that documented nutrient cycling as an important historical ecological determinant for the spruce-fir vegetation type.

Windthrow – Wind has been documented as an important ecological factor in spruce-fir forests of Colorado, where windthrow or blow-down (trees knocked over by wind events) may affect up to 92% of old-growth areas (Alexander 1987, Veblen and others 1991). As mentioned above, spruce beetle populations may be maintained in small windthrow areas (Veblen and others 1992). We found no such similar studies for the Southwest.

Avalanche – We found no studies that documented avalanche as an important historical ecological determinant for the spruce-fir vegetation type. However, there are several apparent avalanche chutes that cut through spruce-fir forests on high elevation sites in the Southwest (E.g., see Figures 8-1 and 8-2 from the San Francisco Peaks), and these avalanche chutes likely contributed to the prevalence of early successional states of spruce-fir forest on steep slopes, and the juxtaposition of old and young age classes with abrupt boundaries (Figure 8-3).

Figure 8-1 Avalanche chute on north side of San Francisco Peaks in upper Abineau Canyon through spruce-fir forest. Avalanche occurred in spring of 2005. Photograph by Edward Smith on 11/06/05.

Figure 8-2. Avalanche chutes on north side of San Francisco Peaks in upper Abineau Canyon through spruce-fir forest. This area is approximately 200 meters upslope from Figure 8-1 above, and shows where the current avalanche began, and possible evidence of older avalanches in foreground and background. Photograph by Edward Smith on 11/06/05.

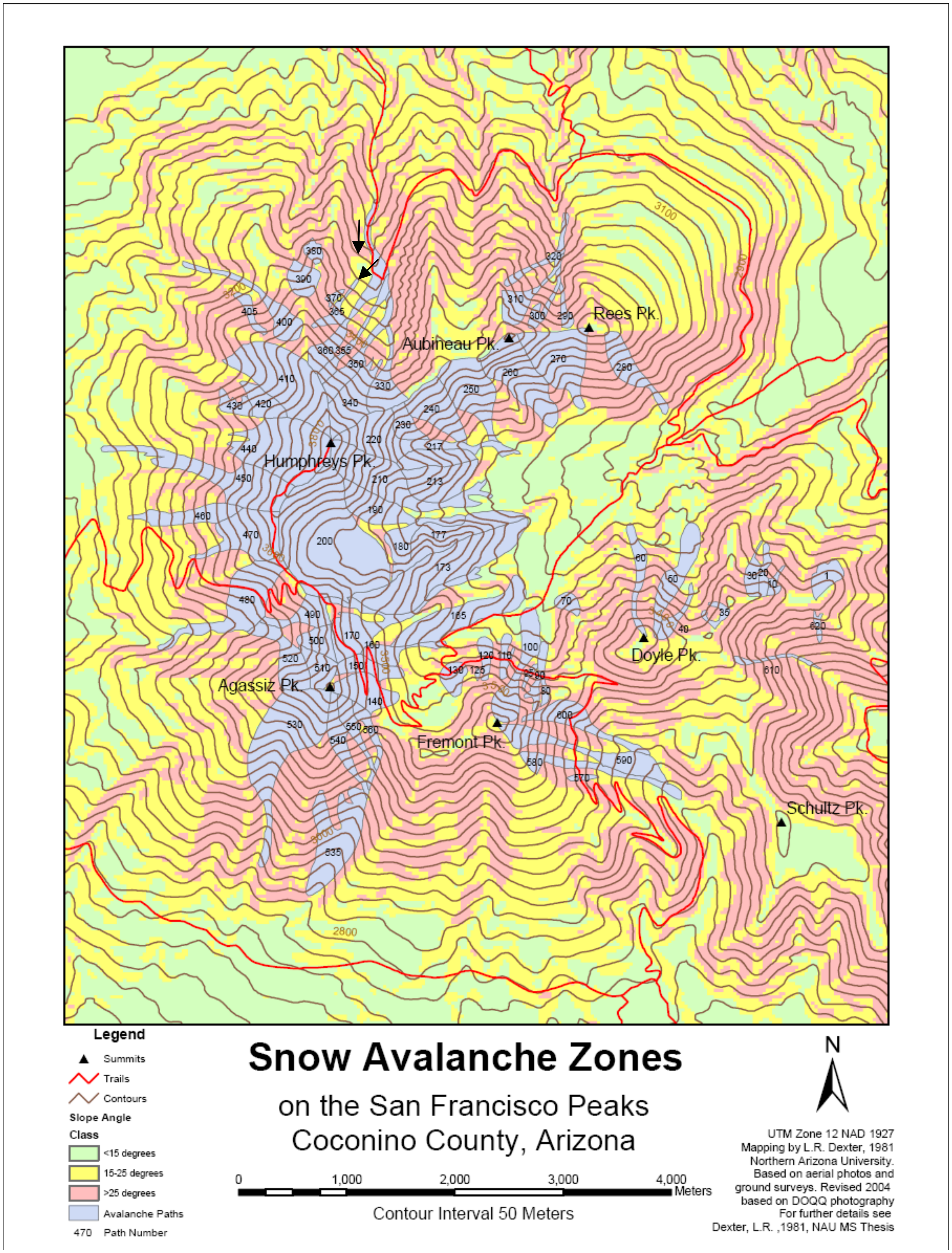

Figure 8-3. Map of snow avalanches (in purple) on San Francisco Peaks near Flagstaff, AZ. Most of these avalanches originate in tundra above treeline, pass through the spruce-fir forest and into the mixed conifer forest. Arrows indicate approximate location and direction taken for photos in Figures 8-1 and 8-2.

Erosion - We found no studies that documented erosion as an important historical ecological determinant for the spruce-fir vegetation type.

Synthesis – Historic fire regimes of the spruce-fir forest in the Southwest are typically of a mixed severity regime, with FRI ranging from 8-30 years for low intensity surface and ground fires, and upwards of 150 to 400 years for high intensity, stand replacing crown fires (Fulé and others 2003, Swetnam and others 2005, Vankat 2006). Based on Vankat (2006), Fulé and others (2003), and Swetnam and others (2005), we hypothesize that lower elevation (or latitude or moisture) spruce-fir forests may have more frequent, less severe fires more similar to surface fire regimes, and higher elevation/latitude/moisture forests may have infrequent, more severe crown fires, and some areas have both. Insects and pathogens possibly were important agents of change in these forests, as were abiotic factors such as windthrow and avalanche, although they have been poorly documented in the literature.

8.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation – We found no studies that documented the historical patch composition of spruce-fir forests.

Overstory – Fulé and others (2003) reconstructed forest structure from 1880 for spruce-fir forests at Grand Canyon National Park's north rim, and Cocke and others (2005) reconstructed forest structure from 1876 for spruce-fir forests on the San Francisco Peaks. Table 8-2 displays reported values for the following spruce-fir forest structure data by trees per acre, basal area, and percentage of basal area by tree species or group of species:

GCNP	**ABCO**	**ABLA**	**PIEN**	**PIPO**	**POTR**	**PSME**	**RONE**	**Total**
Trees/ac	4.1	4.8	15.8	5.8	24.4	5.8	N/A	60.7
BA(ft^2/ac)	4.8	3.9	17.0	3.9	3.9	8.7	N/A	42.3
% BA	11.3	9.3	40.2	9.3	9.3	20.6	N/A	100.0
SFPA	**ABIES**	**PIAR**	**PIEN**	**PIPO**	**POTR**	**PSME**	**PIFL**	**Total**
Trees/ac	25.1	35.6	40.1	0	3.0	1.8	0.9	106.4
BA(ft^2/ac)	11.5	35.1	16.0	0	0.5	4.2	0.2	67.4
% BA	17.0	52.0	23.7	0	0.7	6.3	0.3	100.0

Table 8-2. Historic forest structure reconstructed for two sites (GCNP=Grand Canyon National Park in 1880, SFPA=San Francisco Peaks in 1876) in Arizona. Basal area (BA) is expressed both in square ft. per acre (ft^2/ac) and as a percent of total. Species or groups across column labels are as follows: ABCO=white fir (*Abies concolor*), ABLA=corkbark fir (*Abies bifolia* formerly *A. lasiocarpa*), PIEN=Engelmann spruce (*Picea engelmanii*)+blue spruce (*Picea pungens*), PIPO=ponderosa pine (*Pinus ponderosa*), POTR=aspen (*Populus tremuloides*), PSME=Douglas-fir (*Pseudotsuga menziesii*), RONE=New Mexican locust (*Robinia neomexicana*), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (*Pinus aristata*), PIFL=limber pine (*Pinus flexilis*).

A frequent criticism of pre-settlement stand reconstruction methods has been that some dead and live standing and downed woody material could be lost to fire, logging, and decomposition in the intervening 100+ years between the time of the reconstructed stand

and when it was sampled (Foster and others 1996, Fulé and others 2003). The Grand Canyon National Park site was neither burned nor harvested over the reconstruction period (Fulé and others 2003), and Fulé and others (2003) performed a sensitivity analysis of their trees per acre reconstruction data by comparing their findings with data from Lang and Stewart (1910) from adjacent forest areas of similar composition. They reported that, for spruce-fir forest, the reconstructed data underestimated the Lang and Stewart data by 1.4 to 8.6%, indicating that the reconstructed estimates of tree density (in trees/acre) were reliable (Fulé and others 2003).

Understory - We found no studies that documented the historical understory composition of spruce-fir forests.

Herbaceous Layer- We found no studies that documented the historical herbaceous layer composition of spruce-fir forests.

Patch or Stand Structure of Vegetation – Swetnam and others (2005) found that by the end of the nineteenth century, the canopies of most spruce-fir forests at high elevations were relatively closed, with a broad mixture of old growth, middle-aged, and young trees.

Canopy Cover Class (%) or Canopy Closure – Swetnam and others (2005) compared spruce and fir age structure with average initial growth rates, and plotted this information with known fire dates. They reported that spruce-fir forests were “open canopy” from 1685 until about 1725, “closed canopy” from 1750 to about 1800, with some moderate stand thinning occurring until about 1900. We found no studies that documented numerical values for canopy cover of spruce-fir forests.

Structure Class (Size Class)-

We found no studies that documented the historical structure or size class of spruce-fir forests, although data may be available from Fulé and others (2003), Cocke and others (2005), and Abolt (1997) to determine historical structure class, age structure, and patch dispersion.

Life Form - We found no studies that documented the historical life form composition of spruce-fir forests.

Density - Fulé and others (2003) reconstructed spruce-fir forest at Grand Canyon National Park’s north rim, and determined an 1880-era basal area of 42.3 ft.^{2}/acre (standard error=5.2), and a tree density of 60.7 trees/acre (s.e.=6.0) for trees >1 inch diameter at breast height (dbh). Cocke and others (2005) reconstructed spruce-fir forests on the San Francisco Peaks, and determined an 1876-era basal area of 67.4 ft.^{2}/acre (s.e.=14.3), and a tree density of 106.4 trees/acre (s.e.=16.5) (Table 9-1).

Age Structure - We found no studies that documented the historical age structure of spruce-fir forests.

Patch Dispersion – We found no studies that documented the historical patch dispersion of spruce-fir forests.

Recruitment Dynamics - We found no studies that documented the historical recruitment dynamics of spruce-fir forests.

Reference Sites Used –Grand Canyon National Park's north rim (Fulé and others 2003), the San Francisco Peaks (Cocke and others 2005), and the Pinaleno Mountain's Mount Graham (Swetnam and others 2005) served as reference sites for this information. Other sites that may prove valuable as reference sites include the Valles Caldera National Preserve and the Southern Rocky Mountains in northern New Mexico.

Synthesis –Very few data are available for the historic condition of Southwest spruce-fir forests, although there is some information that indicates that spruce-fir forests ranged from moderately open to moderately closed, with density ranging from about 40-70 BA, and about 60-110 trees/acre. The composition of spruce-fir forests was complex and variable, with seven to nine species of trees represented in this complex forest type.

8.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory - It has been suggested that the extinction of large carnivores such as grey wolf and grizzly bear has affected at least one component of the spruce-fir vegetation type, quaking aspen, which experiences increased mortality due to high levels of herbivory by native and introduced ungulates, especially Rocky Mountain Elk (*Cervus elaphus*). Predator control has allowed herbivores to increase in numbers, and to exert longer duration and higher intensity grazing and browsing effects on select vegetation types, especially aspen (Shepperd and Fairweather 1994, Romme and others 1995, Kay 1997, Ripple and others 2001, Bailey and Whitham 2002).

Spruce-fir forests are affected by a variety of insects and pathogens, including Janet moth (*Nepytia janetae*), spruce beetle (*Dendroctonus rufipennis*), western balsam bark beetle (*Dryocoetes confusus*), the introduced spruce aphid (*Elatobium abietinum*), Douglas-fir tussock moth, the fir engraver beetle (*Scolytus ventralis*), and several root and stem decay fungi including *Armillaria*, annosus root rot (*Heterobasidion annosum*), and tomentosus root/butt rot (*Inonotus tomentosus*). The 2004 USFS Conditions Report shows an overall increasing trend in insect infestations. The overstory on Mt Graham is all but dead as a result of insect and drought mortality (Koprowski and others 2005).

Silviculture – Spruce-fir forests have been minimally impacted by silvicutural activities, primarily due to their inaccessibility on steeper slopes in remote areas, or through their protection in forest reserves such as at Grand Canyon National Park. Some spruce-fir forests have been logged in parts of the Southwest, although this is not well documented. Most logging activity occurred between 1870 and 1970, although in some areas logging activity continued up into the 1990s (Bahre 1998).

Fragmentation – Few data are available on fragmentation in spruce-fir forests, although the recent construction of astrophysical observatories on Mt Graham in the Pinalenos required the controversial clearing of spruce-fir forest (Istock and Hoffman 1995). Ski areas in the Southwest are frequently sited within spruce-fir forests, although the effects of ski areas on these forests have not been documented.

Mining –

We found no studies that documented mining as an important ecological determinant for the spruce-fir vegetation type.

Fire Management –

The disruption of historic fire regimes by introduced grazing animals has been well documented in southwestern ecosystems, and high elevation spruce-fir forests were well utilized as summer range for large numbers of sheep and cattle (Carlson 1969, Allen 1989, Covington & Moore 1994, Bahre 1998, Swetnam et al 1999). In the early 1900s, active fire suppression through the construction of fire lines and roads, and later concerted efforts with fire brigades and air tankers, began to function as the primary mechanism for excluding fire from Southwestern forests. Although the impact of humans on the fire regime, structure, and composition of lower elevation forests has been well documented (Allen and other 2002), the relative inaccessibility and lower commercial value of higher elevation forests have buffered them from some impacts (Swetnam and others 2005), but not all (Koprowski and other 2005), and limited the number of studies addressing their characteristics and ecological change.

Exotic Introductions (Plant & Animal) – Rocky Mountain elk (*Cervus elaphus*) were introduced to the Southwest in a series of introductions from Yellowstone National Park beginning in 1913 (Leopold 1990). Elk have been very successful in southwestern forests, and predator control has allowed herbivores to increase in numbers, and to exert longer duration and higher intensity grazing and browsing effects on select vegetation types, especially aspen (Shepperd and Fairweather 1994, Romme and others 1995, Kay 1997, Ripple and others 2001). Numerous elk browsing studies have documented their impacts on aspen regeneration (Shepperd and Fairweather 1994, Rolf 2001, Kay 2001, Bailey and Whitham 2002, Kaye and others 2005), leading some to suggest that successful aspen management should include adequate control of browsing animals (Shepperd and Fairweather 1993, Bartos 2001).

The introduced spruce aphid (*Elatobium abietinum*) is having an increasingly large effect on spruce-fir forests (Anhold and others 2004, Koprowski and others 2005).

Introduced white pine blister rust (*Cronartium ribicola*) continues to increase in the Sacramento Mountains of New Mexico, with approximately 40% of white pines showing signs of infection from this often fatal disease. Smaller outbreaks have been reported in the Capitan and Gallinas Mountains of New Mexico, and although it has not yet been reported in Arizona, southwestern white pine, limber pine, and bristlecone pine may be susceptible to infection (Anhold and others 2004).

Synthesis – Spruce-fir forests may have been altered by the extinction of large carnivores, the introduction of domestic and introduced wild ungulates, and fluctuations in populations of native and introduced insects and pathogens, but little empirical information exists to characterize the extent and magnitude of change resulting from these factors. These forests have experienced various levels of logging, road construction, and other sources of fragmentation, although these impacts are not well studied. The introduction of grazing animals has altered historic fire regimes, particularly at the lower elevation spruce-fir forests where fires were more frequent.

8.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory – Fulé and others (2003) described current forest structure for spruce-fir forests at Grand Canyon National Park's north rim, and Cocke and others (2005) described current forest structure for spruce-fir forests on the San Francisco Peaks. Table 8-3 displays reported values for the following spruce-fir forest structure data by trees per acre, basal area, and percentage of basal area by tree species or group of species:

GCNP	**ABCO**	**ABLA**	**PIEN**	**PIPO**	**POTR**	**PSME**	**RONE**	**Total**
Trees/ac	7.2	97.7	178.3	8.9	84.0	6.9	0	383.0
Regeneration	85.4	747.8	485.4	0	1695.1	52.2	0	3066.0
BA(ft^2/ac)	3.5	24.8	60.6	9.6	15.7	7.4	0	121.1
% BA	2.9	20.5	50.0	7.9	12.9	6.1	0	100
SFPA	**ABIES**	**PIAR**	**PIEN**	**PIPO**	**POTR**	**PSME**	**PIFL**	**Total**
Trees/ac	120.6	38.7	149.2	0	26.8	2.3	2.6	340.2
Regeneration	88.6	0	70.1	0	88.2	62.2	0	309.1
BA(ft^2/ac)	91.3	31.3	108.8	0	12.6	7.3	0.5	251.7
% BA	36.3	12.4	43.2	0	5.0	2.9	0.2	100

Table 8-3. Current forest structure determined for two sites (GCNP=Grand Canyon National Park, SFPA=San Francisco Peaks) in Arizona. Basal area (BA) is expressed both in square ft. per acre (ft^2/ac) and as a percent of total. Species or groups across column labels are as follows: ABCO=white fir (*Abies concolor*), ABLA=corkbark fir (*Abies bifolia* formerly *A. lasiocarpa*), PIEN=Engelmann spruce (*Picea engelmanii*)+blue spruce (*Picea pungens*), PIPO=ponderosa pine (*Pinus ponderosa*), POTR=aspen (*Populus tremuloides*), PSME=Douglas-fir (*Pseudotsuga menziesii*), RONE=New Mexican locust (*Robinia neomexicana*), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (*Pinus aristata*), PIFL=limber pine (*Pinus flexilis*). Trees are defined as stems having dbh > 1 inch, and regeneration as stems having dbh <= 1 inch.

Fulé and others (2003) censused spruce-fir forests at Grand Canyon National Park's north rim and determined a current-era basal area of 121.1 ft^2/acre (standard error=7.8), a tree density of 383.0 trees/acre (s.e.=40.12) for trees >1 inch (dbh), and a tree density of 3066.0 trees/acre for trees < 1 inch dbh. Cocke and others (2005) censused spruce-fir forests on the San Francisco Peaks and determined current-era basal area of 251.7 ft^2/acre (s.e.=18.3), a tree density of 340.2 trees/acre (s.e.=55.9) for trees >1 inch (dbh), and a tree density of 309.1 trees/acre for trees < 1 inch dbh. (Table 9-3).

Understory- We found no studies that documented the effects of human disturbance on the understory composition of spruce-fir forests.

Herbaceous Layer – We found no studies that documented the effects of human disturbance on the herbaceous layer composition of spruce-fir forests.

Patch or Stand Structure of Vegetation – We found no studies that documented the effects of human disturbance on the patch or stand structure of spruce-fir forests, although data may be available from Fulé and others (2003), Cocke and others (2005), and Abolt (1997) to determine current patch structure, structure class, age structure, and patch dispersion.

Canopy Cover Class (%) or Canopy Closure – Cocke and others (2005) reported canopy cover for current spruce-fir forests on the San Francisco Peaks, with a mean value of 58.2% (s.e.=4.6), a minimum of 28.1% and a maximum of 80.1%. We found no other studies that documented the effects of human disturbance on canopy cover for spruce-fir forests.

Structure Class (Size Class) - We found no studies that documented the effects of human disturbance on the structure class of spruce-fir forests.

Life Form – We found no studies that documented the effects of human disturbance on the life form of spruce-fir forests.

Density - For the GCNP North Rim site, the spruce-fir forest has increased from 60.6 (se=6.0) trees/acre to 383.0 (se=40.1) trees/acre, not counting current regeneration, which is a 532% increase in tree density from pre-settlement times to present. Similarly, the basal area increased from 42.3 (se=5.2) ft^2/ac to 121.1 (se=7.8) ft^2/ac, representing a 187% increase in basal area (Fulé and others 2003). For the San Francisco Peaks study, Cocke and others (2005) reported an increase in tree density from 106.4 (se=16.5) trees/ac to 340.2 (se=55.9) trees/ac, for a 220% increase, and basal area increased from 67.4 (se=14.3) ft^2/ac to 251.7 (se=18.3) ft^2/ac, for a 273% increase in current basal area over pre-settlement basal area.

Age Structure - We found no studies that documented the effects of human disturbance on the age structure of spruce-fir forests. One study (Popp and others 1992) delineated minimum criteria for the structural attributes used to determine old-growth and reported that spruce-fir stands should have living trees of 15 trees/ac of 12 inch dbh, or 20 trees/ac of 10 inch dbh (low end), up to 30 trees/ac of 14 inch dbh or 25 trees/ac of 16 inch dbh, with an age of 140 to 170 years. The range of snag (standing dead tree) density requirements is 3 trees/ac of 12 inch dbh at 20 ft tall, to 4 trees/ac of 16 inch dbh at 30 ft tall. They recommended that downed material should be 5 pieces/ac of 12 inch dbh and 16 ft long. They also recommended a blend of single-storied and multi-storied canopies, a range of basal areas of 120 to 140 ft^2/ac, and canopy cover of 60 to 70%.

Patch Dispersion - We found no studies that quantified the effects of human disturbance on the patch dispersion of mixed conifer forests.

Recruitment Dynamics - We found no studies that documented the effects of human disturbance on the recruitment dynamics of mixed conifer forests.

Synthesis – The overstory density at both the San Francisco Peaks and the Grand Canyon National Park sites has increased 220-530% in trees per acre over the 120-year period covered by the two studies. Similarly, the basal area has increased 190 to 270% in ft^2/acre over the same period, with most of the increase in density contributed by fire sensitive species such as white fir, corkbark fir, Engelmann spruce, and aspen. If regeneration (trees<1 inch DBH) is counted, the differences are even more striking, with even higher densities in the current time period (610% to 5,586% increase in trees per acre). Unfortunately, data are not yet available for concomitant changes in age- or size-class structural changes, nor for canopy cover changes, although the increase in density probably has also increased canopy cover values. The spruce-fir forest on top of the Pinalenos is undergoing massive die-off of mature trees, primarily due to drought, high density of trees and competition, and insect outbreak. With continued drought and fire suppression, more insect outbreaks are likely to occur in other portions of spruce-fir forest throughout the Southwest. The limited spatial distribution, inaccessibility, and high fuel loads of this important, mixed-fire regime adapted forest present difficult challenges for its management.

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Chapter 9 - Aspen Forest and Woodland

9.1 General Description – Trembling or quaking aspen (*Populus tremuloides* Michx.) ranges in occurrence from small discontinuous patches of tens to hundreds of acres to large, contiguous mountainsides and plateau-tops (thousands of acres) throughout Arizona and New Mexico, at elevations ranging from 5,500 feet to 11,500 feet. Aspen is a seral species in several coniferous habitat types, including six spruce-fir habitat types, and eight mixed conifer habitat types (Moir and Ludwig 1979), and mesic ponderosa pine forest, and in montane grasslands with fire exclusion or after heavy livestock grazing (Moir 1993, Allen 1989, Moore and Huffman 2001). Aspen can originate on scree slopes or active talus, where it forms small stands that are relatively persistent (Moir 1983). Aspen can also occur as a stable or persistent forest type, but this type is better documented in Utah and Colorado (Powell 1988). Aspen occurs throughout North America, with greater abundance in the northern Rocky Mountain States and Canadian Provinces, for which there is a rich literature (DeByle and Winokur 1985, Sheppard and others 2001). However, fewer studies have been published for aspen as it occurs in the Southwest (5 out of 42 papers in Sheppard and others 2001). Aspen forests occupy approximately 522,700 acres (or 0.35%) of the total combined land area of Arizona and New Mexico, of which 335,000 acres (or 64.36%) are under US Forest Service management on 10 National Forests, which comprises 1.7% of Region 3 National Forest land area (USGS 2004). The number of acres of aspen forest on each National Forest is found in Table 9-1:

Apache-Sitgreaves	**Carson**	**Cibola**	**Coconino**	**Coronado**	**Gila**
29,000	118,000	13,500	18,300	6,600	90,300
Kaibab	**Lincoln**	**Prescott**	**Santa Fe**	**Tonto**	**Total**
7,700	6,100	0	46,000	100	335,900

Table 9-1. Approximate area (in acres) of aspen forest potential natural vegetation type (PNVT) across eleven Region 3 National Forests in Arizona and New Mexico. Region 3 National Grasslands in New Mexico, Oklahoma and Texas were not included in this analysis. Data used to generate this table included The Southwest Regional Gap Analysis Program (SWReGAP) and the landownership GIS-based layer. Note that SWReGAP data have not been tested for accuracy and are derived from remote sensing; therefore, analyses at the individual National Forest scale may be inaccurate.

Aspen is very important from a biodiversity standpoint, supporting many species of birds and mammals either directly as forage, indirectly through the vast insect community it supports, or through the provision of structural habitat or nesting sites. Some consider aspen to be second only to riparian areas in biodiversity value (Kay 1997). Comprehensive classifications of aspen communities have been established for Idaho, Wyoming, Montana, Colorado, and Utah, and Muldavin and Tonne (2003) described two Plant Associations (PA) for the Quaking Aspen Alliance for upper elevations of the Valles Caldera National Preserve in New Mexico at >8,900 ft elevation. The Aspen/Meadow Rue PA is the most common and is characterized by closed canopies of aspen with few, if any, conifers co-dominating. The understory is typically a luxuriant herbaceous cover represented by a wide variety of mesic forbs and grasses such as meadow rue (*Thalictrum fendleri*), stickywilly bedstraw (*Galium aparine*), strawberry (*Fragaria virginiana)*, violet (*Viola canadensis*), geranium (*Geranium richardsonii*, *G. caespitosum*), deer sedge (*Carex rossii*), and fringed brome (*Bromus ciliatus*). Kentucky bluegrass (*Poa pratensis*), an exotic and invasive rhizomatous grass species, can dominate the understory. A Kentucky Bluegrass Phase of the association has been described where the Kentucky bluegrass exceeds 25% basal cover (it can exceed 60%). This association has been reported widely in the Rocky Mountains from Canada to the Southwest. Mueggler (1985) suggest that bluegrass-dominated types are some of the poorest among aspen communities in terms of wildlife habitat because of low plant species diversity.

In contrast, the Quaking Aspen/Thurber Fescue PA is dominated by native grasses and sedges including Thurber fescue, fringed brome, junegrass (*Koeleria micrantha*), and dryspike sedge (*Carex foenea*). Forb richness is lower than in the previous association and more representative of meadows with species such as vetch (*Vicia americana*), pea (*Lathyrus spp.*), and bluebell bellflower (*Campanula rotundifolia*). As Allen (1984) and Karmarkova et al. (1988) have pointed out, the understory composition suggests that this aspen association may represent an invasion of a montane meadow grassland by trees. Although relatively uncommon, this association has been reported elsewhere in Colorado (Hess and Alexander 1986; Karmarkova et al. 1988). For more discussion of community composition within spruce-fir, mixed conifer, and ponderosa pine habitat types, please see the corresponding HRV descriptions for those PNVTs (Chapters 8, 6, and 7 respectively).

9.2 Historical Range of Variation of Ecological Processes

Vegetation Dynamics – Aspen is considered a shade intolerant, disturbance dependent species, in that it is perpetuated on sites where fire, disease, avalanche or other stand replacing events are active (Bartos 2001). Aspen is also relatively short-lived, reaching maturity in 80 to 120 years (Bartos and Mueggler 1981, and few stems live beyond 300 years (Jones and Schier 1985). Aspen primarily reproduces asexually from underground root suckers that are stimulated by hormonal changes within an adult tree affected by disturbance (Bartos 2001). When a tree is disturbed, the flow of root-sprout inhibiting auxins is disrupted, leading to stimulation of sprouting by cytokinin (Bartos 2001). Reproduction by sexual means or seed production is rare in the Mountain West (Kay 1993), but can occasionally follow wildfire (e.g. in SE Arizona, see Quinn and Wu 2001). Figure 9-1 shows reproduction following wildfire in the Santa Fe Naional Forest.

Figure 9-1. Aspen response to wildfire. From RO-Historic Photos Image 28. 393909. High Country. By Charles Cunningham, April 1940 (Santa Fe NF).

Lateral roots connect aspen sprouts to their ‘parent’ tree for 10 to 25 years in Colorado, after which the trees become separate (Shepperd and Smith 1993). Following disturbance, aspen can produce 12,000 sprouts/acre (wildfire in New Mexico: Patton and Avant 1970) to 20,000 stems/acre (clear-fell in Arizona: Rolf 2001). Aspen quickly self-thins due to competition and shade intolerance: One study showed a 38% mortality rate for 3- and 4-year old suckers on clear-cut plots in Arizona (Jones 1975). Depending on the intensity of the disturbance event, and the proximity, fecundity and density of surviving conifers, and site characteristics, reseeding by conifers begins and shade-tolerant species successfully recruit into the understory of the aspens. Clonal characteristics (genetics) also affect stand structure and morphology, as well as climate, disturbance history (fire, insects, grazing), soil type, impacts of grazing and browsing animals, and the incidence of disease (Jones and DeByle 1985).

As the conifers grow to a position where they overtop the aspen, aspen sprouting decreases, and decay organisms begin to dismantle the senescing aspen trees (Jones and DeByle 1985). This process may take as little as 80 to 100 years on small, even-aged patches of aspen on poor sites, or it may take 200 to 300 years where aspen has experienced long intervals between disturbance events, and in the presence of multiple age classes (Jones and DeByle 1985). Figures 9-2 and 9-3 show photographs from 1954 and 1987 indicating succession of aspen to spruce-fir forest over a 33-year period.

Figure 9-2. Dense aspen overstory with spruce-fir forest in understory. Image30a: 476595. Retake of 193840 taken in 1924. Truchas Peaks from upper end of Hamilton Mesa. Note growth of aspen invading grasslands. Pecos wilderness Area. W.L. Hansen 5/25/54.

Figure 9-3. Aspen have been replaced by spruce-fire forest. Image30b: NM Museum of Natural History Field Number 76. Hamilton Mesa, Pecos Wilderness Area, Santa Fe National Forest, New Mexico. Retaken by B. Sallach. 7/27/87.

Disturbance Processes and Regimes

Climate- In an analysis of aspen stand age structure in spruce-fir and mixed conifer forests of the southern Rocky Mountians in northern New Mexico, Margolis (2003) determined that there were widespread, synchronous stand replacing fires that corresponded with drought determined by the Palmer Drought Severity Index (PDSI). Margolis (2003) also cross-dated aspen initiation dates with fire scars from adjacent conifer trees, and found that 50% of the reconstructed stand replacing fire events occurred prior to settlement (1870). Early growing season frost appears to have a strong effect on aspen in some areas (Jones and others 1985, Fairweather and others 2004). After leaf expansion, frost events of short duration can contribute to mortality, especially of lower elevation, drought stressed small patches (Fairweather and others 2004). More information on climate may be found in the Introduction section on Climate Analysis.

Fire- Fire regimes for aspen are determined by the adjacent or matrix PNVT, ranging at low elevation for ponderosa pine from 2 to 20 years FRI, to 10 to 30 years for mixed conifer at middle elevations, up to 30 to 300 years for spruce-fir forests. Both spruce-fir and mixed conifer forests have mixed severity fire regimes, experiencing both frequent, low intensity surface fires, as well as infrequent, stand-replacing crown fires (c.f., Chapters 6, 7, and 8 of HRV) (White and Vankat 1993, Grissino-Mayer and others 1995, Fule and others 2003). Thus aspen forest fire return interval (FRI) is affected by the surrounding forest. Aspen trees are very susceptible to cambium girdling with even low intensity fire, killing current stems but not the roots (Baker 1925). Abolt (1997) estimated size of burns for aspen to be about 4,000 to 10,000 acres.

Hydrology- We found no studies that documented hydrological processes such as flooding as important historical ecological determinants for the aspen vegetation type.

Herbivory- We found no studies that documented herbivory as an important historical ecological determinant for the aspen vegetation type.

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and introductions as important historical ecological determinants for the aspen vegetation type.

Insects and Pathogens –Swetnam and Lynch (1989) found that there have been 8 or 9 outbreaks of western spruce budworm (*Choristoneura occidentalis*) in northern New Mexico since 1700, with average return intervals of 30 to 40 years. Western spruce budworm populations periodically increase to outbreak proportions, and cause extensive defoliation, tree mortality and altered succession in several mixed conifer species. Lynch and Swetnam (1992) studied several old-growth mixed conifer sites in New Mexico and found evidence of multiple outbreaks of western spruce budworm, but found that outbreaks were not focused on old growth stands. These periodic outbreaks could have created openings within mixed conifer forest stands that offer light gaps suitable for aspen regeneration. The most injurious insects on aspen include larvae of the large aspen tortrix (*Choristoneura conflictana*) and the western tent caterpillar (*Malacosoma californicum*), which both cause defoliation of variable intensity depending upon climate, site characteristics, and age of the stand (Dahms and Geils 1997). Margolis (2003) claims to have evidence that recent outbreaks of western tent caterpillar in New Mexico are anomalous over the last 250 years, and may be due to recent warming trends. Several other species of insects as well as fungi currently use aspen forests, although relatively few kill or seriously injure aspen (Hinds 1985).

Nutrient Cycling - We found no studies that quantified historical nutrient cycling for the aspen vegetation type.

Windthrow - We found no studies that documented windthrow as an important historical ecological determinant for the aspen vegetation type in the Southwest. Wind has been documented as an important ecological factor affecting aspen in spruce-fir forests of Colorado, where blowdowns (trees knocked over by wind events) may affect up to 92% of old-growth areas (Alexander 1987, Veblen and others 1991). In spruce-fir forests containing aspen, windthrow may contribute to gap openings where aspen may have recruited into small openings.

Avalanche - We found no studies that documented avalanche as an important historical ecological determinant for the aspen vegetation type, although periodic avalanche may have created gap openings in spruce-fir and mixed conifer forests where aspen could regenerate.

Erosion - We found no studies that documented erosion as an important historical ecological determinant for the aspen vegetation type.

Synthesis **–** Aspen forests and woodlands are an important successional community in ponderosa pine, mixed conifer, and spruce-fir forests. Aspen forests and woodlands are primarily affected by fire, insects, herbivores and climate, and climate interactions with fire, insects, and herbivores. Aspen exists as a stable community type in other areas, but does not appear to have this characteristic in the Southwest (or at least has not been reported in the literature).

9.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation - We found no studies that documented historical patch composition of aspen forests.

Overstory - Fule and others (2003) reconstructed forest structure from 1880 for aspen forests at Grand Canyon National Park’s north rim, and Cocke and others (2005) reconstructed forest structure from 1876 for aspen forests on the San Francisco Peaks. Table 9-2 displays reported values for the following aspen forest structure data by trees per acre, basal area, and percentage of basal area by tree species or group of species:

GCNP	**ABCO**	**ABLA**	**PIEN**	**PIPO**	**POTR**	**PSME**	**RONE**	**Total**
Trees/ac	4.9	2.2	5.0	17.8	67.9	4.1	N/A	101.8
BA(ft^2/ac)	4.4	1.3	4.4	21.3	10.9	5.2	47.0	42.3
% BA	9.3	2.8	9.3	45.4	23.2	11.1	0	100.00
SFPA	**ABIES**	**PIAR**	**PIEN**	**PIPO**	**POTR**	**PSME**	**PIFL**	**Total**
Trees/ac	0.9	3.7	0.3	3.9	38.9	3.9	3.6	55.3
BA(ft^2/ac)	1.1	3.3	0.1	4.9	10.6	9.5	4.9	34.5
% BA	3.3	9.5	0.4	14.1	30.8	27.4	14.4	100.00

Table 9-2. Historic forest structure reconstructed for two sites (GCNP=Grand Canyon National Park in 1880, SFPA=San Francisco Peaks in 1876) in Arizona. Basal area (BA) is expressed both in square feet per acre (ft^2/ac) and as a percent of total. Species or groups across column labels are as follows: ABCO=white fir (*Abies concolor*), ABLA=corkbark fir (*Abies bifolia* formerly *A. lasiocarpa*), PIEN=Engelmann spruce (*Picea engelmanii*)+blue spruce (*Picea pungens*), PIPO=ponderosa pine (*Pinus ponderosa*), POTR=aspen (*Populus tremuloides*), PSME=Douglas-fir (*Pseudotsuga menziesii*), RONE=New Mexican locust (*Robinia neomexicana*), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (*Pinus aristata*), PIFL=limber pine (*Pinus flexilis*). Data for GCNP are from Fule and others (2003), and for SFPA are from Cocke and others (2005).

Aspen at the GCNP site comprised 66% of the trees/acre, but only 23% of the basal area, which was dominated by ponderosa pine. The total pre-settlement tree density was determined to be 101.8 trees/ac, and total basal area was 42.3 ft^2/ac. At the SFPA site, aspen trees were 70% of the total stems (55.3 trees/ac), and although aspen contributed only 31% of the total basal area (34.5 ft^2/ac) it was the dominant contributor to basal area with 10.6 ft^2/ac.

Understory - We found no studies that documented the historical understory composition of aspen forests.

Herbaceous Layer- We found no studies that documented the historical herbaceous layer composition of aspen forests.

Patch or Stand Structure of Vegetation – We found no studies that documented the historical stand structure of aspen forests. From some historic photographs, we see a single-storied stand of aspen (Figures 9-1 and 9-2), while others indicate a mixture of age or size classes (Figure 9-4).

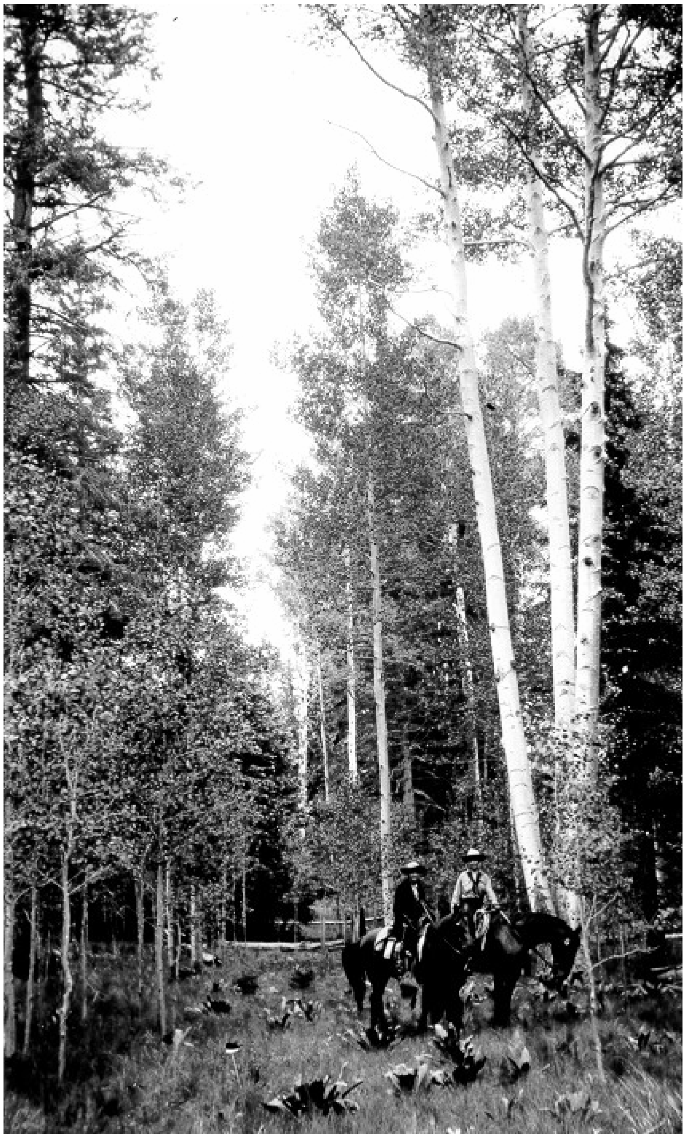

Figure 9-4. Early photograph of mixed age stand of aspen. Unknown date, unknown location (photograph #b1p020 from USFS Region 3).

Canopy Cover Class (%) or Canopy Closure - We found no studies that documented the historical canopy closure of aspen forests. We know from current sprouting response to burning and cutting that aspen re-sprouts from roots very vigorously, and that canopy cover is closed (>40%) within 3 to 5 years with stems reaching a height of 12 to 15 feet (Rolf 2001).

Structure Class (Size Class) - We found no studies that documented the historical structure class of aspen forests. Data used for Table 9-2 have not been analyzed for structure, size, or age class delineation.

Life Form - We found no studies that documented the historical life form composition of aspen forests.

Density – See *Overstory*, above, for historic density values.

Age Structure – Many aspen stands in the Southwest are all the same age, originating in a single large fire or a year of synchronous fires in the region (Jones and DeByle 1985, Abolt 1997, Margolis 2003). There were several large fires in 1904 in the White Mountains of the Apache-Sitgreaves National Forests of central-eastern Arizona that gave rise to aspen stands, and almost all aspen stands in New Mexico originated from fires that happened since the 1850s. However, Touchan and others (1996) analyzed fire scars in northern New Mexico and ages of living aspen trees, and found that within mixed conifer stands, aspen tree recruitment occurred sporadically from 1795 to 1918, with most of the recruitment occurring between 1850 and 1910. They found an imperfect correspondence between recruitment events and major fire years, leading them to hypothesize that most recruitment events from the late 19th century "correspond to relatively intense, patchy, stand opening fires" (Touchan and others 1996 p. 39).

Patch Dispersion – We found no studies that documented the historical patch dispersion of aspen forests.

Recruitment Dynamics – See discussion about recruitment in *Recruitment Dynamics*, above.

Reference Sites Used – Grand Canyon National Park's North Rim was used due to its longtime lack of logging and grazing (Fule and others 2002) and the San Francisco Peaks due to their fairly pristine condition (Cocke and others 2005). The Valles Caldera National Preserve is protected as of 2000, and although it has experienced periodic grazing, logging, and geothermal exploration, its lands and resources are being catalogued and studied extensively (Muldavin and Tonne 2003).

Synthesis –Aspen exists in small to large patches in the Southwest, that have resulted from a combination of intense stand replacing fires, and smaller surface fires, and other disturbance events. There are some data available on stand density and basal area from both Arizona and New Mexico, but little information on historic age or size structure of aspen stands.

9.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory – Several authors have noted the palatability of aspen to a wide variety of herbivores, from insects and other invertebrates (Jones and others 1985), to pocket gophers, and domestic and wild ungulates (DeByle 1985). Moore and Huffman (2001) reported aspen encroachment in subalpine grasslands of Grand Canyon National Park's North Rim, but noted a distinct gap in aspen recruitment during the Kaibab mule deer population explosion of the mid-1920s. Bailey and Whitham studied aspen stands affected by a wildfire in northern Arizona (2002, 2003), and noted the interactions of stand replacing fire, aspen recruitment, elk browsing, insect diversity, and insectivorous birds. They determined through sampling of elk exclosures that aspens sprouted better where burn severity was highest, but also that elk preferentially browsed where burn severity was highest. Thus intermediate-severity burn areas had better short-term survival of aspen sprouts, and supported greater richness and abundance of arthropods (Bailey and Whitham 2002). Bailey and Whitham (2003) further demonstrated that elk browsing reduced gall initiation between a sawfly and aspen sprouts, and that insectivorous birds preferentially attacked galls on unbrowsed galls. Bailey and Whitham (2002) reported that after three growing seasons, elk had consumed 36 to 85% of aspen shoots in the unfenced burned area. However, five years after this burn, not one of seventy regeneration plots outside of elk fences showed any living aspen sprouts, indicating heavy browsing by ungulates (Rolf unpublished data). In 1991, an elk-proof fence was removed from an aspen stand that was clear-felled and fenced in 1986 (Shepperd and Fairweather 1994). After one growing season, elk had reduced total stems in the newly unfenced area by 40%, although the authors cautioned that nearly all remaining stems less than 1.5 ft tall were damaged, as were half of the mid-size (1.5 to 4.5 ft) stems, and 60% of the large (>4.5 ft tall) stems. Most of the severely wounded stems also had infection with Cytospora canker (*Cytospora chrysosperma*) (Shepperd and Fairweather 1994). See figure 9-5 for appearance of elk damage to aspen trees including bark damage and fungal infection.

Figure 9-5. Damaged aspen with fungal infections on bark (black discolorations). RO-historic photo Image1. by Dr Mielke, Bureau of plant industry, unknown location in R3 unknown date. “Showing elk damage to aspen.”

Silviculture – Aspen is considered a minor commercial species in northern and central Rocky Mountain states, where some forests are intensively managed for commercial production (Jones and others 1985). However, aspen acreage is sufficiently small in the Southwest, that little to no commercial utilization has been supported; although fuel-wood permits have been used to treat some areas for regeneration purposes (Rolf 2001).

Fragmentation – We found no studies that documented fragmentation as an important ecological determinant for the aspen vegetation type.

Mining – We found no studies that documented mining as an important ecological determinant for the aspen vegetation type.

Fire Management –

The disruption of historic fire regimes by introduced grazing animals has been well documented in southwestern ecosystems, and high elevation aspen forests were well utilized as summer range for large numbers of sheep and cattle (Carlson 1969, Allen 1989, Covington & Moore 1994, Swetnam et al 1999). In the early 1900s, active fire suppression through the construction of fire lines and roads, and later, concerted efforts with fire brigades and air tankers, began to function as the primary mechanism for excluding fire from Southwestern forests (deBuys 1985, Allen 1989). As an early seral species, aspen reproduction and recruitment benefit from surface fires, and the reduction in occurrence of surface fires has reduced the amount of reproduction by aspen, and led to the replacement of aspen by other vegetation types (Jones and DeByle 1985). Severe fire is thought to be the primary agent resulting in even-aged aspen stands in the West (Jones and DeByle 1985). As a result of fire suppression activities, fire frequencies in all vegetation types that include aspen as a seral stage have decreased considerably, and in some sites, fire has not recurred since 1900 or before (Table 9-3). Fire sizes for post-settlement times have been estimated to be about one-fifth to one-seventh of pre-settlement fires, or about 600 to 2,000 acres (Abolt 1997).

Vegetation Type (PNVT)	**Pre-settlement MFI (SD)**	**WMPI**	**Range**	**Fire Cessation date**	**Location**	**Citation**
Ponderosa Pine	8	1.74-13.83	2-17	1860-1900	Several sites throughout AZ and NM	Swetnam and Baisan 1996
Ponderosa Pine		1.79-3.93	1-24	1896	Prescott NF, W-Central AZ	Sneed and others 2002
Mixed Conifer		9.7-14	4-32		Jemez Mtns., Northern NM	Touchan and others 1996
Mixed Conifer	11	4-6			Pinalenos, SE AZ	Grissino-Mayer 1995
Mixed Conifer	10	9.3	3-21	1876-1892	SF Peaks, Northern AZ	Heinlein and others 2005
Mixed Conifer	9.9				Rincon Mtns, SE AZ	Baisan and Swetnam 1990
Mixed Conifer	22				White Mtns., E Central AZ	Dieterich 1983
Mixed conifer/Spruce-Fir Transition		37			Gila NF, SW NM	Abolt 1997
Spruce-Fir	30-400 (est)				Coronado NF, SE AZ	Grissino-Mayer 1995

Table 9-3. Fire return interval data for ponderosa pine, mixed conifer, and spruce-fir forest types in Arizona and New Mexico from several studies. MFI is the mean fire return interval, WMPI is the Weibull distribution median probability interval, and pre- and post-settlement refer to a date selected within each study for the most likely date after which fires were less frequent and less discernible from either fire-scar or stand age structure analysis. Fire cessation date is the last year of a confirmed fire scar.

Exotic Introductions (Plant & Animal) – It has been suggested that the extinction of large carnivores such as grey wolf and grizzly bear has affected the aspen vegetation type, by increasing mortality on aspen sprouts (recruitment) due to high levels of herbivory by native deer and introduced ungulates, such as domestic cattle and Rocky Mountain elk (*Cervus elaphus*). Rocky Mountain elk were introduced to the Southwest in a series of introductions from Yellowstone National Park beginning around 1915 (Leopold 1990).

Elk have been very successful in Southwestern forests, and predator control has allowed herbivores to increase in numbers, and to exert longer duration and higher intensity grazing and browsing effects on select vegetation types, especially aspen (Shepperd and Fairweather 1994, Romme and others 1995, Kay 1997, Ripple and others 2001). Numerous elk browsing studies have documented their impacts on aspen regeneration (Rolf 2001, Kay 2001, Bailey and Whitham 2002, Kaye and others 2005), leading some to suggest that successful aspen management must include adequate control of browsing animals (Shepperd and Fairweather 1993, Bartos 2001).

Synthesis –There is widespread concern over aspen decline in the west (Romme and others 1995, Shepperd and others 2001). One study quantified a 46% decline in aspen cover in Arizona and New Mexico over a recent 25-year period (Johnson 1994), and some authors claim that aspen may face extirpation of local genotypes (Kay 1997, Bartos and Campbell 1998, Shepperd and Fairweather 1994). Several factors have been hypothesized as causal agents in the decline of aspen, including fire suppression, ungulate browsing, insects and pathogens, and interactions with climate. All of these factors are likely to contribute to aspen decline, and are likely to be interacting with each other at fine to coarse spatial scales, interacting with genetic and site variability of aspen stands throughout the Southwest (Romme and others 1995, Margolis 2003).

9.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory – As an early seral species, aspen benefits in its reproduction and recruitment from both low intensity surface fires and severe stand replacing fires. The recent reduction in occurrence of surface fires has reduced the amount of reproduction by aspen, and led to the replacement of aspen by other vegetation types (Jones and DeByle 1985). Stand replacing fire is thought to be the primary agent resulting in even-aged aspen stands in the West (Jones and DeByle 1985). Fule and others (2003) described current forest structure for aspen forest at Grand Canyon National Park’s North Rim, and Cocke and others (2005) described current forest structure for aspen forest on the San Francisco Peaks. Table 9-4 displays reported values for the following aspen forest structure data by trees per acre, basal area, and percentage of basal area by tree species or group of species:

GCNP	**ABCO**	**ABLA**	**PIEN**	**PIPO**	**POTR**	**PSME**	**RONE**	**Total**
Trees/ac	79.5	123.9	169.7	27.9	142.7	15.9	0	559.6
Regeneration	414.9	1285.8	825.1	19.0	2366.8	56.3	0	4968.0
BA(ft^2/ac)	15.7	13.5	33.5	16.6	49.2	9.6	0	137.1
% BA	11.4	9.8	24.1	12.0	35.7	6.9	0	100
SFPA	**ABIES**	**PIAR**	**PIEN**	**PIPO**	**POTR**	**PSME**	**PIFL**	**Total**
Trees/ac	21.1	14.2	18.7	3.4	175.3	44.5	33.9	311.2
Regeneration	390.4	0	0	1.9	799.6	47.5	37.5	1044.6
BA(ft^2/ac)	8.6	7.4	5.5	3.7	131.6	17.6	17.5	191.9
% BA	4.5	3.9	2.8	1.9	68.6	9.2	9.1	100

Table 9-4. Current forest structure determined for two aspen forest sites (GCNP=Grand Canyon National Park , SFPA=San Francisco Peaks) in Arizona. Basal area (BA) is expressed both in square feet per acre (ft^2/ac) and as a percent of total. Species or groups across column labels are as follows: ABCO=white fir (*Abies concolor*), ABLA=corkbark fir (*Abies bifolia* formerly *A. lasiocarpa*), PIEN=Engelmann spruce (*Picea engelmanii*)+blue spruce (*Picea pungens*), PIPO=ponderosa pine (*Pinus ponderosa*), POTR=aspen (*Populus tremuloides*), PSME=Douglas-fir (*Pseudotsuga menziesii*), RONE=New Mexican locust (*Robinia neomexicana*), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (*Pinus aristata*), PIFL=limber pine (*Pinus flexilis*). Trees are defined as stems having dbh > 1 inch, and regeneration as stems having dbh <= 1 inch. Data for GCNP are from Fule and others (2003), and for SFPA are from Cocke and others (2005).

For the GCNP North Rim site, the aspen forest has increased from 101.78 trees/acre to 559.60 trees/acre, not counting current regeneration, which is a 450% increase in tree density from pre-settlement times to present. Similarly, the basal area increased from 47.04 ft^2/ac to 137.65 ft^2/ac, representing a 193% increase in basal area (Fule and others 2003). For the San Francisco Peaks study, Cocke and others (2005) reported an increase in tree density from 55.26 trees/ac to 311.20 trees/ac, for a 463% increase, and basal area increased from 34.50 ft^2/ac to 191.90 ft^2/ac, for a 456% increase in current basal area over pre-settlement basal area.

Understory- We found no studies that documented the effects of human disturbance on the understory composition of aspen forests.

Herbaceous Layer – We found no studies that documented the effects of human disturbance on the herbaceous layer composition of aspen forests.

Patch or Stand Structure of Vegetation – We found no studies that documented the effects of human disturbance on the patch or stand structure of aspen forests.

Canopy Cover Class (%) or Canopy Closure - We found no studies that documented the effects of human disturbance on the canopy cover of aspen forests.

Structure Class (Size Class) - We found no studies that documented the effects of human disturbance on the structure class of aspen forests.

Life Form – We found no studies that documented the effects of human disturbance on the life form of aspen forests.

Density - We found no studies that documented the effects of human disturbance on the density of aspen forests, although see discussion in *Overstory*, above.

Age Structure - We found no studies that documented the effects of human disturbance on the age structure of aspen forests.

Patch Dispersion - We found no studies that quantified the effects of human disturbance on the patch dispersion of aspen forests.

Recruitment Dynamics – Several studies have indicated that due to a combination of fire exclusion and heavy ungulate browsing, aspen regeneration has declined (Johnson 1994, Romme and others 1995, Shepperd and others 2001).

Synthesis – Few studies have quantified the direct effects of anthropogenic disturbance on aspen forest characteristics. From the few studies that have been conducted comparing pre- and post-settlement stand density and composition, aspen stands are becoming more dense, and dominated by conifer species (Fule and others 2003, Cocke and others 2005). Also, aspen regeneration is reduced in many areas due to elimination of fire, and in those areas that have burned, aspen regeneration is largely ineffective (unless fenced) due to heavy browsing by domestic, wild, and introduced wild ungulates. Climatic events including drought and frost may be exacerbating effects of insects and fire suppression, leading to further declines of aspen forests and woodlands in the Southwest.

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Chapter 10 - Alpine Tundra

10.1 General Description – Tundra refers to a cold climate landscape or vegetation type that is above tree-line and is dominated by herbaceous or shrubby vegetation. The word tundra derives etymologically either from the Russian tundra, which means "land of no trees," or from the Finnish tunturi which means "arctic or barren hill" (Zwinger and Willard 1972) or "treeless plain" (Smith 1990). It is generally recognized that four distinct types of tundra exist on Earth, including Arctic Tundra, Alpine Tundra, Sub-arctic or Sub-alpine Tundra, and Maritime and Sub-antarctic Tundra (Rosswall and Heal 1974). In the Southwest, only Alpine Tundra is represented by several occurrences in the San Juan, Sangre de Cristo and Sacramento Mountains of New Mexico, and in the White Mountains and San Francisco Mountains of Arizona (Figure 10-1). Alpine tundra has the second lowest representation of all Potential Natural Vegetation Types by land area for the two-state region, comprising only about 7,700 acres, with about 1,600 acres falling under USFS management and the remaining 6,100 acres in private land ownership (USGS 2004). However, alpine tundra occurs in very small patches, and not every patch was detectable in the SWReGAP analysis (e.g., acreage is not listed for known areas in the Carson, Cibola, and Lincoln National Forests). Thus, the total land area of alpine tundra may be underestimated.

Alpine tundra occurs in the Southwest at elevations over 11,500 feet, and because of its relative inaccessibility and limited distribution, very few scientific studies have been published on its classification, structure, composition, and function (Moir 1993). A National Science Foundation funded Long-Term Ecological Research (LTER) site exists at Niwot Ridge, Colorado that encompasses alpine tundra vegetation (NW of Boulder in the Roosevelt National Forest, see http://culter.colorado.edu/NWT/index.html). Most of the research cited in this HRV description comes from published scientific work that was conducted at the Niwot Ridge site, unless noted from Arizona or New Mexico.

Figure 10-1. 1946—At timberline on the Blue Lake Trail [on the Carson National Forest, New Mexico. The trees are Englemann Spruce. Photo by R. King, FS **#440702. http://www.fs.fed.us/r3/about/history/carson/pages/car024_jpg.htm**

Alpine tundra in the Southwest has not been adequately studied or described from an autecological standpoint. Consequently, all alpine tundras in Arizona and New Mexico have been lumped into one single series, the *Acomastylus rossii* (formerly *Geum rossii*) series (Moir 1983), although Baker (1983) delineated ten community types on Wheeler Peak in New Mexico. At Niwot Ridge LTER, several systems have been proposed for delineating plant alliances, noda, or habitat types (Komarkova 1980, May and Webber 1982, Walker and others 2001).

Figure 10-2 depicts the currently accepted delineation of habitat types in alpine tundra based on snow distribution and topographic features (Walker and others 2001, after Billings 1973), for alpine tundra present in Colorado.

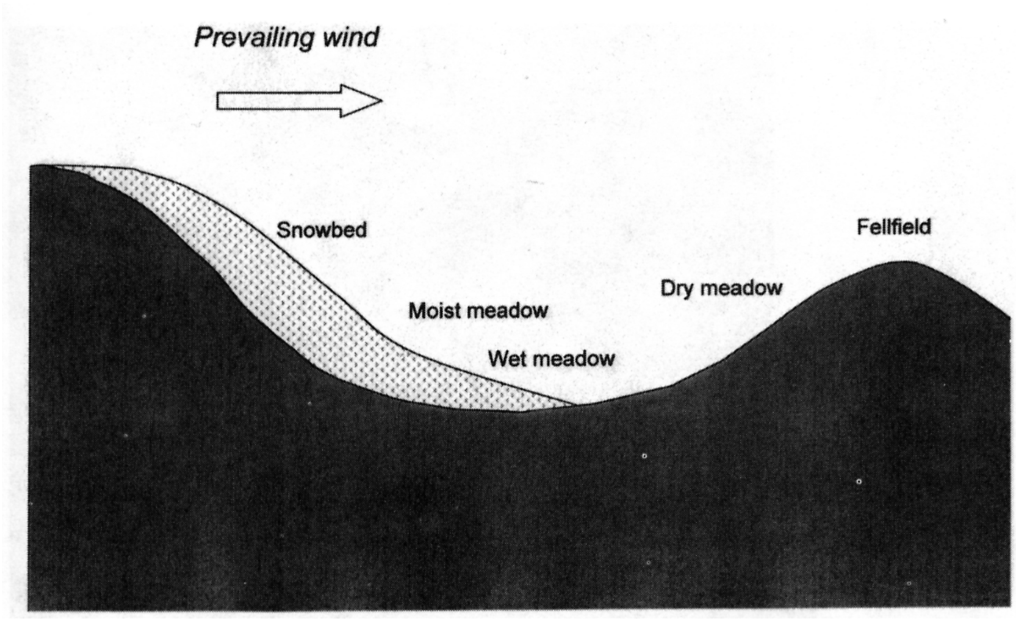

Figure 10-2. Habitat types for Alpine Tundra based on topographic position and snow deposition. (Figure from Walker and others 2001, after Billings 1973).

Fellfields occur on windblown slopes and ridge crests, have 10-50% of the ground surface covered with cobbles and exposed gravels, and have an open canopy and high diversity of cushion plants. Crustose lichens are very diverse in these sites, but bryophytes are uncommon. Fellfields remain snow-free throughout the year, although they may be covered with a thin crust of snow in depressions. **Dry Meadows** are dominated by bog sedge (*Kobresia*) and sedge (*Carex)* turfs. These areas typically have thin snow cover in winter that melts early in the season, providing for a relatively long growing season (150 to 200 days). **Moist Meadows** receive a modest snow cover that melts later in the spring, providing a shorter growing season of 100-150 days. These are in relatively rich and productive sites dominated by a lush cover of forbs and grasses, predominantly Ross' avens (*Acomastylis rossii*) and tufted hairgrass (*Deschampsia caespitosa*). **Late-melting Snowbank or Snowbed** communities as the name implies, experience harsh conditions due to the longer persistence of deeper snow, and have a growing season <100 days. These areas may be dominated by various willow species (*Salix spp.*) and sibbaldia (*Sibbaldia procumbens*) and sedges (*Carex spp*). Due to the longer duration of snowmelt, these sites often have saturated soils and plants that are adapted to standing water. **Wet Meadows** are often downslope of snowbeds, and have plants adapted to subhygric conditions, such as sedges, bistort (*Polygonum bistortoides*), ledge stonecrop (*Rhodiola integrifolia),* and white marsh marigold (*Caltha leptosepala*). **Shrub Tundra** areas are dominated by willow species, but also have bistort and stonecrop present, and line the edges of ponds and streams in lower alpine tundra areas.

Other habitat delineations have been described for all (Moir 1993) or part (Baker 1983) of New Mexico's alpine tundra vegetation. Moir (1993) described nine different New Mexican communities that suggest strong floristic and structural affinities with alpine tundra in Colorado. His delineation included a **Krummholz** community that is found at the lower elevation alpine tundra edge, and is dominated by Engelmann spruce

(*Picea engelmanii*) and subalpine fir (*Abies lasiocarpa*) of low stature (<3 feet). There is a strong shrub component in these low thickets. Moir also describes a **Fellfield** or **Rock Field** community that is similar in composition to Colorado fellfield described above. The **Kobresia Turf** is similar to the Colorado dry meadow, but in New Mexico kobresia shares dominance with Ross' avens, which is included in Colorado's wet meadows.

Moir (1993) asserts that the most common type of alpine tundra in New Mexico is what Baker (1983) described as the ***Carex rupestris*/Cushion** type. This type falls intermediate to the fellfield and kobresia communities, with many cushion plants dominating, but kobresia is minor or absent and Drummond's sedge (*Carex rupestris var. drummondiana*) is common and forms a sod. The wettest and most limited alpine tundras in New Mexico are the **Snowbank** and **Rivulet** communities, which border either melting snowbanks or small rivulets formed downslope of melting snowbanks and are both dominated by different shrub communities. Snowbanks are dominated by dwarf willows (*S. arctica* and *S. reticulata*) in low mats along with several species of lichens and herbaceaous plants, while rivulet communities are dominated by showy forbs such as groundsels (*Senecio spp.*), ledge stonecrop, rose crown (*Clementsia rhodantha*), and Parry's primrose (*Primula parryi*). These two wet communities resemble Colorado's wet meadow and shrub alpine tundras. Moir (1993) and Baker (1983) describe two other wet alpine communities that are either dominated by Drummond's rush (*Juncus drummondii*), found near melting snowbanks, or tufted hairgrass, or found on sites that have late-melting snowbanks with high insulation.

The last community type described by Moir (1993) is the **Rock Outcrop** or **Rubbleland** found at high elevation, which supports specialized plant communities such as talus or scree slope dependent plant species like the San Francisco Peaks groundsel (*Senecio franciscanus*), a federally listed Threatened species. Other species found in this habitat include (*Senecio atratus*), (*Ligularia soldanella*), (*Saxifraga chrysantha*), and (*S. flagellaris*). However, the most abundant vegetation in this habitat type is crustose and foliose lichens, whose abundance reflects whether the rock substrate is geomorphically active or inactive, with less lichen coverage correlated with more disturbance or snow coverage (Moir 1993).

The alpine tundra vegetation in Arizona is not as well studied as that in New Mexico. Zwinger and Willard (1972) assert that the alpine tundra on the San Francisco Peaks and the White Mountains of eastern Arizona is more closely related to that of the Great Basin, while New Mexico's alpine tundra is included with that of the southern Rocky Mountains. However, Moore's report (1965) indicates that, based on floristic analysis, alpine tundra community on the San Francisco Peaks is "definitely related to the alpine flora of the main Rocky Mountain peaks." However, Moore (1965) also indicated that the plant community of the San Francisco Peaks is "extremely poorly developed and floristically impoverished" in comparison with northern locations of alpine tundra in the more northern Rocky Mountains, indicating that it was isolated about 10,000 years ago. This finding led him to hypothesize that adverse conditions of drying and warming are diminishing the character of alpine tundra on the San Francisco Peaks, leading to retrogression of community diversity to a simpler flora (Moore 1965).

10.2 Historical Range of Variation of Ecological Processes

Vegetation Dynamics – Very little is known about the current condition and ecological processes, let alone the historic condition and processes of alpine tundra (Baker 1983, Moir 1993, Bowman and Seastedt 2001). Here we present a brief review of some of the physical and biological factors that are known to influence the patterns of alpine tundra vegetation.

Because of the location of alpine tundra in exposed, cold, and windy sites, physical factors such as topography, wind, snow drifts, and snow melt runoff are the major determinants of species and community distribution. Growing season length is determined by the amount of snow on a site, and snow depth is determined by aspect, slope, wind speed and direction. Snow cover can provide protection from extremely cold temperatures, frost damage, dehydration, and physical damage from snow and wind-blown particles, and snow cover of soil reduces intensive and deep soil freezing and subsequent frost heaving and soil weathering. Conversely, snow cover can reduce growing season length through soil cooling (Walker and others 2001). Wind is a major determinant of cushion-plant and tussock graminoid dominance in fellfields and exposed tundra turfs. These compact life forms reduce wind effects in winter and drought stress in summer, whereas plants that are protected by snow drifts in winter tend to have erect growth forms, soft leaves, and are not as drought tolerant (Walker and others 2001). The temperature of an alpine tundra community is determined by microtopography and slope position, and exhibits wide variation (Walker and others 2001). Soil moisture is a strong determinant of alpine tundra community type, and is dependent upon spatial and temporal patterns of precipitation and snow melt. Soil moisture also influences soil nutrient availability to plants (Walker and others 2001).

Biotic factors that affect alpine tundra plant communities include interspecific competition, facilitation, complex herbivore-plant interactions, and soil micro-flora and fauna. Interspecific competition has been documented for a few alpine tundra species, indicating that competition is important in structuring this vegetation type. Facilitation, or positive interactions among plants has been less studied, although in environments where physical stress levels are high, such as alpine tundra, it has been hypothesized that facilitation may be more important than competition through mechanisms of nurse plants that provide seed protection, seedling protection, and the decreased formation of needle ice detrimental to seedlings (Bertness and Calloway 1994, Walker and others 2001). Several herbivore species have been studied in alpine tundra, but little research has been focused on direct effects of herbivory on alpine tundra plant species or community composition. Below-ground herbivory by gophers has an indirect effect on vegetation through increased nutrient availability (especially nitrogen), and its direct effects on community composition through preferential grazing and disturbance: gophers prefer to graze on forbs over grasses, thus increasing the abundance of graminoids (Dearing 2001).

Disturbance Processes and Regimes

Climate- See *Introduction* for climate information.

Fire- We found no studies that documented fire as an important historical ecological determinant for the alpine tundra vegetation type.

Hydrology- We found no studies that documented hydrological processes such as flooding as important historical ecological determinants for the alpine tundra vegetation type.

Herbivory- Marmots, pikas, voles, and gophers are the primary herbivores that have had an impact on alpine tundra vegetation. Bighorn sheep once lived on the upper slopes of the San Francisco Peaks in Arizona and Wheeler Peak in New Mexico until about 1900, and bighorn sheep were successfully reintroduced to Wheeler Peak in 1993. Little is known about historic impacts of these herbivores, although current research at the Niwot Ridge site in Colorado has documented herbivory effects on vegetation. Marmots consume 2.6 to 6.4% of the available primary production in their foraging areas, with the majority of early forage consisting of grasses in the early part of the season, gradually being replaced by forbs and then seeds later in the season (Dearing 2001). Marmot burrows and adjoining areas are typically devoid of vegetation due to foraging and burrowing activities (Dearing 2001).

Pikas tend to forage out and away from their central rock and talus refuges, maintaining cushion plant communities (fellfield) in close proximity. They preferentially graze on forbs in adjacent moist and dry meadows (especially *A. rossii*) giving selective advantage to graminoids (Dearing 2001). Pathways created by these herbivores also alter the landscape through soil compaction and vegetation removal. Burrowing activities can cover and kill plants (Huntly and Inouye 1988), and accelerate decomposition; this type of chronic disturbance lowers carbon storage in the upper soil horizon, reducing long-term soil fertility (Cortinas and Seastedt 1996, Dearing 2001). However, small herbivores also concentrate soil nutrients and aid nutrient transport and mineralization through plant consumption and subsequent deposition of about 50% of consumed biomass in urine and feces, but typically in localized areas near burrows (Dearing 2001). With most herbivory pressure exerted by small mammals on forbs, graminoids should be more dominant on the alpine tundra landscape. However, forbs may competitively exclude graminoids through faster growth rates, or other herbivores that preferentially consume graminoids (such as grasshoppers and elk) may counteract the effects that have been quantified for gophers, pika and marmots, through complex interactions that have yet to be quantified (Dearing 2001).

Predator/Prey Extinction and Introductions - We found no studies that implicated predator/prey extinctions and introductions as important historical ecological determinants for the alpine tundra vegetation type.

Insects and Pathogens – We found no studies that documented insects and pathogens as important historical ecological determinants for the alpine tundra vegetation type.

Nutrient Cycling - We found no studies that documented nutrient cycling as an important historical ecological determinant for the alpine tundra vegetation type.

Windthrow - We found no studies that documented windthrow as an important historical ecological determinant for the alpine tundra vegetation type.

Avalanche - We found no studies that documented avalanche as an important historical ecological determinant for the alpine tundra vegetation type.

Erosion - We found no studies that documented erosion as an important historical ecological determinant for the alpine tundra vegetation type.

Synthesis – There have been very few studies on the HRV of alpine tundra ecological patterns and processes, although some work has been conducted in Colorado. The factors that influence the spatial distribution of different alpine tundra types have been well defined for the southern and central Rocky Mountains. Some of the mechanisms of native herbivore influence on alpine tundra have been studied, but most ecological processes have not yet been studied.

10.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation - We found no studies that documented historical patch composition of the alpine tundra vegetation type.

Overstory – Not applicable.

Understory - We found no studies that documented the historical understory composition of the alpine tundra vegetation type.

Herbaceous Layer- We found no studies that documented the historical herbaceous layer composition of the alpine tundra vegetation type.

Patch or Stand Structure of Vegetation – We found no studies that documented the historical stand structure of the alpine tundra vegetation type.

Canopy Cover Class (%) or Canopy Closure – Not applicable.

Structure Class (Size Class) - We found no studies that documented the historical structure class of the alpine tundra vegetation type.

Life Form - We found no studies that documented the historical life form of the alpine tundra vegetation type.

Density - We found no studies that documented the historical density of the alpine tundra vegetation type.

Age Structure - We found no studies that documented the historical age structure of the alpine tundra vegetation type.

Patch Dispersion – We found no studies that documented the historical patch dispersion of the alpine tundra vegetation type.

Recruitment Dynamics - We found no studies that documented the historical recruitment dynamics of the alpine tundra vegetation type.

Reference Sites Used – All current alpine tundra sites are potential reference sites, however, since there are no historical data from the pre-settlement period, no reference sites were identified.

Synthesis – Because alpine tundra is primarily herbaceous vegetation consisting of forbs and graminoids, there remains little evidence of historical vegetation patterns. Due to the limited distribution of alpine tundra at high elevation and in inaccessible areas, there are few historic photographs documenting the appearance of historic vegetation.

10.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory – Domestic sheep and cattle have grazed many areas of alpine tundra in the Rocky Mountains since Euro-American settlement, although cattle are susceptible to a high-altitude induced disease affecting survivorship that has precluded widespread grazing (Bowman and others 2002). Sheep have caused trampling, trail-cutting, and erosion when the animals were allowed to congregate (Bowman and others 2002). The extirpation of grizzly bears in the southern Rocky Mountains may have contributed to decreased vegetation and surface disturbance through elimination of bears' excavation of roots and tubers in alpine meadows, reducing creation of early successional states in small patches (Bowman and others 2002). Also, Rocky Mountain elk may have increased in abundance due to lower predation pressures, and higher elk populations have had detrimental effects on willow communities in Rocky Mountain Park, CO (Bowman and others 2002).

Silviculture – Not Applicable.

Fragmentation – Recreation impacts have been noted but not quantified for alpine tundra. Backpacking, alpine skiing, climbing, and sightseeing are all activities that occur in alpine tundra ecosystems, and have the potential to fragment the alpine landscape primarily from trampling of vegetation and erosion (Bowman and others 2002). Hartley (1976) studied the effects of trampling on alpine and sub-alpine vegetation in Glacier National Park, Montana, and found that 35 alpine species decreased as the trail system grew, and seven species increased. Some of the plants that increased were "semi-weedy" invasive native plants from lower elevation. Hartley (1976) also determined that stored carbohydrates decreased 20 to 50% within 0.5m of the trail, compared to >2m away from the trail, suggesting that trampling or trail proximity reduced plants' ability to make and store reserves, thereby reducing vigor.

Mining – We found no studies that documented mining as an important ecological determinant for the alpine tundra vegetation type in the Southwest. Many alpine tundra sites in Colorado have been affected by precious metal and molybdenum mines (Bowman and others 2002), and there is a historic photograph of a molybdenum mine from the Carson National Forest historic archives, but this 1941 mine appears to be at lower elevation than alpine tundra (Figure 10-3).

Figure 10-3. 1941—A Molybdenum mine near Red River [on the Carson National Forest, New Mexico]. About 75 men worked at this mine. Photo by E. O. Buhler, FS #413400. http://www.fs.fed.us/r3/about/history/carson/pages/car008_jpg.htm

Fire Management – Not applicable.

Exotic Introductions (Plant & Animal) – Trout stocking of alpine lakes has occurred in Colorado and other western states, and except for mention of exotic trout stocking of alpine lakes on the Carson National Forest, no data were available for the SW (http://www.fs.fed.us/r3/carson/recreation/wilderness/wheeler_peak_info.shtml). Non-native fish introductions can dramatically change native communities and extirpate amphibians, fish, invertebrates, and zooplankton (Bowman and others 2002).

Synthesis – Some generalized trends in grazing pressure and exotic introductions have been documented in Colorado, but no studies focused on the Southwest. Mining and recreation activities are also known to influence alpine vegetation, but no studies mentioned any specific impacts for the Southwest. Carbon and nitrogen from anthropogenic sources are deposited in alpine tundra systems. Carbon dioxide (CO2) levels have been monitored at Niwot Ridge since 1968, the longest record of atmospheric CO2 in North America, and have increased from 322 parts per million (ppm) to almost 370 ppm over thirty years (Sievering 2001). The anthropogenic contribution of total annual N deposition is estimated to be about half of the 5 lbs. N/acre that falls as wet and dry forms of N (Sievering 2001).

10.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory – Not applicable.

Understory- We found no studies that documented the effects of human disturbance on the understory composition of alpine tundra vegetation.

Herbaceous Layer – Due to the extreme environment of alpine tundra systems, recovery of vegetation following disturbance is thought to be very slow, on the order of hundreds to thousands of years (Bowman and others 2002). In one of the few long-term studies of recovery by Alaskan tundra after more than 20 years following disturbance caused by fire and bulldozing, there were distinct differences between areas affected by the two disturbance types, and between disturbed areas and undisturbed areas (Vavrek and others 1999). Primary productivity, species richness, and species diversity were not different between burned and unburned plots, but depth of thaw was greater in burned plots. By contrast, depth of thaw was the only factor that was not significantly different in bulldozed versus non-bulldozed plots. Primary productivity and species richness were higher in bulldozed plots, but diversity was higher in the control plots (Vavrek and others 1999). This study indicated that the effects of the anthropogenic disturbance on the vegetation are more important than changes in the abiotic environment. Vegetative propagules remained in the soil following fire, but did not remain in the soil following bulldozing. Thus, both seeds and vegetative propagules were involved in recolonization after fire, but only seeds were involved in recovery following bulldozing (Vavrek and others 1999).

Patch or Stand Structure of Vegetation - We found no studies that documented the effects of human disturbance on the patch or stand structure of alpine tundra vegetation.

Canopy Cover Class (%) or Canopy Closure - We found no studies that documented the effects of human disturbance on the canopy cover of alpine tundra vegetation.

Structure Class (Size Class) - We found no studies that documented the effects of human disturbance on the structure class of alpine tundra vegetation.

Life Form – We found no studies that documented the effects of human disturbance on the life form of alpine tundra vegetation.

Density - We found no studies that documented the effects of human disturbance on the density of alpine tundra vegetation.

Age Structure - We found no studies that documented the effects of human disturbance on the age structure of alpine tundra vegetation.

Patch Dispersion - We found no studies that quantified the effects of human disturbance on the patch dispersion of alpine tundra vegetation.

Recruitment Dynamics - We found no studies that documented the effects of human disturbance on the recruitment dynamics of alpine tundra vegetation.

Synthesis – Because there is so little information on historic alpine tundra vegetation, we found no studies linking human disturbance to changes in conditions of alpine tundra from pre-settlement to current. One study that tracked recover of Alaskan tussock tundra follwing disturbance concluded that recovery of vegetation was most affected by the type of disturbance and its effect on the vegetation rather than the effect of the disturbance on the abiotic environment (Vavrek and others 1999).

The areas of inquiry showing the greatest potential for hypothesis testing of environmental effects of human disturbance include climate change, changes in N deposition, and changes in atmospheric ultraviolet light transmission that have been monitored during the post-settlement period. Due to its high elevation, and lack of land area upslope, alpine tundra is uniquely susceptible to changes in climate, particularly if temperatures continue to increase (Welker and others 2001). However, despite recent warming trends, repeat photography indicates tree-line elevation has been stable in both the northern Rocky Mountains of Montana and Wyoming (Butler and others 1994) and central Rocky Mountains of Colorado (Baker and others 1995) since at least the beginning of the 20th century, indicating that alpine tundra is slow to respond in spatial distribution to climate change (Bowman and others 2002). However, recent work in Switzerland alpine vegetation indicates that community composition has changed (increased species richness) over the last century, and that the rate of change has increased in the last 20 years when compared to the first 80 years of the 20th century (Walther and others 2005).

Increased atmospheric N deposition is important for several reasons, and has become the subject of intensive study recently (Welker and others 2001). Due to the cold temperature and short growing season of alpine tundra ecosystems, N cycling rates are relatively low, and primary productivity is also lower than other montane systems. Thus, increases in N supply due to elevated deposition levels have the potential to impact plant and microbial community composition through selection for N-utilizing plants and microbes, and to increase N losses to downslope aquatic and other terrestrial ecosystems (Welker and others 2001). Conceptual models have been developed to explain and predict consequences of climate change (Grant and French 1990), N-deposition (Welker and others 2001), and the interactions of climate, hydrology-geochemisrty, and geomorphology-paleoecology (French and others 1986) on Front Range alpine tundra in Colorado.

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Chapter 13 - Vegetation Models for Southwest Vegetation

13.1 Introduction

In response to the USDA Forest Service Southwest Region's need for landscape scale planning tools, we developed broad-scale state and transition models for 8 Potential Natural Vegetation Types (PNVTs) in the Southwest based on a comprehensive literature review. We utilized this information to describe vegetation model states, identify parameter values for these models and to run quantitative scenario analysis, using Vegetation Dynamics Development Tool (VDDT) software, to determine the relative proportion of model states on the landscape. Vegetation Dynamics Development Tool software is a non-spatial model that allows the user to model vegetation change over time as a series of vegetation states that differ in structure, composition, and cover and to specify the amount of time it takes to move from one vegetation state to another in the absence of disturbance. Various disturbance agents affecting the movement of vegetation between states (or transitions) are incorporated (e.g., surface fires, stand-replacing fires, grazing, insect outbreaks, and drought events). By varying the types and rates of disturbance across the landscape, the effects of different disturbance regimes, such as historic and current fire regimes, or different management treatments, such as wildland fire use, fire suppression, prescribed burning, grazing practices, and mechanical fuel treatments, on vegetation can be investigated. These models will summarize and synthesize the current state of scientific knowledge of vegetation dynamics. Additionally, they will provide forest planners and managers with powerful tools for understanding, investigating, and demonstrating the effects of alternative scenarios for the management of vegetation on national forests at scales ranging from the Ranger District to the Southwest Region.

The region-wide scale at which the models were constructed, as well as the sole reliance on published scientific information to build and parameterize the models, necessarily limits the level of detail in a model as well as the applicability of the model to a given site. Given these constraint, it is important to utilize information from these models to understand general trends in vegetation change and dynamics at large scales while utilizing finer scale models (such as those found in Ecological Site Descriptions developed by the Natural Resources Conservation Service) and/or expert information to model and evaluate land management at the site level.

13.2 Methodology

State and Transition Models - We defined all model states, transitions between states, and transition probabilities using information from published, peer-reviewed journal articles, as well as published conference proceedings, reports, theses and dissertations, and book chapters. We limited our search to relevant literature that came from studies of Southwest ecosystems, with a geographical emphasis on Arizona, New Mexico, and northern

Mexico to ensure compatibility and relevance to Southwest ecosystems. This information is synthesized in narrative form for each PNVT in a companion document entitled "Historic Range of Variation for Potential Natural Vegetation Types of the Southwest" (Schussman and Smith 2006).

We described each model state by 1) its dominant vegetation and/or life form, 2) percent canopy cover or density of one vegetation component (ie grass, shrubs or trees), and 3) the number of years that can be spent in that state (without a disturbance) before it transitions to another state. Dominant vegetation and life form definitions followed the USFS's guidelines which break down or identify dominance types in terms of a single dominant species or genera when either accounts for ≥ 60% canopy cover, or in terms of co-dominant species or genera when 2 or more species or genera account for ≥ 80% canopy cover together with each individually having ≥ 20% canopy cover. Life forms are classified as tree if tree canopy cover is ≥ 10%, shrub if shrub canopy cover is ≥ 10%, and herbaceous if herbaceous canopy cover is ≥ 10% herbaceous canopy cover (Brohman and Bryant 2005). We utilized USFS guidelines in the model building process in order to make the models directly comparable to Region 3's mid-scale mapping of current vegetation. Parity of this nature will allow modeled estimates of historic vegetation to be compared with current vegetation in order to determine departure from historic and too help identify desired future conditions.

We identified nineteen types of transitions that are likely under historical (pre-1880) and/or current (post-1880) conditions: stand replacing fire, mixed severity fire, surface fire, in-growth, drought event, wet event, large droughts followed immediately by erosion events such as large wet events or wind events (Drought/Wet/Wind), windthrow, avalanche, insect outbreak, disease outbreak, herbivory (native and non-native), use by Native people, plant growth, pre-scribed fire or wildland fire use, spread of non-native species, and mechanical or chemical treatments. This is not an exhaustive list of possible transitions but rather represents a list for which there was information available to determine the effect and/or frequency of the transition.

The level of model complexity (number of model states and transitions) varies by PNVT based on the amount of available information. For example, there is a great deal of disturbance, cover, and post-disturbance regeneration information available for the ponderosa pine PNVT, hence a 10 state model with 5 transitions was created. In contrast, there is little to nothing known about these same factors for the Madrean encinal PNVT, hence no model was not created.

Vegetation Dynamics Development Tool - We used VDDT software to model historic and current proportions of the landscape in all model states. We included transitions in the models only if 1) there was documentation that consistently identified the frequency and effect of that transition on vegetation composition and structure; and 2) if that transition was applicable to a majority of the vegetation within the regional PNVT being modeled. For example, we know that mechanical and chemical treatments of interior chaparral occurred at varying frequencies and intensities throughout small portions of Arizona's interior chaparral between 1950 and 1980, however, these treatments were variable across the landscape and applicable to only a small portion of interior chaparral vegetation in Arizona and New Mexico. Given the variability in treatments and the low applicability of these transitions to the regional description of the PNVT, these transitions were not modeled. However, if some or all of these treatments are being considered for future management they can easily be incorporated into the model at a later date.

Model Parameters – Vegetation Dynamics Development Tool models are non-spatial models with between 0 and 50,000 sample units (pixels) for all states that can be simulated over 1 to 1000 year time horizons. Sample units are assigned to a state at the start of the model and change from one state to another based on the probability of transition occurrence. The proportion of the modeled landscape (number of pixels) in any given state is identified for all years modeled.

In order to minimize the variability in model output that arises from variation in sample size (i.e., the number of pixels modeled) and to standardize models for all PNVTs, we conducted a sensitivity analysis of a "simple" grassland model to determine the appropriate number of sampling units (pixels) and model runs (simulations) to use in scenario analysis. The "simple" grassland model is a 4 box model that includes 3 transitions (fire, drought, and plant growth) (Figure 13-1). Results of the sensitivity analysis showed that variation due to sample size was minimized when 1,000 or more sample units were used (Table 13-1). Based on this result we set the modeled landscape at 1000 pixels and ran each scenario for a total of 10 runs (simulations) in order to calculate a mean and standard deviation value for each modeled state. This analysis also highlighted the need to perform a sensitivity test on the range of values identified for the probability of a transition in each model, as seemingly small differences in the probability of a transition had large impacts on model output when the transitions are very **frequent** yet had little impact on model output when transitions are very **infrequent** (Tables 13-2 and 13-3). Given these results and the fact that information from different studies of the same PNVT yielded a range of values for the frequency of transitions, we decided to use sensitivity analysis to determine the impact of imprecise information on all models for which a range of values was identified in the literature. Specifically, when a range of values was given for a transition, we ran the model using the average value, as well as the high and low ends of the value range and reported the results from all three model runs.

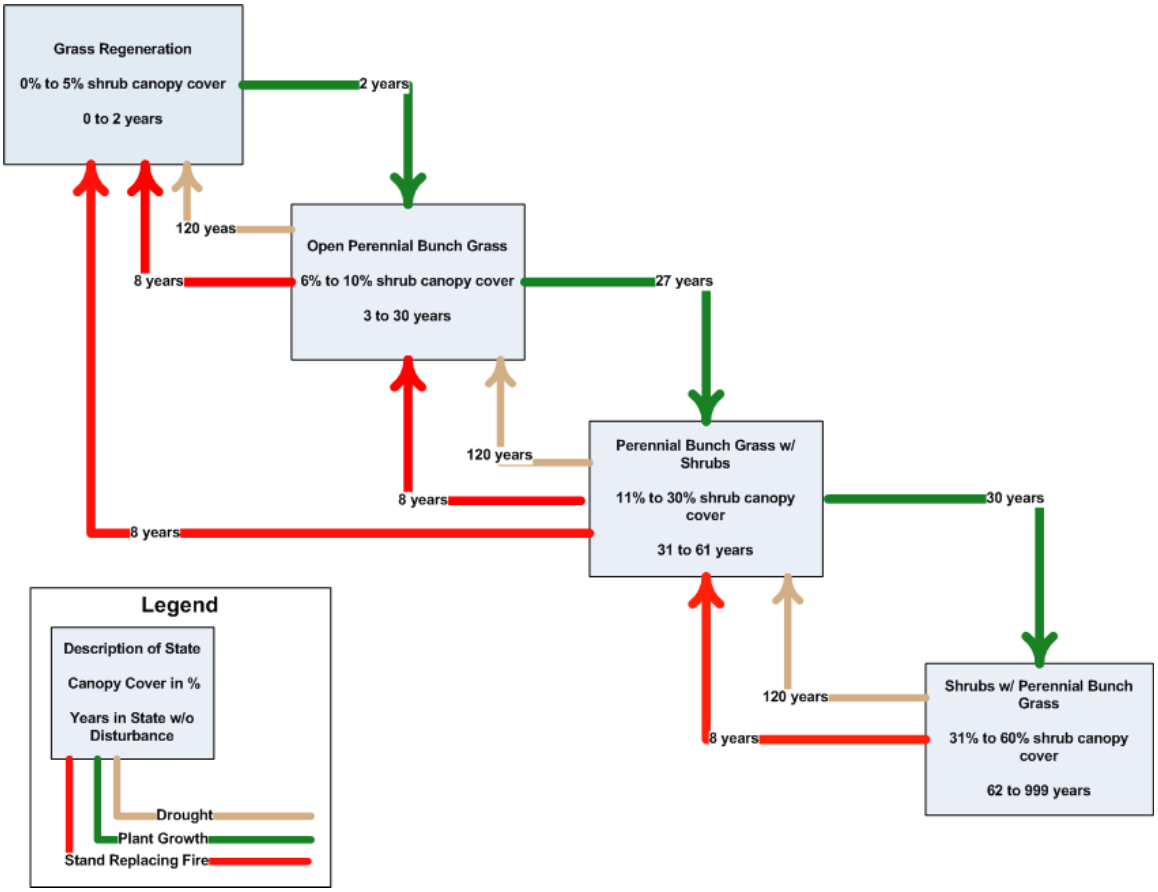

Figure 13-1. Simple grassland model used in sensitivity testing of VDDT software

Table 13-1. Sensitivity analysis showing the stabilization of model output, as indicated by average percent of the modeled landscape in each vegetation state and average standard deviation, when model is run at or above 1,000 sample units.

Sample Number	State A (%)	Standard Deviation (%)	State B (%)	Standard Deviation (%)	State C (%)	Standard Deviation (%)	State D (%)	Standard Deviation (%)
10	14.0	10.6	54.2	16.1	17.8	11.0	14.0	11.8
100	15.1	3.8	56.6	5.3	17.2	3.3	13.1	3.0
1000	13.5	1.0	57.4	1.4	16.5	1.0	12.5	1.1
10000	13.7	0.4	57.3	0.6	16.4	0.4	12.6	0.4

Table 13-2. Sensitivity analysis showing dramatic changes in the average percent of the landscape in each state when the frequency of the fire transition (every 8 years) is multiplied by a range of values between 0 and 2. Increasing the frequency of fire by a factor of 2 drastically changed the average percent of states A, C, and D. Similarly, decreasing the frequency by roughly a half (Every 20 years) also drastically changed the average percent of most of the states.

Fire Frequency Multiplier	Fire Frequency	State A (%)	State B (%)	State C (%)	State D (%)
0.0	none	0.0	0.0	0.0	100
0.4	Every 20 years	1.1	18.1	22.2	58.6
0.8	Every 10 years	8.6	48.5	20.1	22.8
1.0	Every 8 years	13.7	57.6	16.2	12.5
1.2	Every 7 years	15.7	66.3	11.8	6.2
1.6	Every 5 years	26.9	66.0	5.2	1.9
2.0	Every 4 years	31.5	65.9	1.9	0.0

Table 13-3. Sensitivity analysis showing little change in the average percent of the landscape in each state when the frequency of the drought transition (every 120 years) is multiplied by 0, 1, and 2. Increasing the frequency of drought by a factor of 2 increased the average percent of state A by only 5%, while state B saw a change of 6%. Decreasing the probability to 0 decreased A by about 4% and B by 2.5%, increased D by 5% and had little effect on state C.

Drought Frequency Multiplier	Drought Frequency	State A (%)	State B (%)	State C (%)	State D (%)
0.0	None	16.3	56.4	14.5	12.8
1.0	Every 120 years	20.4	59.0	13.2	7.4
2.0	Every 60 years	15.9	65.3	13.0	5.8

We ran the historic models for 1000 years, as this temporal span corresponds with the widest frame of reference offered by the scientific literature. Additionally, 1000 year long runs allowed for infrequent transitions, such as stand replacing fires in the spruce fir PNVT and extreme drought events in all PNVTs, to occur several times within each simulation. Ultimately, this level of temporal depth makes for a robust historic model that allows for multiple replicates of infrequent events while not over reaching the bounds of our historic knowledge. Current models were run for 120 years as this corresponds to the post-European settlement era when large scale changes to historic fire, flooding and grazing regimes in the Southwest were first documented.

We began all historic model runs with equal proportions of the modeled landscape in each state. For example if the model had 4 states then the historic model would start the 1000-year simulation with each state making up 25% of the landscape. However, for the current models, we began the 120-year simulations with the proportions of each state equal to the output values (900-year averages) from the historic model runs. This allowed us to simulate how the last 120 years of management has changed the historic proportions of the vegetative states.

Variability - One of the main concerns with vegetation models is the use of mean values to model the frequency of events that are variable in space and time. This is a valid concern and criticism as the mean value is not a metric for describing variability. For example, in the Madrean pine oak woodland, mean fire return interval (MFRI) for all fires, at 15 sites located in Arizona and northern Mexico, ranged between 3 and 7 years, while the MFRI for fires that scarred 25% of the trees ranged between 5 and 13.2 years (Fulé and Covington 1998; Fulé and others 2005; Kaib and other 1996; Swetnam and Baisan 1996; Swetnam and others 1992). Additionally, the minimum and maximum number of years between any given fire was between 1 and 38 years (Fulé and others 2005; Kaib and other 1996; Swetnam and Baisan 1996; Swetnam and others 1992).

Given concern over the use of mean values and the variability in the frequency of Southwest transitions we investigated the ability of VDDT to model variability in vegetation dynamics. Specifically, we analyzed year to year variability in our simple grassland model. Results of this analysis showed there to be little variability from year 10 to 1000 (13- 2). This was due to the consistency with which the probability of the transitions occurred (i.e., every year, each sample unit in which fire could occur had a probability of 0.12 of having that fire) as well as the large number of sampling units.

Climatic factors are known to be important drivers for many of the transitions we modeled, such as fire occurrence and insect outbreaks. Given this connection, we investigated the incorporation of climate variation on these transitions within the models. This was accomplished through the use of VDDT's "annual multiplier" function. This function allows the user to identify the frequency of year types that are known to increase or decrease the frequency of a transition, and then apply a multiplier value to the mean probability based on the occurrence of the year types. As year types vary, so too does the probability of a transition occurring. The result of the inclusion of hypothetical multipliers into the simple grassland model was year to year variability in the probability of a transition resulting in year to year variability in the proportion of the landscape in any given state (Figure 13-2 and Table 13-4). The inclusion of annual variability into the models allowed us to estimate not only the mean proportion of the landscape in a given state, but also the minimum, maximum, and standard deviation values for a state.

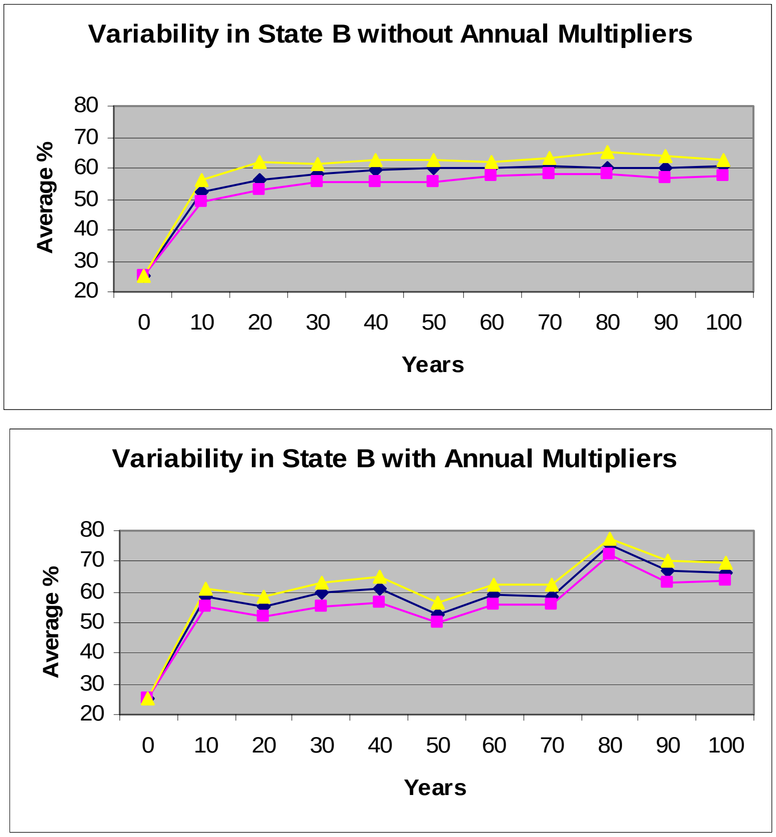

Figure 13-2. Comparison of year to year variability in state B of the simple grassland VDDT model with and without the use of annual multipliers. Maximum values in yellow, average values in blue, and minimum values in pink.

Table 13-4. Sensitivity analysis showing differences in annual variability with and without the use of the annual multiplier function.

Model State	Average Percent (No Multiplier)	Average Standard Deviation	Average Percent (Multiplier)	Average Standard Deviation
A	15.5	1	13.5	9.6
B	59.8	3.6	57.6	11.5
C	14.6	1.1	16.8	6.1
D	10.1	1.8	14.4	5.9

Fire Variability – The connection between fire occurrence and climate in the Southwest has been well established (Crimmins and Comrie 2004; Swetnam and Betancourt 1998). Based on this knowledge and our understanding of modeling year to year variability with VDDT, we modeled climate-mediated fire transitions using the annual multiplier function. To run the annual multiplier function we needed to identify the frequency of year types that increased and/or decreased fire occurrence as well as identify the magnitude of the effect. We obtained this information by analyzing the percent of regional fires that occurred in each year type using contingency table analysis (for an example see (Table 13-5). The regional fires were identified by Swetnam and Betancourt (1998) on the basis of having been recorded at two thirds of all sites, 41 of 63 sites, with fire history reconstructions in the Southwest; these fires occurred between1709 and 1879. The year types (severe drought, drought, normal, wet, and extremely wet) were identified from an in-depth analysis of Ni and others' (2002) 989-year winter precipitation reconstruction. Details of this analysis are described in a companion document entitled "Assessing Low, Moderate, and High Severity Drought and Wet Events Across the Southwestern United States from Year 1000 to 1988" (Schussman 2006).

Table 13-5. Example of contingency table analysis used to identify the magnitude of connection between regional fires and year type with a significant ($p < 0.001$) difference.

Year Types	Regional Fire No % of years (total count)	Regional Fire Yes % of years (total count)
Severe Drought	74.8 (238)	25.2 (80)
Drought	81.4 (131)	18.6 (30)
Normal	89.2 (538)	10.8 (65)
Wet	96.6 (113)	3.4 (4)
Extremely Wet	99.7 (339)	0.3 (1)

We identified the frequency of year types by simply totaling the percent of years, out of 989, for each individual year type. Finally, we derived the annual multiplier from the contingency table analysis by dividing the frequency of fire occurrence in a given year type by the mean probability of fire occurrence within the model. For example, if the frequency of regional fire occurrence in the severe drought year type was 0.252 (or regional fires occurred 25.2% of the time in severe drought years) and the mean probability of fire occurrence in the model was 0.12, then we applied a multiplier of 2.1 to the fire transition for all severe drought years. This change increases fire probability from 0.12 to 0.252 in severe drought years but maintains the mean fire frequency across all year types.

Finally, in order to make this information specific to a PNVT model, we selected data for inclusion in each PNVT fire/climate analysis based on the geographical overlap of winter precipitation climate data, which are identified for the 15 climate divisions within Arizona and New Mexico, with a PNVT boundary.

*Model Reporting –*We developed a descriptive state and transition diagram for historic and current conditions as well as a current photographic diagram for each PNVT. For all historic transitions, the historic frequency, or range of frequencies, of each transition is identified. Additionally, all possible transitions for which there was some level of information are included in the state and transition model. However, only those transitions for which the transition impacted the majority of the vegetation within a PNVT and for which information regarding the frequency and effect of the transition on the vegetation was consistently identified were included into the quantitative VDDT models. Identification of the frequency of transitions, source(s) used to identify transitions, and assumptions made in identifying the frequency or effect of transitions are detailed in tabular form for both historic and current models, for each PNVT separately in the following chapters.

For the historic models, we report the 900-year average, minimum, maximum, and average standard deviation for each state. We report results from the last 900 of the 1000 years because it takes the model 50-100 years to come to equilibrium from initial conditions. For the current models, we report the average, minimum, maximum, and standard deviation of the final year of the 120-year model run. The summary statistics were calculated based on 10 model runs (simulations) for both the historic and current models.

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Chapter 14 - Semi-Desert Grassland Model

14.1 Mixed Native Vegetation Dynamics

Mixed native grasslands are the dominant grassland type within the uplands of Arizona and have been shown to trend from open grasslands with low shrub canopy cover (less than 10% = state) towards higher shrub cover and ultimately to convert (> 35% total canopy cover and > 15% mesquite or juniper cover) to shrublands without frequent fire (Gori and Enquist 2003). While we know frequent fires, on the order of every 2.5 to 10 years, to have historically maintained these grasslands in an open, shrub-free state, it is unclear exactly how many missed fire cycles will generate shrub conversion or how drought and livestock grazing interact and affect the rate of shrub increase (Brown and others 1997; Cable 1971; McPherson 1995; Robinett 1994; Thornber 1907 in Humphrey 1949; Wright 1980). Wet winters have been correlated with increases in woody species density and cover; hence prolonged wet periods also act to increase shrub density and cover of the dominant shrub species (mesquite, juniper, creosote, and burroweed) (Barton and others 2001; Grissino-Mayer and Swetnam 2000; Miller and Rose 1999; Savage 1991; Swetnam and Betancourt 1998). Shrubland conversion occurs when total shrub canopy cover gets above 35% (or mesquite or juniper cover >15%) and results in the loss of perennial grasses which increases the amount of bareground exposed to wind and water (Gori and Enquist 2003; Whitford 2002). Increases in soil exposure can result in losses of topsoil and argillic horizons, ultimately making it difficult for grasses to re-colonize a site even if shrub cover is decreased. However, the amount of erosional loss varies by soil type and location and, while loss of the argillic horizon transforms some areas into shrublands, areas where erosion is less of a factor (ie cobble protected uplands) and water infiltration occurs at sufficient depths to promote shrub growth, fire is key for maintaining these low shrub grasslands (McAuliffe 1995).

Graphical and photographic depictions of these vegetation dynamics are displayed in Figures 14-1, 14-2 and 14-3; results of the quantitative VDDT models are shown in Tables 14-1 and 14-2.

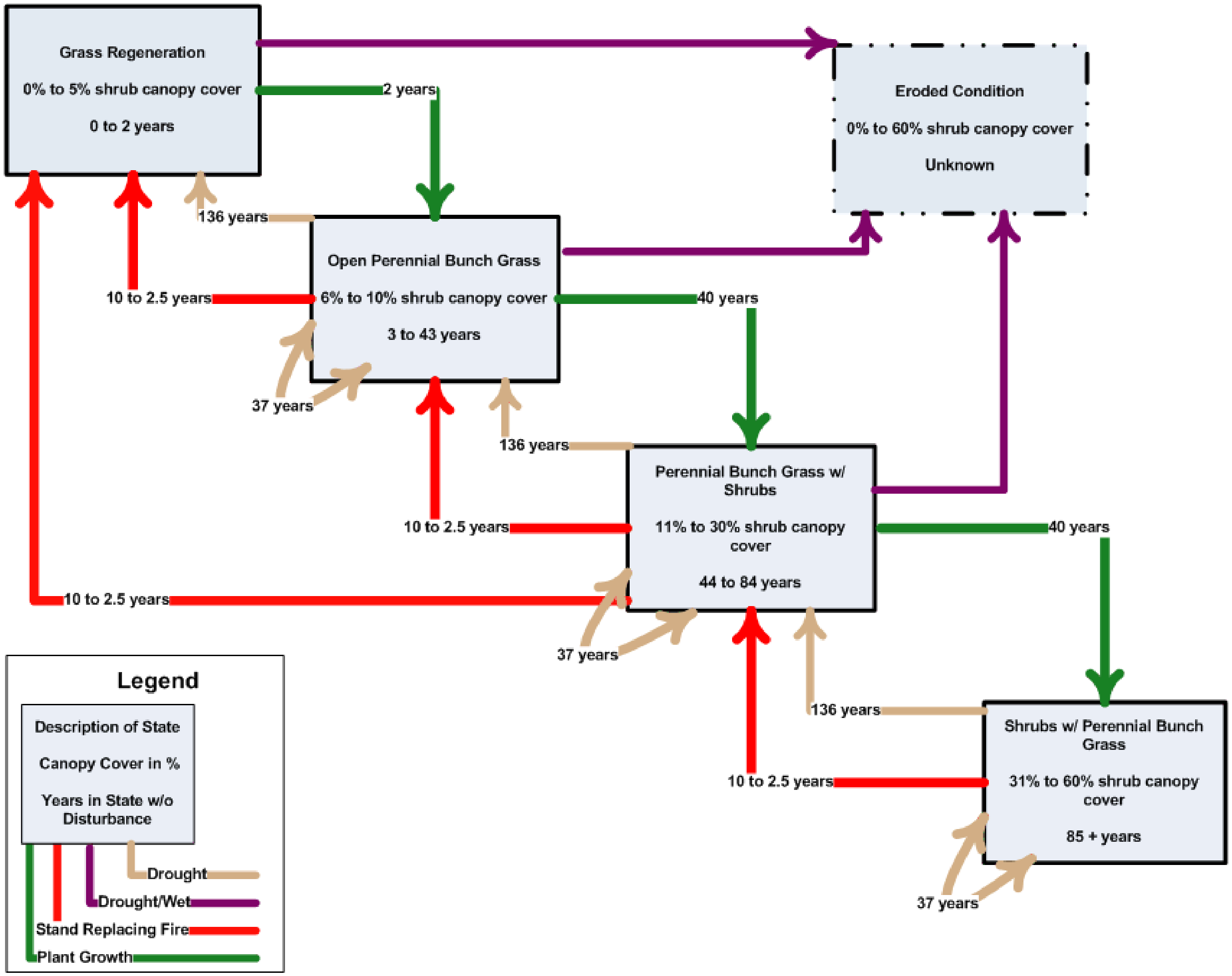

Figure 14-1. Conceptual historic state and transition model for the semi-desert grassland mixed native vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no information exists on the frequency of transitions the arrow is blank. Dashed outlines represent states which have crossed an ecological threshold.

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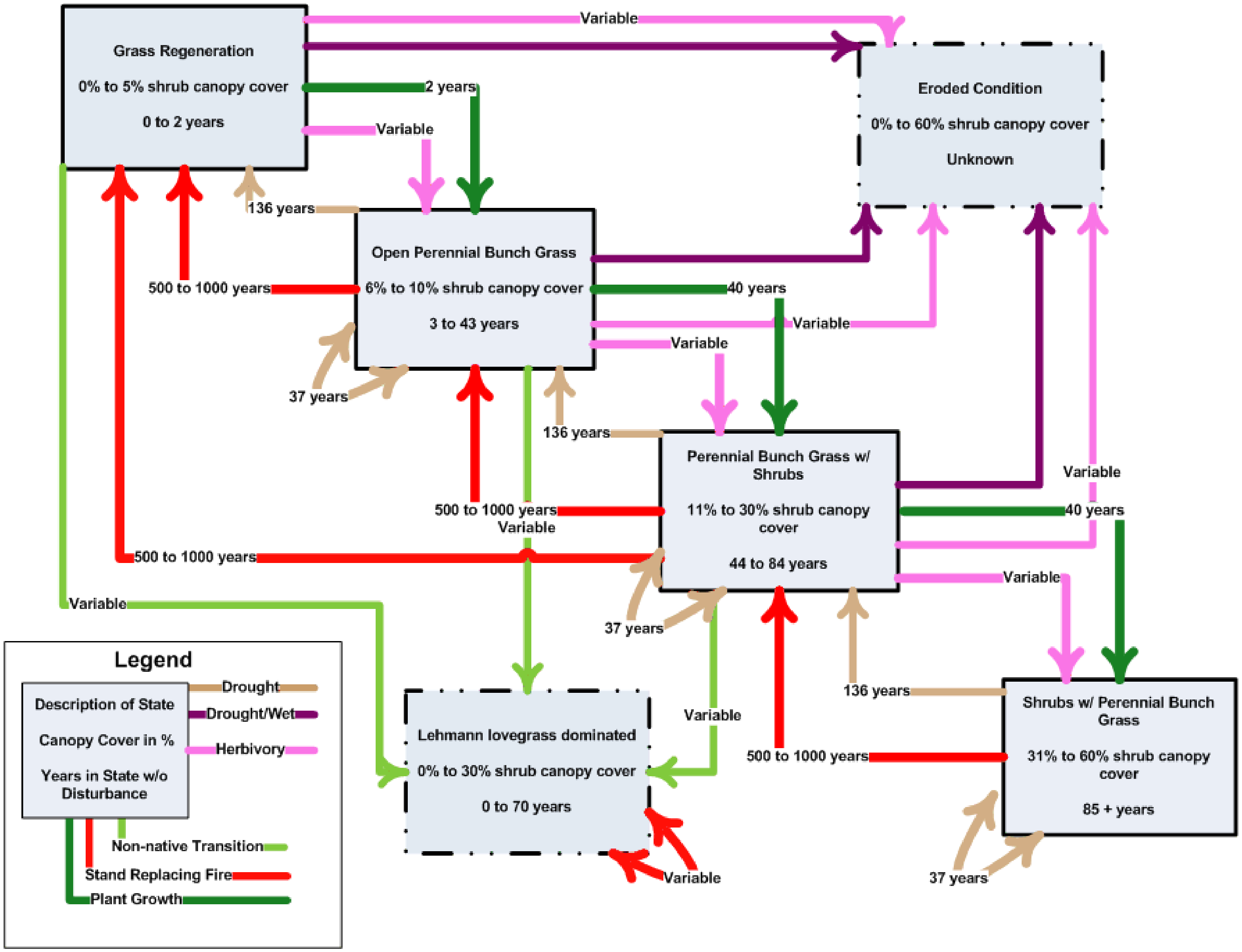

Figure 14-2. Conceptual current state and transition model for the semi-desert grassland mixed native vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, a blank arrow or variable, respectively, is the notation. Dashed outlines represent states which have crossed an ecological threshold.

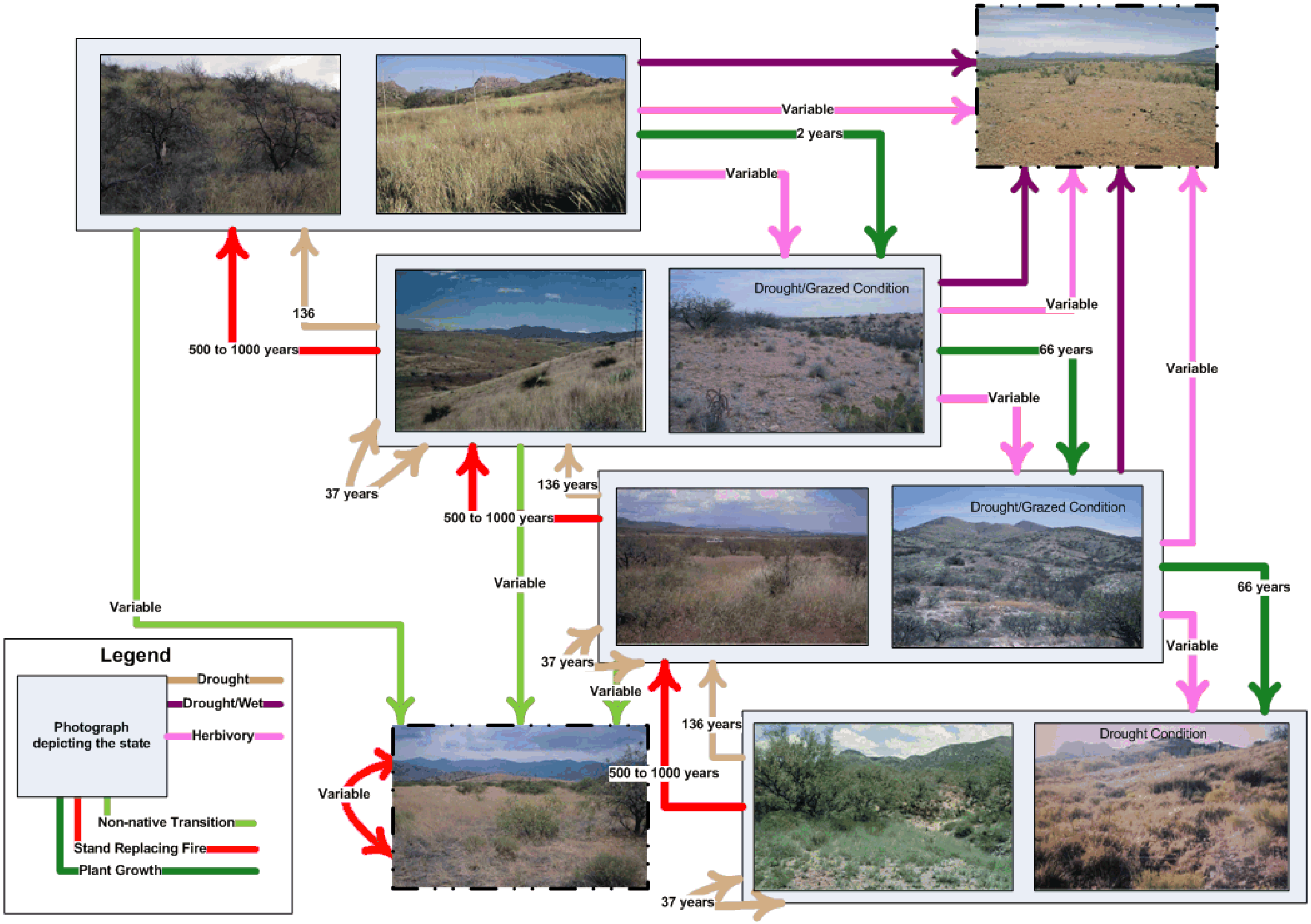

Figure 14-3. Photographic depiction of current conceptual state and transition model for the semi-desert grassland mixed native vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, a blank arrow or variable, respectively, is the notation. Dashed outlines represent states which have crossed an ecological threshold.

Model Parameters

In Tables 6 and 7 below, we describe the parameters included and not included within the historic and current VDDT models, as well as the sources of information and any assumptions used to create model parameters. Parameter information was drawn from studies conducted within the semi-desert grassland mixed native vegetation type unless other wise noted.

Table 14-1. Identification of historic transitions, frequency of transitions, sources of information used, and assumptions used to develop effect and frequency of transitions included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Drought (Moderate Events)	Every 37 years	We calculated frequency of moderate drought events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified droughts effect on shrubs from the following publications: Bock and Bock1997; McClaran 2003; Turner and others 2003.	Prolonged drought has been shown to cause declines in shrub density and cover within these grasslands. Based on this, we used moderate drought events (equivalent to 1950's drought) in the model to transition vegetation back to its starting conditions within that state (i.e. lowest shrub cover value) using the average historic moderate drought frequency.
Drought (Extreme Events)	Every 136 years	We calculated frequency of extreme drought events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified droughts effect on shrubs from the following publications: Bock and Bock 1997; McClaran 2003; Turner and others 2003.	Prolonged drought has been shown to cause declines in shrub density and cover within these grasslands. Based on this, and the above use of moderate droughts, we used extreme drought events (more severe than 1950's drought) to transition higher shrub density states to lower shrub density states using the average historic extreme drought frequency.
Drought/ Wet	Blank, Not included in model	Hennessey and others 1983 In order to utilize information available on erosion impacts, we included a study from New Mexico. This may result in erroneous information for the mixed native grasslands of the Arizona uplands.	We determined that some level (unknown) of drought coupled with some level (probably at least equivalent to the 1950's drought, Hennessey and others, 1983) of wet event could create conditions that would lead to a loss of topsoil and an eroded condition. Additionally, transitions out of the eroded condition are not known. Given this lack of information we did not model this transition.
Stand Replacing Fire (SRF)	Every 2.5 to 10 years	We identified mean Fire Return Interval (FRI) from the following publication: Bahre 1985; Kaib and others 1996; McPherson 1995.	Based on direct (fire scar data) and indirect lines (fire ecology of grassland species) of evidence, a mean FRI of 2.5 to 10 years, is cited in the literature for these grasslands. Given this range of values, we used the high, middle, and low (10, 6, 2.5) ends of the FRI range in the model.
Plant Growth Following SRF	2 years	We identified grass recovery time from the following publications: Bock and Bock 1992; Cable 1972; Martin 1983; Wright 1980.	Studies on grass recovery following fires suggests that perennial grasses recover fully from fire in 1 to 2 growing seasons with average precipitation, but can take 3 to 4 growing seasons to recover under drought conditions. Based on this

Transition Type	Transition Frequency or Length	Sources	Assumptions
			information, we used a mean value of 2 years to model plant growth immediately following fire.
Plant Growth Without SRF	40 years	We calculated frequency of wet events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified effect of wet events on shrubs from the following publications: Bock and Bock 1997; Brown and others 1997; McClaran 2003.	Based on the average frequency of low, moderate, and extreme wet winter precipitation events (every 20 years) and the time (about 20 years) it takes for shrubs to show large cover increases (3 fold) following these wet periods, we used a value of 40 years to model the plant growth transition in the absence of SRF.

Table 14-2. Identification of current transitions, frequency of transitions, sources of information used, and assumptions used to develop effect and frequency of transitions included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Drought (Moderate Events)	Every 37 years	We calculated frequency of moderate drought events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified droughts effect on shrubs from the following publications: Bock and Bock 1997; McClaran 2003; Turner and others 2003.	Prolonged drought has been shown to cause declines in shrub density and cover within these grasslands. Based on this, we used moderate drought events (equivalent to 1950’s drought) in the model to transition vegetation back to its starting conditions within that state (i.e. lowest shrub cover value) using the average historic moderate drought frequency.
Drought (Extreme Events)	Every 136 years	We calculated frequency of extreme drought events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified droughts effect on shrubs from the following publications: Bock and Bock1997; McClaran 2003; Turner and others 2003.	Prolonged drought has been shown to cause declines in shrub density and cover within these grasslands. Based on this, and the above use of moderate droughts, we used extreme drought events (more severe than 1950’s drought) to transition higher shrub density states to lower shrub density states using the average historic extreme drought frequency.
Drought/ Wet	Blank, Not included in model	Hennessey and others 1983 In order to utilize information available on erosion impacts, we included a study from New Mexico. This may result in erroneous information for the mixed native grasslands of the Arizona uplands.	We determined that some level (unknown but probably at least equivalent to the 1950’s drought, Hennessey and others, 1983) of drought coupled with some level of wet event could create conditions that would lead to a loss of topsoil and an eroded condition. Additionally, transitions out of the eroded condition are not known. Given this lack of information we did not

Transition Type	**Transition Frequency or Length**	**Sources**	**Assumptions**
			model this transition.
Herbivory (non-native)	Variable, not included in model	We identified possible effects of grazing on shrubs from the following publications: Bock and Bock 1997; Brown and Archer 1987; Brown and others 1997; Drewa and Havstad 2001; McClaran 2003; McPherson 1997; Smith and Schmutz 1975; Valone and Kelt 1999) In order to utilize the breadth of studies on grazing effects we included studies from Arizona, New Mexico, and Texas. This may result in erroneous information for the mixed native grasslands of the Arizona uplands.	We determined that information regarding effects of livestock grazing on shrub cover is conflicting. For example, Valone and Kelt (1999) and Brown and Archer (1987) found *Gutierrezia sarothrae* and mesquite to be more common on grazed plots. Drewa and Havstad (2001), Smith and Schmutz (1975) and McClaran (2003) found grazing to have no effect on shrub density, while Bock and Bock (1997) identified a negative effect of grazing on *Isocoma sp*. density. Given the conflicting results of studies we did not include this transition in our model.
Non-native	Variable, not included in model	We identified information on *Eragrostis lehmanniana* spread from the following publications: Anable and others1992; Angel and McClaran 2001; Cable 1971; Cox and Ruyle 1986; Gori and Enquist 2003; Ruyle and Cox, 1988; Schussman and others in press.	Studies indicate that the transition of a state from native to non-native grassland is dependent on the presence of a non-native seed source and soil type. Given that we were trying to model a large area that has multiple soil types and is likely to only have a seed source on a portion of the landscape, we decided not to model this parameter for the regional model.
Stand Replacing Fire (SRF), Native	Every None to 500 years	Kaib and others 1996; Swetnam and Betancourt 1998	We based our estimate of fire on fire scar data. Specifically, regional fire scar data shows drastic declines in the number of fires from 1880 to present. Additionally, a fire scar study that is directly applicable to these grasslands (Kaib and others 1996) shows decreases to no fires and fire every 25 years. Based on the knowledge that some areas had not seen a fire at all, and others had some fire, we used a range of relatively infrequent fire occurrence (no fire to fire across the whole landscape every 500 years).
Stand Replacing Fire, Exotic	Variable, not included in model	We identified information on the effects of burning on *Eragrostis lehmanniana* spread from the following publications: Anable and others 1992; Cable 1971; Erika Geiger, personal communication; McPherson 1995; Ruyle and Cox 1988.	It is well documented that stand replacing fires increase the abundance of *E. lehmanniana* on sites where it already exists, the frequency of fires at sites dominated by *E. lehmanniana* is site specific. Given this constraint, and the regional, not site specific, nature of the model we decided not to model this transition.
Plant	2 years	We identified grass recovery time from the	Studies on grass recovery following fires suggests that

Transition Type	Transition Frequency or Length	Sources	Assumptions
Growth Following SRF		following publications: Bock and Bock 1992; Cable 1972; Martin 1983; Wright 1980.	perennial grasses recover fully from fire in 1 to 2 growing seasons with average precipitation, but can take 3 to 4 growing seasons to recover under drought conditions. Based on this information, we used a mean value of 2 years to model plant growth immediately following fire.
Plant Growth Without SRF	40 years	We calculated frequency of wet events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified effect of wet events on shrubs from the following publications: Bock and Bock; Brown and others 1997; McClaran 2003.	Based on the average frequency of low, moderate, and extreme wet winter precipitation events (every 20 years) and the time (about 20 years) it takes for shrubs to show large cover increases (3 fold) following these wet periods, we used a value of 40 years to model the plant growth transition in the absence of SRF.

Results

Results of the semi-desert grassland – mixed native historic VDDT model show a good deal of variability in the 900-year average for each state based on the fire return interval (Table 14-3). Even with this variability, the pattern was consistent between the three models with the bulk of historic vegetation occurring in the Open Grass state (82.4 %, 75.5 % and 56.2 % for fire return intervals of 10, 6, and 2.5 years, respectively) and very little of the historic vegetation occurring in the Shrub & Grass state (0.0 %, 0.0 %, and 0.0 % for fire return intervals 10, 6, and 2.5 years). A comparison of simulated historic conditions and current conditions shows a large decrease in the percent of the landscape in the Open Grass state (decrease of roughly 61%, 54%, and 35% for fire return intervals 10, 6, and 2.5, respectively) with a correspondingly large increase in the percent of the landscape in the Shrub & Grass state (roughly 30% to 41% for all fire return interval runs) (Table 14-4).

Table 14-3. Results of the semi-desert grassland - mixed native historic VDDT model, reported as the 900-year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (6 years), high (10 years), and low end (2.5 years), of the estimated fire return interval range.

Fire Return Interval Modeled	Model Output	Grass Regeneration	Open Grass	Grass & Shrub	Shrub & Grass
Every 10 years					
	Average	17.1	82.4	0.4	0.0
	Minimum	0.6	61.1	0.0	0.0
	Maximum	38.7	82.6	2.2	0.1
	Standard Deviation	7.2	7.2	0.3	0.0
Every 6 years					
	Average	24.5	75.5	0.0	0.0
	Minimum	0.5	45.0	0.0	0.0
	Maximum	55.0	71.5	0.4	0.0
	Standard Deviation	10.2	10.2	0.0	0.0
Every 2.5 years					
	Average	43.8	56.2	0.0	0.0
	Minimum	0.9	4.1	0.0	0.0
	Maximum	95.9	52.5	0.0	0.0
	Standard Deviation	19.5	19.5	0.0	0.0

Table 14-4. Results of the semi-desert grassland - mixed native current VDDT model, reported as the 120-year end value for average, minimum, maximum, and average standard deviation of the percent of the modeled landscape each state.

Fire Return Interval Modeled	Model Output	Grass Regeneration	Open Grass	Grass & Shrub	Shrub & Grass
No Fire					
	Average	0.1	13.7	45.4	40.8
	Minimum	0.0	11.9	41.5	36.3
	Maximum	0.2	16.6	47.1	44.5
	Standard Deviation	0.0	1.3	1.4	2.1
Every 1000 years					
	Average	0.3	29.0	40.3	30.4
	Minimum	0.1	27.8	37.4	28.9
	Maximum	0.6	32.2	42.8	31.8
	Standard Deviation	0.2	1.3	1.5	1.0
Every 500 years					
	Average	0.42	21.3	44.8	33.5
	Minimum	0.2	17.7	42.6	30.9
	Maximum	0.8	23.2	46.6	35.3
	Standard Deviation	0.16	1.7	1.3	1.3

Discussion

This analysis highlights the importance of frequent fire within the semi-desert grassland mixed native vegetation type. In the absence of frequent fire, the model simulates a landscape with increasingly less open grasslands dominated by perennial grasses. This result is in agreement with recent assessments of historic change within southeastern Arizona’s semi-desert grasslands that show an increase in shrubs and a loss of open grasslands (Turner and others 2003; Gori and Enquist 2003). This suggests the need to maintain historic fire regimes if we are to maintain open grassland vegetation.

14.2 Black Grama Vegetation Dynamics

Black grama dominated grasslands within New Mexico have been shown to trend towards shrublands over the last 100 years (Buffington and Herbel 1965; Gibbens and others 2005). It is unclear if the loss of grass and replacement by shrub species (primarily mesquite and creosote bush) is due to the absence or presence of fire or due to grazing and/or drought stress. In contrast to the mixed native type where shrub cover increases are primarily tied to a lack of fire events, shrub increase within black grama dominated grasslands have been seen following disturbances that have caused grass cover to drop, allowing shrub seedling establishment and soil erosion to occur (Whitford 2002). Disturbances such as drought, fire, and livestock grazing have all been shown to decrease black grama cover as well as cause mortality within this perennial grass (Buffington and Herbel 1965; Drewa and Havstad 2001; Gosz and Gosz 1996; Reynolds and Bohning 1956). The recent (last 120 years) spread of mesquite has been tied to increased seed dispersal by livestock as well as a sharp decline in mesquite use by Native Americans due to their declining population size (Frederickson and others 2006). As with vegetation dynamics within the mixed native type, areas converted to shrublands or dunelands are difficult to move back into grassland states as scattered nutrients and high erosion rates characteristic of the former reinforce a shrub/duneland system (Whitford 2002).

Graphical depictions of these vegetation dynamics are displayed in Figures 14-4, 14-5, and 14-6. Quantitative models were not created due to the lack of empirical data on which to determine the frequency of transitions, for a detailed discussion of transitions see Tables 14-5 and 14-6.

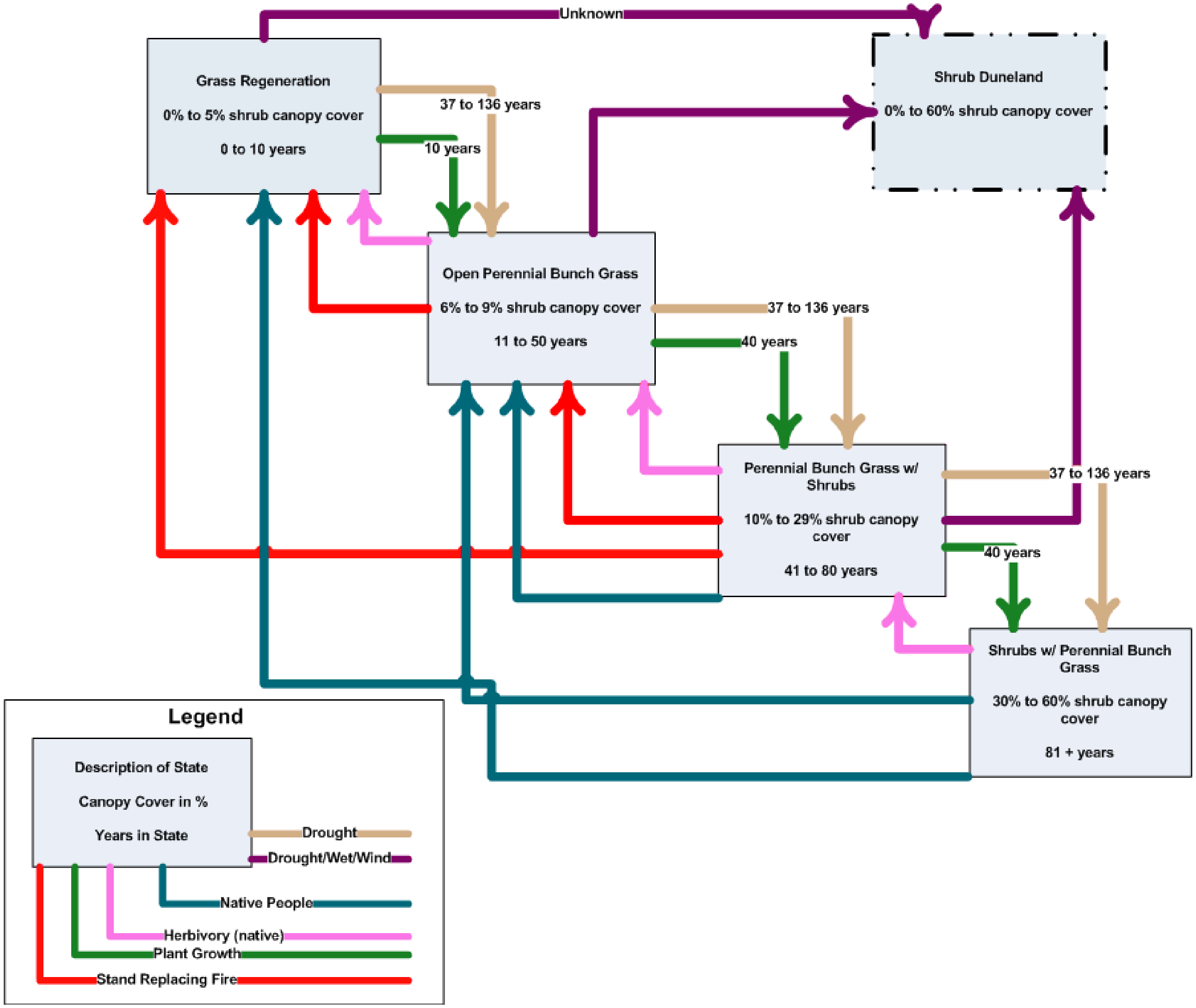

Figure 14-4. Conceptual historic state and transition model for the semi-desert grassland, black grama type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, a blank arrow or variable, respectively, is the notation. Dashed outlines represent states which have crossed an ecological threshold.

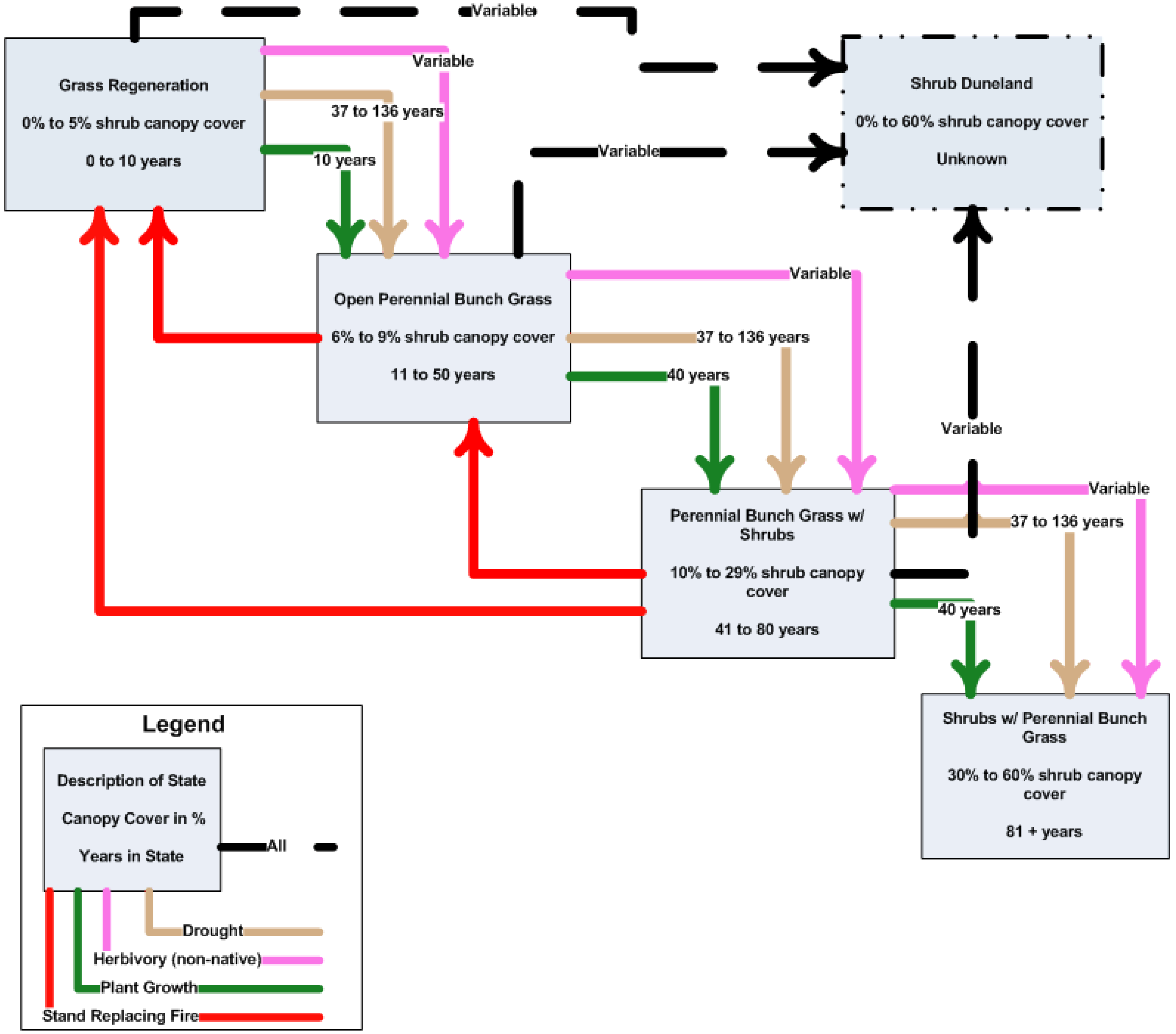

Figure 14-5. Conceptual current state and transition model for the semi-desert grassland, black grama vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, a blank arrow or variable, respectively, is the notation. Dashed outlines represent states which have crossed an ecological threshold.

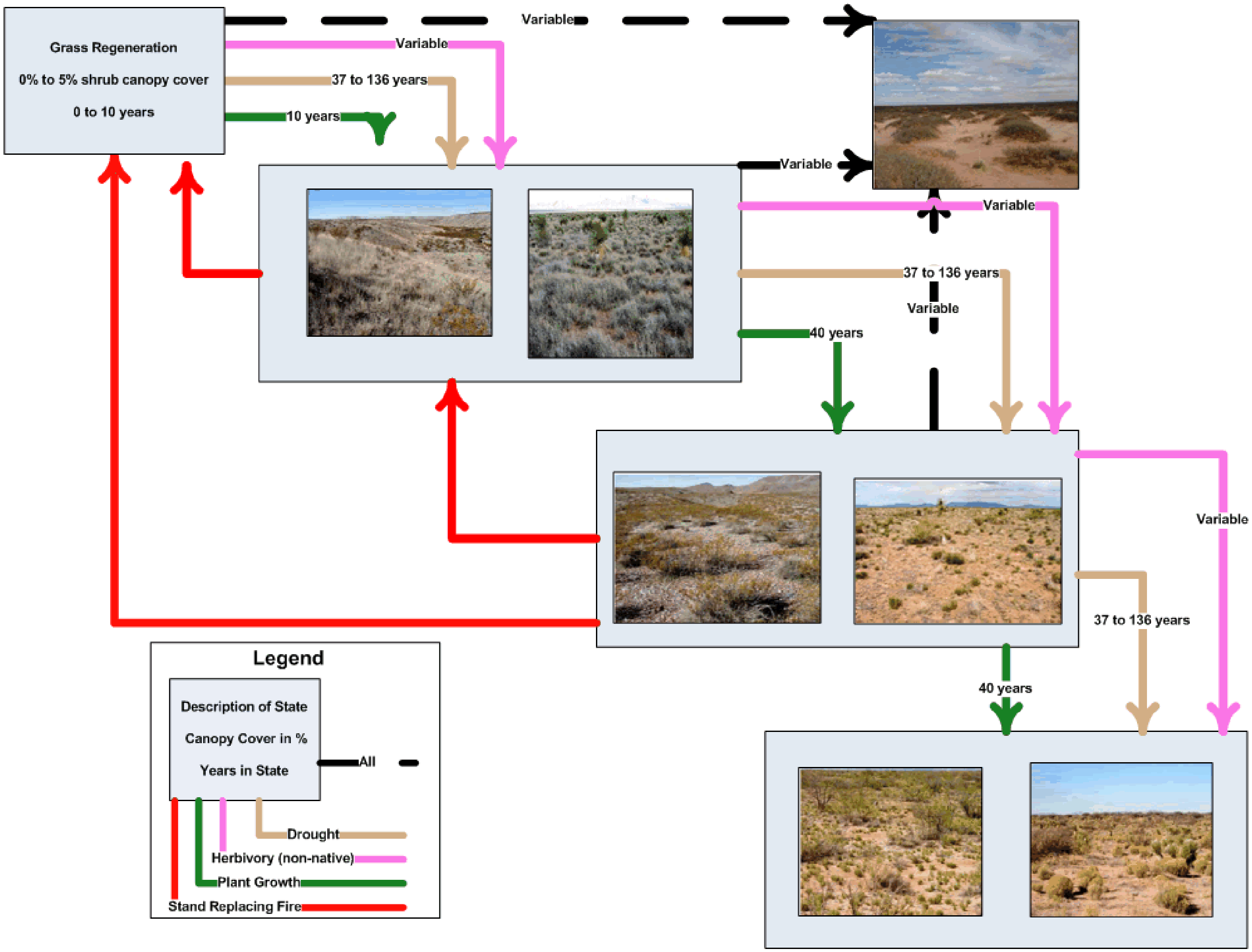

Figure 14-6. Photographic depiction of current conceptual state and transition model for the semi-desert grassland black grama vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, a blank arrow or variable, respectively, is the notation. Dashed outlines represent states which have crossed an ecological threshold. Photographs are from NRCS ecological site descriptions (http://www.nm.nrcs.usda.gov/technical/fotg/section-2/esd/sd2.html).

Model Parameters

There was not enough empirical information to build a quantitative VDDT model. A detailed description of transitions that did and did not have enough information, the sources of information and any assumptions used to try and develop a model parameter are shown in Tables 14-5 and 14-6. Parameter information was drawn from studies conducted within the semi-desert grassland black grama vegetation type unless other wise noted.

Table 14-5. Identification of historic transitions, frequency of transitions, sources of information used, and assumptions used to develop effect and frequency of transitions included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Drought (Moderate and Extreme Events)	Every 37 to 136 years	We calculated frequency of moderate and extreme drought events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified droughts effect on mesquite from Frederickson and others (2006).	Frederickson and others (2006) discuss the likelihood that prolonged drought increased mammalian dispersal of mesquite seeds that would have led to episodic expansion and increased densities of mesquite. Based on this we used the frequency of moderate to extreme drought events to determine the frequency of this transition.
Drought/ Wet/Wind	Blank	Devine and others 1998; Gibbens and others 1983; Hennessey and others 1983; Hupy 2004; Parsons and others 2003; Nash and others 2003; Neave and Abrahams 2001; Parsons and others 2003; Wainwright and others 2002	We determined that some level (unknown, but probably at least equivalent to the 1950's drought, Hennessey and others, 1983) of drought coupled with some level of wet event or wind erosion could create conditions that would lead to a loss of topsoil and an eroded shrub duneland. Transitions out of the eroded condition are not known.
Herbivory (Native)	Blank	Frederickson and others 2006; Weltzin and Archer 1997. In order to utilize the breadth of studies on the effects of small mammals, we included a study from Texas. This may result in erroneous information for the black grama grasslands of New Mexico.	Frederickson and others (2006) suggest that mammalian herbivory would have had an overall negative effect on mesquite spread and increase. Similarly, research by Weltzin and Archer also suggest that small mammals played a role in regulating mesquite. While the role for mammals in decreasing mesquite is established, the frequency or magnitude of an effect that they would have had on mesquite shrub encroachment is unknown.
Native People	Blank	Frederickson and others 2006	Frederickson and others (2006) suggest that Native Americans would have had a substantial impact on mesquite populations due to their use of seed pods as a food staple and wood for fuel and building. The magnitude of this effect is not quantified.
Stand Replacing Fire (SRF)	Blank	Branscomb 1956 in Buffington and Herbal 1965; Buffington and Herbal 1965; Drewa 2003; Drewa and Havstad 2001; Gosz and	There is no direct or indirect evidence that fires played a dominant role in these grasslands. There is evidence that fires have negative effects on black grama (Buffington and Herbel

Transition Type	Transition Frequency or Length	Sources	Assumptions
		Gosz 1996; Reynolds and Bohning 1956; Wright 1960	1965; Drewa and Havstad 2001; Gosz and Gosz 1996; Reynolds and Bohning 1956) during times of drought and that fire may have little negative effect on *Gutierrezia sarothrae* (Drewa and Havstad) or mesquite (Drew 2003).
Plant Growth Following SRF	10 years	Gosz and Gosz 1996; Reynolds and Bohning 1956	Black grama takes longer than other grasses to recover from fire events, with suggestions of multiple years of average to above average precipitation. Based on this we estimated a fire recovery time of 10 years.
Plant Growth Without SRF	40 years	We calculated frequency of wet events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified effect of wet events on shrubs from the following publications: Bock and Bock 1997; Brown and others 1997; McClaran 2003. In order to utilize the breadth of information on shrub response to winter precipitation all of these studies were conducted in the mixed native vegetation type of Arizona. This may result in erroneous information for the black grama grasslands of New Mexico.	Based on the average frequency of low, moderate, and extreme wet winter precipitation events (every 20 years) and the time (about 20 years) it takes for shrubs to show large cover increases (3 fold) following these wet periods, we used a value of 40 years to model the plant growth transition in the absence of SRF.

Table 14-6. Identification of current transitions, frequency of transitions, sources of information used, and assumptions used to develop effect and frequency of transitions included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
All	Variable	Buffington and Herbel 1965; Drewa and Havstad 2001; Gibbens and Beck 1988; Gibbens and others 2005; Gosz and Gosz 1996; Whitford and others 1999	Fire, livestock grazing, and drought have all been shown to negatively effect black grama and play a role in exposing the soil to wind and water erosion. The magnitude and frequency of these transitions to move black grama grasslands into a shrub duneland is not quantified.
Drought (Moderate and Extreme	Every 37 to 136 years	We calculated frequency of moderate and extreme drought events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified droughts effect on	Frederickson and others (2006) discuss the likelihood that prolonged drought increased mammalian dispersal of mesquite seeds that would have led to episodic expansion and increased densities of mesquite. Based on this we used the frequency of

Transition Type	Transition Frequency or Length	Sources	Assumptions
Events)		mesquite from Frederickson and others (2006).	moderate to extreme drought events to determine the frequency of this transition.
Drought/ Wet/Wind	Blank	Devine and others 1998; Gibbens and others 1983; Hennessey and others 1983; Hupy 2004; Parsons and others 2003; Nash and others 2003; Neave and Abrahams 2001; Parsons and others 2003; Wainwright and others 2002	We determined that some level (unknown, but probably at least equivalent to the 1950's drought, Hennessey and others, 1983) of drought coupled with some level of wet event or wind erosion could create conditions that would lead to a loss of topsoil and an eroded shrub duneland. Transitions out of the eroded condition are not known.
Herbivory (Non-Native)	Variable	Devine and others 1998; Gibbens and others 1983; Hupy 2004; Parsons and others 2003; Nash and others 2003; Neave and Abrahams 2001; Parsons and others 2003; Wainwright and others 2002	Erosion, and subsequent conversion from grassland to shrubland, increases with decreasing vegetative cover. Livestock grazing decreases cover, hence it is a disturbance that can cause a transition from grassland to shrubland. However, livestock grazing is variable across the landscape and so the frequency and magnitude of this transition is not quantified.
Stand Replacing Fire (SRF)	Blank	Branscomb 1956 in Buffington and Herbal 1965; Buffington and Herbal 1965; Drewa 2003; Drewa and Havstad 2001; Gosz and Gosz 1996; Reynolds and Bohning 1956; Wright 1960	Given the uncertainty surrounding the historic occurrence of fire, we left the fire transition as a possibility but did not identify a frequency.
Plant Growth Following SRF	10 years	Gosz and Gosz 1996; Reynolds and Bohning 1956	Black grama takes longer than other grasses to recover from fire events, with suggestions of multiple years of average to above average precipitation. Based on this we estimated a fire recovery time of 10 years.
Plant Growth Without SRF	40 years	We calculated frequency of wet events from 1000 years of reconstructed winter precipitation data (Ni and others 2001). We identified effect of wet events on shrubs from the following publications: Bock and Bock 1997; Brown and others 1997; McClaran 2003. In order to utilize the breadth of information on shrub response to winter precipitation all of these studies were conducted in the mixed native vegetation type of Arizona. This may result in erroneous information for the black grama grasslands of New Mexico.	Based on the average frequency of low, moderate, and extreme wet winter precipitation events (every 20 years) and the time (about 20 years) it takes for shrubs to show large cover increases (3 fold) following these wet periods, we used a value of 40 years to model the plant growth transition in the absence of SRF.

Valley Bottom Grassland Vegetation Dynamics

Some valley bottom, or basin floor, grasslands with deep argillic horizons, isolated within both states (San Rafael valley in Arizona and Animas valley in New Mexico), have not shown shrub or tree encroachment and/or conversion in the absence of fire or presence of livestock grazing (McAuliffe 1995; Muldavin and others 2002). These deep soil systems have maintained open grassland characteristics despite fire suppression, drought, and livestock grazing due to the maintenance of soils that prevent shrub and tree establishment (McAuliffe 1995). However, there are other valley bottom areas that once supported grasslands, such as the San Simon valley, that have been converted to shrublands due to soil erosion. It is unclear exactly what mechanisms are responsible for the resilience seen in some areas and not in others, however, higher average precipitation in the San Rafael and Animas valleys may be one factor. Ultimately, while these isolated valley bottom grasslands have unique features and vegetation dynamics, there is not enough empirical information available to develop a conceptual or quantitative model.

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Chapter 15 - Interior Chaparral Model

15.1 Interior Chaparral Vegetation Dynamics

Interior chaparral appears to be a fairly stable vegetation type due to the majority of its species having the ability to quickly re-sprouting following disturbance events, such as fire and mechanical or chemical removal (Cable 1975; Lillie and others 1964; Pase and Ingebo 1965; Pond and Cable 1960). Additionally, the few species that regenerate from seed require fire to prepare the seedbed (Carmichael and others 1978). Historically, this led to quick recovery of chaparral following the dominant natural disturbance, fire. In current times, these same mechanisms have allowed chaparral to maintain its dense canopy cover character regardless of human disturbance but have increased densities due to disturbance exclusion.

Graphical and photographic depictions of these vegetation dynamics are displayed in Figures 15-1, 15-2 and 15-3; results of the quantitative VDDT models are shown in Tables 15-1 and 15-2.

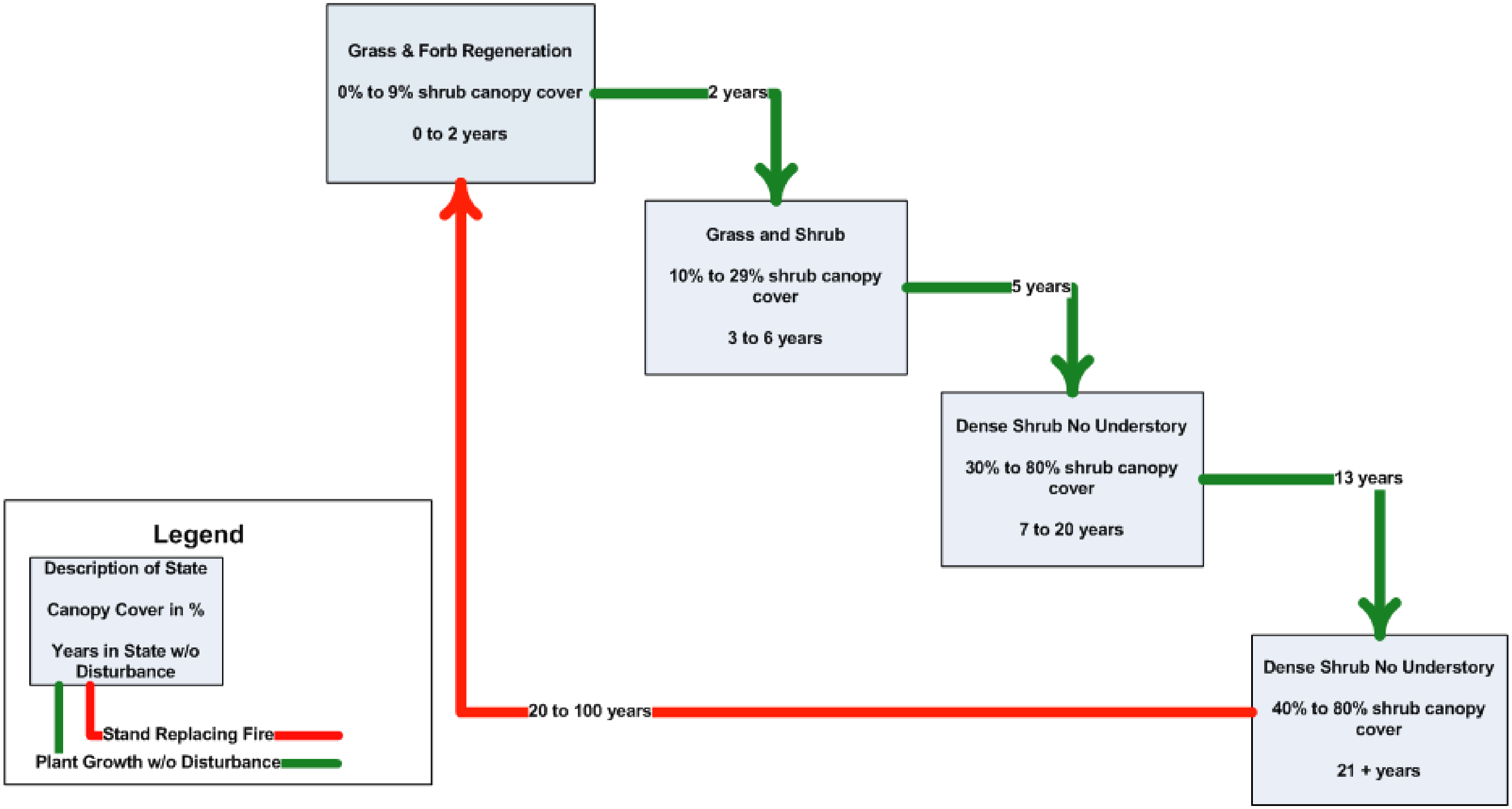

Figure 15-1. Conceptual historic state and transition model for the interior chaparral vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation.

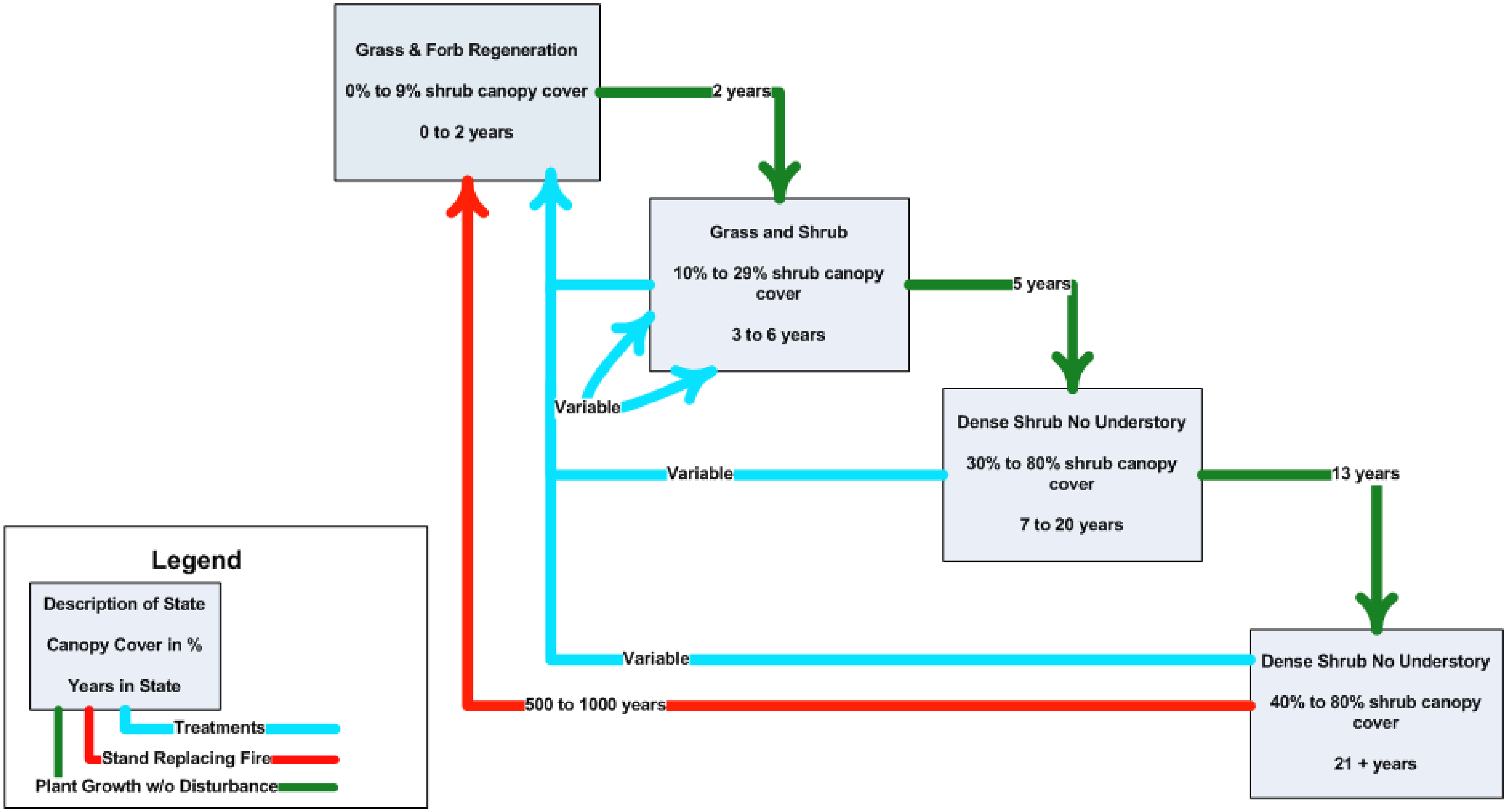

Figure 15-2. Conceptual current state and transition model for the interior chaparral vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown or variable, respectively, is the notation.

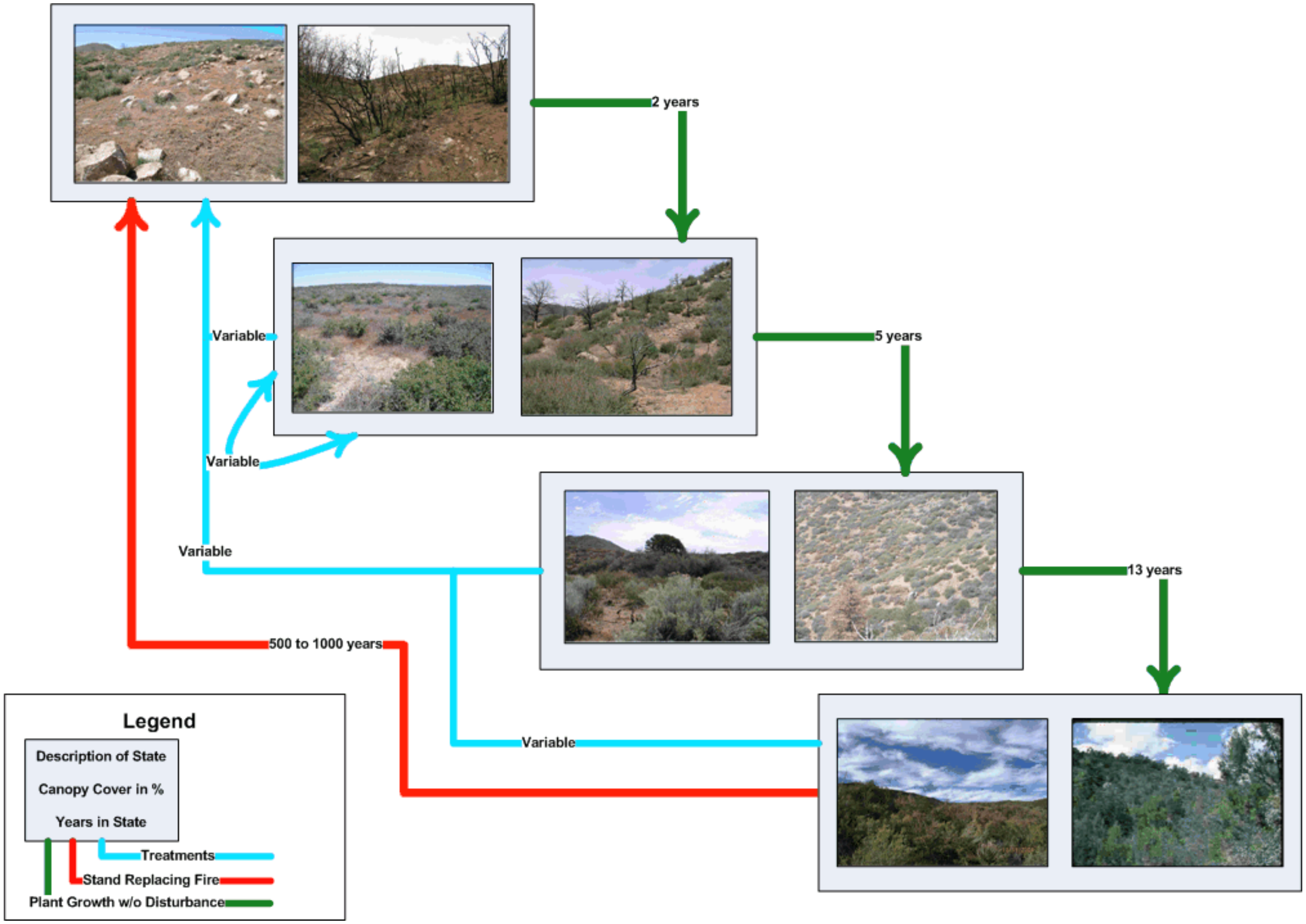

Figure 15-3. Photographic depiction of current conceptual state and transition model for the interior chaparral vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown or variable, respectively, is the notation. Bottom photographs courtesy of Jeff Saroka (USFS).

15.2 Model Parameters

Below in Tables 15-1 and 15-2, we identify the transitions, and probabilities associated with those transitions, used for both historic and current VDDT model runs.

Table 15-1. Identification of historic transition types, probabilities, and source of information used to inform the interior chaparral VDDT model.

Transition Type	Transition Frequency or Length	Source	Assumptions
Stand Replacing Fire	Every 20 to 100	Pase and Brown 1982; Sneed and others 2002; Wright and Bailey 1982	Based on indirect (Pase and Brown 1982; and Wright and Bailey 1982) and direct (Sneede and others 2002) lines of evidence we compiled a fire return interval of between 20 and 100 years for the interior chaparral vegetation type.
Plant Growth w/o Disturbance	2,5, and 13 years	Carmichael and others 1978, Cable 1975; Lillie and others 1964; Pase and Brown 1982; Pase and Ingebo 1965; Pond and Cable 1960	We identified multiple sources (with similar results) that documented the time it took chaparral vegetation to reach the canopy cover classes of each state. Specifically, multiple studies showed that by 7 years chaparral has returned to pre-burn densities and/or densities high enough to eliminate the grass understory (Cable 1975; Carmichael and others 1978; Pond and Cable 1962). Additionally, it is suggested that chaparral quickly recovers to high cover levels (30 to 80%) but may not burn until roughly 20 years have passed (Cable 1975). Based on these studies we determined how long it would take to transition from one state to the next in the absence of a disturbance. We also used this information to determine the canopy cover ranges for each state.

Table 15-2. Identification of current transition types, probabilities, and source of information used to inform the interior chaparral VDDT model.

Transition Type	Transition Frequency or Length	Source	Assumptions
Treatments (Mechanical, herbicide, and/or prescribed fire)	Variable, not included in model	Baldwin 1968; Cable 1975; Lillie and others 1964; Pase and Ingebo 1965; Pond and Cable 1960	We identified multiple studies, conducted primarily within the Tonto National Forest, that documented a wide range of mechanical, chemical, and fire treatments for interior chaparral vegetation. We concluded that the type of treatment was variable and occurred on a relatively small portion of interior chaparral within Arizona and New Mexico, hence we decided not to model this parameter in the regional current model.
Non-native seeding	Variable, not included in model	Cable 1975; Hibbert and others 1974	We found documentation of the seeding of non-native perennial grasses (*Eragrostis lehmanniana* and E*ragrostis. curvula*) following conversion treatments. However, as with the treatment parameter, we determined that seeding was variable and occurred on a relatively small portion of interior chaparral in Arizona and New Mexico, hence we decided not to model this parameter in the current model.
Stand Replacing Fire	Every 0 to 500 years	Savage and Swetnam 1990; Swetnam and Betancourt 1998; Sneede and others 2002	We based our estimate of fire on fire scar data. Specifically, regional fire scar data, along with data more localized to southeastern Arizona and the Prescott Basin, shows drastic declines in fires from 1900 to present (Savage and Swetnam 1990; Sneede and others 2002; Swetnam and Betancourt 1998). Given this information, we estimated a fire return interval of 0 to every 500 years.

Transition Type	Transition Frequency or Length	Source	Assumptions
Plant Growth w/o Disturbance	2,5, and 13 years	Carmichael and others 1978, Cable 1975; Lillie and others 1964; Pase and Brown 1982; Pase and Ingebo 1965; Pond and Cable 1960	We identified multiple sources (with similar results) that documented the time it took chaparral vegetation to reach the various canopy cover classes of each state. Specifically, multiple studies suggest that by 7 years chaparral has returned to pre-burn densities and/or densities high enough to eliminate the grass understory (Cable 1975; Carmichael and others 1978; Pond and Cable 1962). Additionally, it is suggested that chaparral quickly recover to high cover levels (30 to 80%) but may not burn until roughly 20 years have passed (Cable 1975). Based on these studies we determined how long it would take to transition from one state to the next in the absence of a disturbance. We also used this information to determine the canopy cover ranges for each state.

15.3 Results

Results of the interior chaparral historic VDDT model show some variability in the 900 year average percent of the modeled landscape in each state based on the fire return interval range (Table 15-3). Even with this variability, all models showed a consistent pattern of the majority of the historic vegetation occurring in the Dense Shrub (greater than 21 years olds) state (83.6 %, 76.2 % and 50.5 % +/- 2.3 %, 3.2 %, and 5.2 % for fire return intervals of 100, 60, and 20 years respectively). A comparison of simulated historic conditions and current conditions shows an increase in the amount of Dense Shrub (21+ years) present under current management (Table 15-4). Specifically, increases of between 50% and 13% are seen for the 100, 60, and 20 year historic runs compared to the 0 to 500 year current runs.

Table 15-3. Results of the interior chaparral historic VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (60 years), high (100 years), and low end (20 years), of the estimated fire return interval range.

Fire Return Interval Modeled	**Model Output**	**Grass & Forb**	**Grass & Shrub**	**Dense Shrub (7 to 20 years)**	**Dense Shrub (21 + years)**
Every 100 years	Average	1.6	3.3	11.5	83.6
	Minimum	0.0	0.4	5.6	76.6
	Maximum	4.7	8.0	18.5	90.1
	Standard Deviation	0.8	1.1	2.0	2.3
Every 60 years	Average	2.4	4.8	16.7	76.2
	Minimum	0.0	0.6	7.2	66.2
	Maximum	6.7	11.0	26.3	88.7
	Standard Deviation	1.2	1.7	2.8	3.2
Every 20 years	Average	4.9	9.9	34.7	50.5
	Minimum	0	0.3	18.2	34.4
	Maximum	12.8	19.6	50.1	72.2
	Standard Deviation	2.3	3.1	4.9	5.2

Table 15-4. Results of the interior chaparral current VDDT model, reported as the 120 year end value for average, minimum, maximum, and standard deviation of the percent of the modeled landscape in each state.

Fire Return Interval Modeled	**Model Output**	**Grass & Forb**	**Grass & Shrub**	**Dense Shrub (7 to 20 years)**	**Dense Shrub (21 + years)**
No Fire					
	Average	0	0	0	100
	Minimum	0	0	0	100
	Maximum	0	0	0	100
	Standard Deviation	0	0	0	0

Every 1000 years	**Average**	0.2	0.4	1.4	98.0
	Minimum	0.0	0.2	1.0	97.0
	Maximum	0.5	0.9	2.1	98.5
	Standard Deviation	0.2	0.2	0.4	0.4
Every 500 years	**Average**	0.4	0.8	2.3	96.5
	Minimum	0.0	0.2	1.3	95.6
	Maximum	1.1	1.2	3.5	97.5
	Standard Deviation	0.3	0.3	0.7	0.7

15.4 Discussion

These results suggest that the last 120 years of land management, mainly fire suppression, have had some effect on historic chaparral landscape conditions. Changes primarily within stand age and density are reasonable as we wouldn't expect 120 years of fire suppression to have large effects on vegetation structure within a PNVT with a historic fire return interval of 20 to 100 years. Additionally, this is in agreement with changes within chaparral vegetation documented by Huebner and others (1999). These results suggest that current interior chaparral vegetation has lost the mosaic of less dense and younger aged states. Maintenance of the 20 to 100 year fire return interval will be important for restoring and maintaining the historic range of conditions for this landscape in the future.

15.5 Interior Chaparral Model References:

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Chapter 16 - Madrean Pine Oak Woodland

16.1 Madrean Pine Oak Woodland Vegetation Dynamics

Madrean pine oak woodlands have been shown to trend away from woodlands with an open pine dominated overstory and oak understory to dense woodlands with oaks and other sprouting tree species dominating the overstory, see Figure 16-1 (Barton 1999, 2002, 2005; Barton and other 2001; Fulé and Covington 1996, 1997, 1998). This change in density and species composition is the result of switching from centuries of predominantly low intensity frequent surface fires to decades without fire followed by stand replacing fires (Barton 1999, 2002; Fulé and Covington 1998). Frequent low intensity surface fires allow fire resistant pines to dominate a site and maintain an open stand structure. Additionally, short fire free periods, between 20 and 30 years in length, are necessary to allow for periodic pine regeneration (Barton and others 2001). The recent shift in fire regimes has been more conducive to the reduction of pine species and regeneration of oak and other sprouting species.

Graphical and photographic depictions of these vegetation dynamics are displayed in Figures 16-1, 16-2 and 16-3; results of the quantitative VDDT models are shown in Tables 16-1 and 16-2.

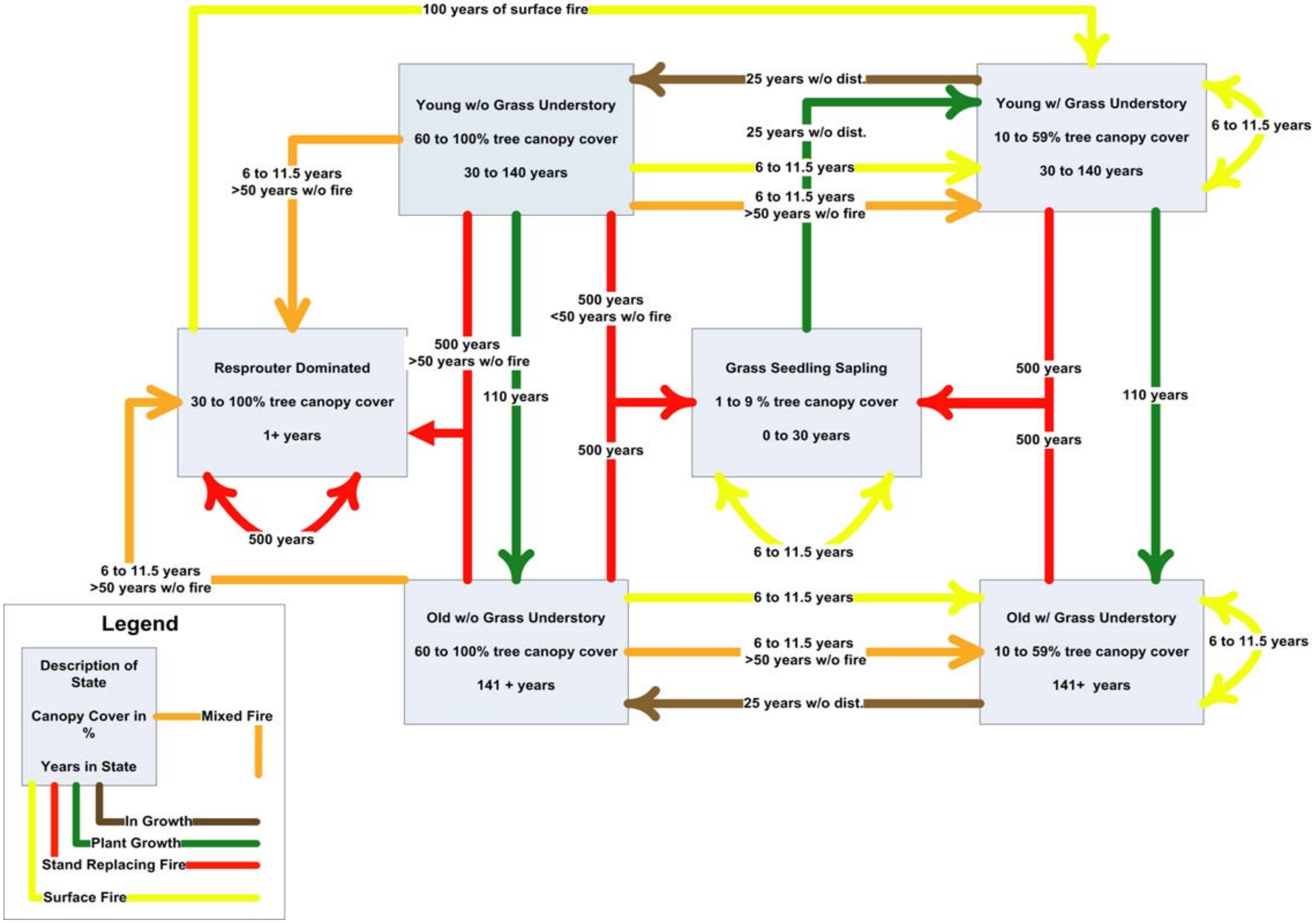

Figure 16-1. Conceptual historic state and transition model for the Madrean pine oak woodland vegetation type. Frequency of transitions are noted.

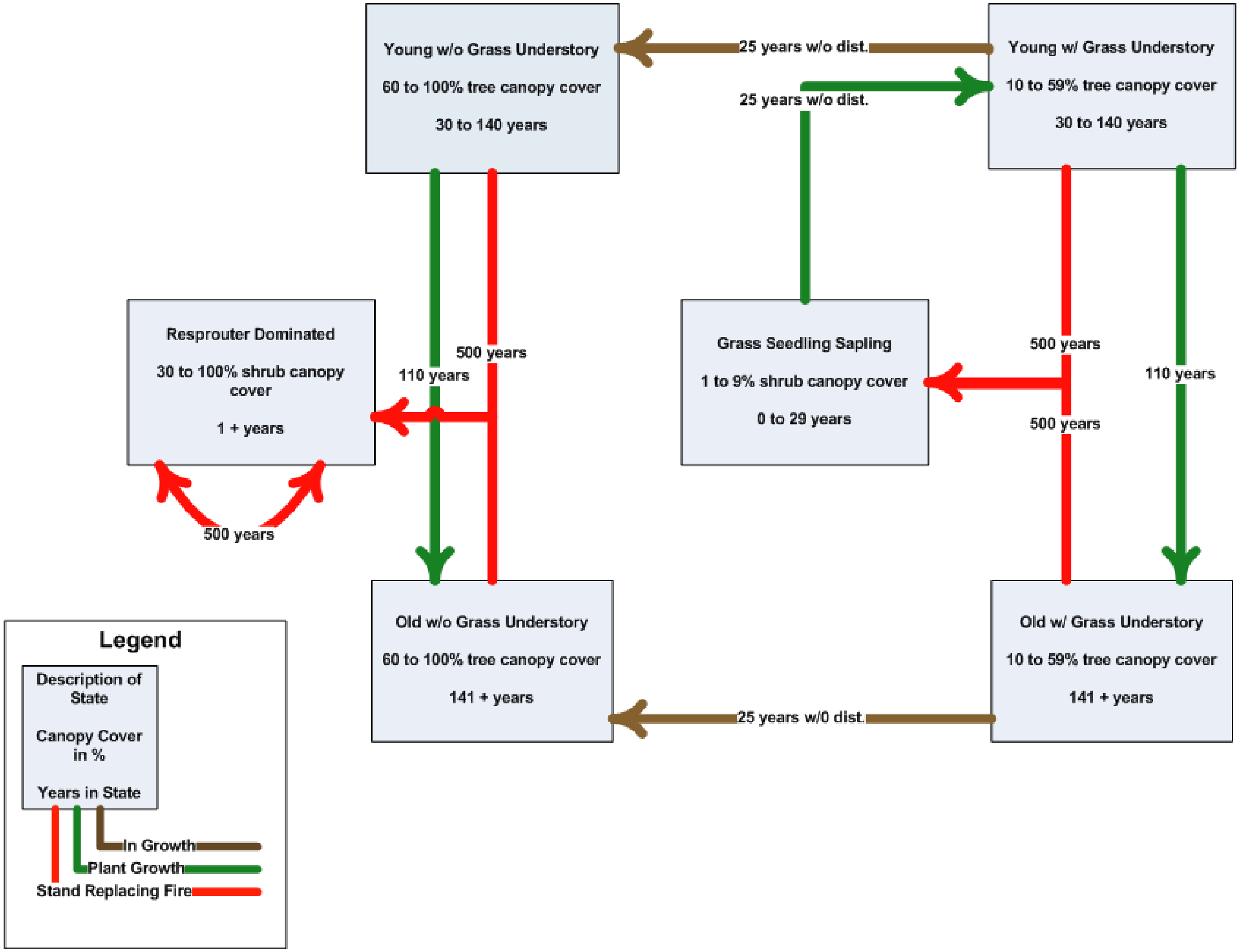

Figure 16-2. Conceptual current state and transition model for the Madrean pine oak woodland vegetation type. Frequency of transitions are noted.

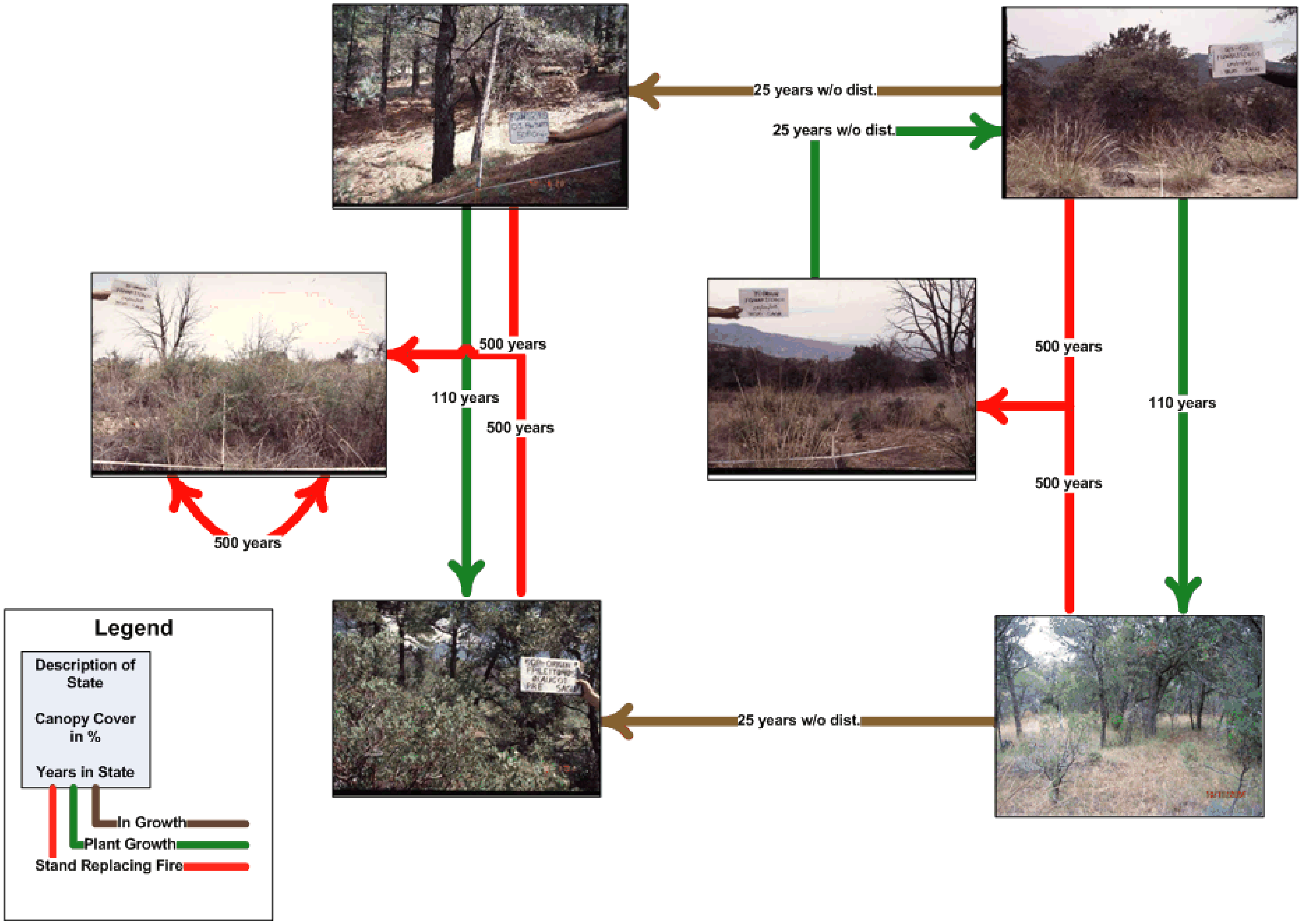

Figure 16-3. Photographic depiction of current conceptual state and transition model for the Madrean pine oak woodland vegetation type. Frequency of transitions are noted. Photographs courtesy of James Leckie (Saguaro National Park) and Coronado National Forest.

16.2 Model Parameters

Below in Tables 16-1 and 16-2, we identify the transitions, and probabilities associated with those transitions, used for both historic and current VDDT model runs.

Table 16-1. Identification of historic transition types, probabilities, and source of information used to inform the Madrean pine oak woodland VDDT model.

Transition Type	Transition Frequency or Length	Source	Assumptions
In Growth	After 25 years without fire	Barton and others 2001	Based on Barton and others (2001) study that showed that pines need 20 to 30 years of fire free periods in order to establish, we determined that in growth would occur in areas that had been free of fire for 25 years.
Plant Growth (25 years)	After 25 years without fire	Barton and others 2001	Based on Barton and others (2001) study that showed that pines need 20 to 30 years of fire free periods in order to establish, we determined that plant growth would occur in areas that had been free of fire for 25 years, resulting in a transition from a grass/seedling/sapling state to a young pine oak woodland.
Plant Growth (110 years)	110 years	Andrew Barton personal communication	Based on unpublished data on the age of various size classes of pine in the Chiricahua mountains of southeastern Arizona, we determined that it takes 110 years for a pine to grow from a young pine (5 cm dbh) to an old pine (20 cm dbh).
Stand Replacing Fire	Every 500 years	Don Falk and Jose Iniguez, personal communication and unpublished data.	In a study of large homogenous stands of ponderosa pine, Falk could find no tree-ring evidence of a stand-replacing fire that was 1000 acres or larger in size over a period of 500 years. His sampling design, however, could not discern a stand-replacing fire of 100 acres in size over the same time period. In general, Falk's theoretical and empirical work indicates that estimates of fire frequency are area sensitive. Based on a recently-completed study of Madrean pine-oak in the Catalina and Rincon Mountains, Jose Iniguez estimates the probability of a SRF 1000 acres in size in large homogenous pine-oak stands as once every 600 years (0.0017) and in small heterogeneous stands as once every 300

			years (0.0033). Considering pine-oak woodland regionally (1,416,765 acres in Region 3) and using 1000 pixels in the simulation, pixel size would be 1,416 acres which is close to the event size (1000 acres) of the above SRF estimates. For this reason and because there was no a priori way to distinguish large homogeneous vs. small heterogeneous areas of Madrean pine-oak based on existing data, we used a SRF probability of every 500 years (0.002). It is important to note that there is little difference in final output using a SRF probability of 0.001 and 0.002 which bracket the values for a 1000-acre stand-replacing event discussed above. However, there is a significant change in the output if a SRF probability of 0.0033 is used. SRF in open canopy states (with grass understory) moves the vegetation to the Grass Seedling Sapling state as it does in closed canopy states providing that ≤ 50 years has elapsed since the last low-intensity surface fire. If more than 50 years has elapsed since the last surface fire, oaks and other resprouters increase in abundance relative to pines and SRF will move the vegetation to the resprouter state (with few to no pines) as a result of the differential response of pines and oaks to high intensity, stand-replacing fire (Fulé and others 1999; Barton 1999, 2002)
Surface Fire	Every 6 to 11.5 years	Fulé and Covington 1996, 1997, 1998, 1999; Fulé and others 2005; Kaib and others 1996; Swetnam and Baisan 1996; Swetnam and others 1992; Swetnam and others, in press).	Mean fire return intervals (MFRI) for fires that scarred 25% or more trees at 17 sites located in Arizona and northern Mexico ranged between 6 and 14.5 years. This is more than a two-fold difference between sites at the low vs. high end of the MFRI range. All but 1 site (Bear Canyon, Catalina Mtns) fell between 6 and 11.5 years with Bear Canyon appearing to be an outlier so we eliminated this site from the analysis. Running the model at an MFRI of 14.5 years (0.069) significantly changes the output. Surface fire potentially affects all vegetation states including the Resprouter Dominated state which can move back into pine-oak woodland following a period of 100 years that allows for pine recruitment and frequent surface fires to reduce resprouting species (Drew Barton, pers. comm.).

Mixed Severity Fire	At the same frequency as surface fires but only affects states or stands that have not experienced a surface fire for > 50 years	Barton 1999, 2002; Fulé and others 2000	An increase in the density of pines, oaks and other resprouters as well as a compositional shift in favor of oaks following a fire-free period of more than 50 years leads to a mixed severity fire with some patches experiencing high-intensity, crown replacing fire that moves vegetation from a closed canopy stand (i.e., without a grass understory) to the resprouter dominated state with few to no pines. Other patches, however, experience lower intensity fire that thins vegetation and moves it from a closed canopy stand to an open canopy stand with a grass understory (after a post-burn recovery period). The probability of high vs. low intensity effects on patches for the mixed severity fire was assumed to be equal (= 0.5). The model was also run assuming the probability of high vs. low intensity effects was 0.67 vs. 0.33, respectively. The results were very similar to those in Table 6-3, except that more of the historic landscape ended up in the resprouter dominated state (1.3 to 5.2% more, on average, depending on the FRI); output for this second set of runs is available from D. Gori on request.

Table 16-2. Identification of current transition types, probabilities, and source of information used to inform the Madrean pine oak woodland VDDT model.

Transition Type	Transition Frequency or Length	Source	Assumptions
In Growth	After 25 years without fire	Barton and others 2001	Based on Barton and others (2001) study that showed that pines need 20 to 30 years of fire free periods in order to establish, we determined that in growth would occur in areas that had been free of fire for 25 years.
Plant Growth (25 years)	After 25 years without fire	Barton and others 2001	Based on Barton and others (2001) study that showed that pines need 20 to 30 years of fire free periods in order to establish, we determined that plant growth would occur in areas that had been free of fire for 25 years, resulting in a transition from a grass/seedling/sapling state to a young pine oak woodland.
Plant Growth	110 years	Andrew Barton, personal	Based on unpublished data on the age of various size classes of

(110 years)		communication	pine in the Chiricahua mountains of southeastern Arizona, we determined that it takes 110 years for a pine to grow from a young pine (5 cm dbh) to an old pine (20 cm dbh).
Stand Replacing Fire	Every 500 years	Don Falk and Jose Iniquez, personal communication and unpublished data.	In a study of large homogenous stands of ponderosa pine, Falk could find no tree-ring evidence of a stand-replacing fire that was 1000 acres or larger in size over a period of 500 years. His sampling design, however, could not discern a stand-replacing fire of 100 acres in size over the same time period. In general, Falk's theoretical and empirical work indicates that estimates of fire frequency are area sensitive. Based on a recently-completed study of Madrean pine-oak in the Catalina and Rincon Mountains, Jose Iniguez estimates the probability of a SRF 1000 acres in size in large homogenous pine-oak stands as once every 600 years (0.0017) and in small heterogeneous stands as once every 300 years (0.0033). Considering pine-oak woodland regionally (1,416,765 acres in Region 3) and using 1000 pixels in the simulation, pixel size would be 1,416 acres which is close to the event size (1000 acres) of the above SRF estimates. For this reason and because there was no a priori way to distinguish large homogeneous vs. small heterogeneous areas of Madrean pine-oak based on existing data, we used a SRF probability of every 500 years (0.002). It is important to note that there is little difference in final output using a SRF probability of 0.001 and 0.002 which bracket the values for a 1000-acre stand-replacing event discussed above. However, there is a significant change in the output if a SRF probability of 0.0033 is used.

16.3 Results

Results of the Madrean pine oak woodland historic VDDT model show some variability in the 900-year average percent of the modeled landscape in each state based on the fire return interval range (Table 16-3). Even with this variability, all models showed a consistent pattern of the majority of the historic vegetation occurring in the Old Pine Oak w/ Understory state (48.6 %, 60.2 % and 71.2 % for fire return intervals of 11.5, 9, and 6 years respectively). A significant proportion of the vegetation also occurs in the Young Pine Oak with Understory state. A comparison of simulated historic conditions and current conditions shows an increase in the amount of Old Pine Oak w/o Understory (76.5 %) from all historic model runs (Table 16-4). Additionally, the current model has 14.8% of the landscape in the Resprouter Dominated state compared to less than 10% in the historic model (Table 16-4).

Table 16-3. Results of the Madrean pine oak woodland historic VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the middle (9 years), high (11.5 years), and low end (6 years), of the estimated mean fire return interval range for surface fire and 500 year stand replacing fire.

Fire Return Interval Modeled	Model Output	Grass/ Seedling/ Sapling	Young Pine Oak w/o Understory	Young Pine Oak w/ Understory	Old Pine Oak w/ Understory	Old Pine Oak w/o Understory	Resprouter Dominated
Every 11.5 years	**Average**	3.6	6.8	24.5	48.6	6.9	9.6
	Minimum	1.0	1.5	16.8	31.8	2.0	4.9
	Maximum	7.1	8.7	41.5	55.6	15.7	18.4
	Standard Deviation	0.9	1.4	1.8	2.4	2.0	1.6
Every 9 years	**Average**	4.5	3.3	23.7	60.2	3.9	4.5
	Minimum	1.9	0.8	18.5	38.1	0.9	1.4
	Maximum	7.9	7.0	43.3	66.6	12.7	11.0
	Standard Deviation	0.9	0.9	1.3	1.9	1.6	0.9
Every 6 years	**Average**	5.1	0.9	20.2	71.2	0.9	1.7
	Minimum	2.4	0.0	15.2	43.1	0.0	0.0
	Maximum	9.2	5.3	42.0	77.7	4.0	9.4
	Standard Deviation	0.9	0.3	1.2	1.4	0.6	0.3

Table 16-4. Results of the Madrean pine oak woodland current VDDT model, reported as the 120 year end value for average, minimum, maximum, and standard deviation of the percent of the modeled landscape in each state. Current model simulates 120 years without surface fire with a stand replacing fire interval of every 500 years.

Fire Return Interval Modeled	Model Output	Grass/ Seedling/ Sapling	Young Pine Oak w/o Understory	Young Pine Oak w/ Understory	Old Pine Oak w/ Understory	Old Pine Oak w/o Understory	Resprouter Dominated
Every 500 years							
	Average	0.0	8.7	0.0	0.0	76.5	14.8
	Minimum	0.0	8.1	0.0	0.0	73.9	13.3
	Maximum	0.0	9.6	0.0	0.0	78.4	16.6
	Standard Deviation	0.0	0.5	0.0	0.0	1.3	1.0

16.4 Discussion

These results highlight the importance of maintaining a frequent surface fire regime within the Madrean pine oak woodland vegetation type. Without frequent fires the modeled landscape shifted from **open canopied** young and old pine oak woodland to **denser canopied** old pine oak woodland. The increase in canopy cover and oak abundance resulted in a larger proportion of the stand replacing fires transitioning to a resprouter dominated community. These modeled results are similar to the changes that have been documented within the vegetation type (Barton 1999, 2002; Barton and others 2001; Fulé and Covington 1997, 1998; Fulé and others 2000).

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Chapter 17 - Vegetation Model for the Mixed Conifer Forest Type

17.1 Mixed Conifer Vegetation Dynamics – Mixed conifer with aspen forests occur at high elevation between ponderosa pine and spruce-fir forests, and in cool canyons and other microsites throughout Arizona and New Mexico, covering approximately 1,780,100 acres, of which 68% or 1,216,300 acres are managed by the United States Forest Service (USGS 2004). Like many vegetation types in the Southwest, mixed conifer forests are very diverse in composition, structure, and ecological processes, and at least 11 different Associations or habitat types have been identified for the two-state region (Moir and Ludwig 1979), and more detailed habitat descriptions are given in the Mixed Conifer HRV document (Schussman and Smith 2006).

Prior to about 1880, these forests experienced frequent fire of mixed severity and intensity, with widespread lower intensity and severity surface fires, and smaller scale crown fires of higher intensity and severity (Allen 2002, Fule and others 2003, Vankat 2004). The frequency of fire reported in several studies for the SW ranges widely (1 to 89 years); it is very likely that these studies cover more than one disturbance regime, but lack of complete vegetation descriptions in the literature precludes discriminating between and among different vegetation-disturbance groups at this time (Swetnam and Baisan 1996, Touchan and others 1996, Fule and others 2003, Heinlein and others 2005). Swetnam and others identified this issue in developing a detailed fire chronology for Mount Graham in the Pinaleno Mountains of SE Arizona, when they reported an apparently anomalous fire frequency of one to three fires per decade for mixed conifer forests above 9,000 feet elevation (Swetnam and others 2005), compared to their previous studies which had reported a fire frequency of one fire occurring every 15 to 30 years (Swetnam and Baisan 1996). They attributed this unusually high fire frequency to the topographical setting, with mixed conifer forests occurring on the tops of ridges surrounded by ponderosa pine forest, and hypothesized that the more fire-frequent ponderosa pine forests below conveyed fire up into the mixed conifer forests (Swetnam and others 2005). Similarly, Heinlein and others (2005) reported one fire per decade for two ponderosa pine-mixed conifer sites on the slopes of the San Francisco Peaks in northern Arizona, with a range of 3 to 21 years. Fule and others (2003) reported that large fires (those scarring 25% or more of the sampled trees) were infrequent, occurring very 31 years on average.

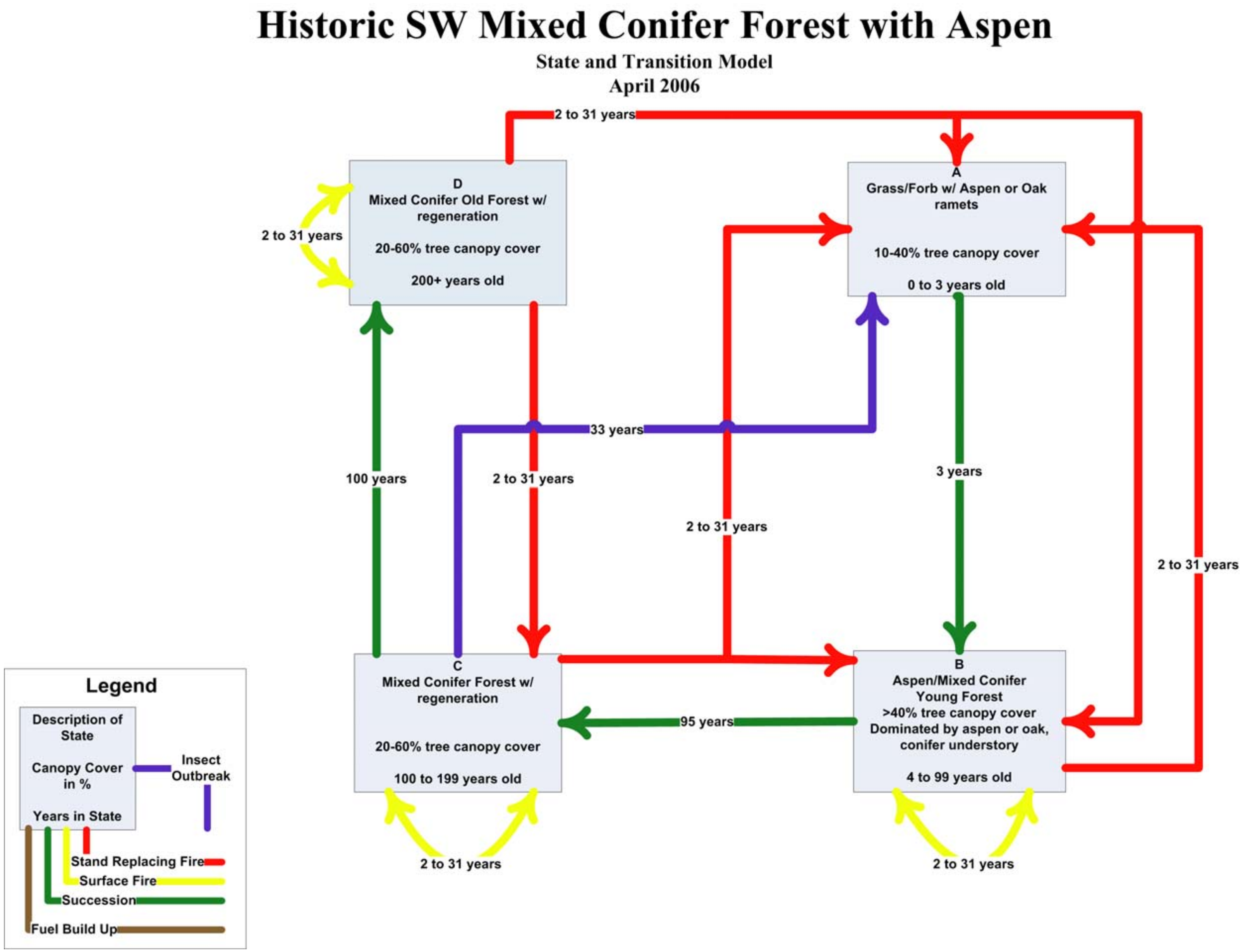
Since we have not yet differentiated mixed conifer forest into more than one disturbance group, for the purposes of this model we averaged the reported values for large fires (>= 25% scarred) of the mean fire return interval (MFI), minimum, and maximum for 14 studies reported for mixed conifer of one or more types throughout Arizona and New Mexico. We used an average value of 15 years, an average maximum of 33 years, and an average minimum of 5 years for the pre-1880 *Historic* fire return interval for mixed conifer forests.

Grazing animals have been implicated for the removal of surface fuels and the subsequent decrease in surface fire frequency (Leopold 1923, Savage and Swetnam 1996), but the extent of influence of grazing animals has not been quantified systematically across the Southwest Region. Swetnam and Baisan (1996) determined that climate has influenced fuel production and fuel moisture, thereby affecting the fire regime of mixed conifer forests, with large fire years correlated with drought years, but contrary to ponderosa pine, periods of drought and high fire activity do not need to be preceded by one to three years of higher than average precipitation, because high fire years are driven by fuel moisture levels rather than fine fuel production. Years with fewer fires are correlated with higher precipitation (Swetnam and Betancourt 1990). Western spruce budworm (*Christoneura occidentalis*) has periodically affected mixed conifer forests in both historic and current times (Swetnam 1987). The size of patches of forest affected by budworm in historic times is unknown, but may be available from current outbreaks (Swetnam 1987). Swetnam (1987) reported that the average interval between outbreaks during both historic and modern or current times is the same value, 33.8 years. Lynch and Swetnam (1992) reported that western spruce budworms do not defoliate old-growth trees preferentially, but do defoliate mature trees.

Quaking aspen (*Populus tremuloides*) is an important early seral species in some mixed conifer forests (Bartos 2001). Following disturbance, aspen can produce 12,000 sprouts/acre (wildfire in New Mexico: Patton and Avant 1970) to 20,000 stems/acre (clear-fell in Arizona: Rolf 2001). Aspen quickly self-thins due to competition and shade intolerance: One study showed a 38% mortality rate for 3- and 4-year old suckers on clear-cut plots in Arizona (Jones 1975). Depending on the intensity of the disturbance event, and the proximity, fecundity and density of surviving conifers, and site characteristics, reseeding by conifers begins and shade-tolerant species successfully recruit into the understory of the aspens. As the conifers grow to a position where they overtop the aspen, aspen sprouting decreases, and decay organisms begin to dismantle the senescing aspen trees (Jones and DeByle 1985). This process may take as little as 80 to 100 years on small, even-aged patches of aspen on poor sites, or it may take 200 to 300 years where aspen has experienced long intervals between disturbance events, and in the presence of multiple age classes (Jones and DeByle 1985).

However, the introduction of exotic Rocky Mountain elk (*Cervus elaphus*) in the early part of the 20th century, and the concurrent extirpation of large predators such as the wolf and grizzly bear may have contributed to excessive browsing pressure on aspen that has been documented in other parts of the intermountain West (Hessl 2002). Bailey and Whitham (2002) reported that after three growing seasons, elk had consumed 36 to 85% of aspen shoots in an unfenced burned area within a mixed conifer-ponderosa pine forest in northern Arizona. However, five years after this burn, not one of seventy regeneration plots outside of elk fences showed any living aspen sprouts, indicating heavy browsing by ungulates (Rolf unpublished data). In 1991, an elk-proof fence was removed from an aspen stand that was clear-felled and fenced in 1986 (Shepperd and Fairweather 1994). After one growing season, elk had reduced total stems in the newly unfenced area by 40%, and the authors cautioned that nearly all remaining stems less than 1.5 ft tall were damaged, as were half of the mid-size (1.5 to 4.5 ft) stems, and 60% of the large (>4.5 ft tall) stems. Most of the severely wounded stems also were infected with Cytospora canker (*Cytospora chrysosperma*) (Shepperd and Fairweather 1994). The effects of browsing on aspen may not be as severe throughout the entire range of mixed conifer forests, especially where elk are not present.

17.2 Vegetation Models - Based on this understanding of vegetation dynamics, we created state and transition models depicting historic (pre-1880) and current (1880 to present) vegetation dynamics within this forest type (Figures 17-1 through 17-2). Additionally, we used information from the state and transition models to develop quantitative Vegetation Dynamics Development Tool (VDDT) models. The VDDT software allows the user to model succession as a series of vegetation states that differ in structure, composition, and cover and to specify the amount of time it takes to move from one vegetation state to another in the absence of disturbance. Various disturbance agents affecting the movement of vegetation between states can then be incorporated (e.g., surface fires, stand-replacing fires, grazing and browsing, insect outbreaks). By varying the types and rates of disturbance across the landscape, the effects of different management treatments, such as wildland fire use, fire suppression, prescribed burning, grazing practices, and mechanical fuel treatments, on future vegetation can be investigated. While VDDT models can be used to "game play" with different management scenarios, the models we ran in this analysis only include states and transitions for which there is published information to support their inclusion within the model. We discuss model parameters, output, and analysis below (Tables 5 through 8).



Historic SW Mixed Conifer Forest with Aspen

State and Transition Model

April 2006

Figure 37-1. Conceptual Historic state and transition model for the mixed conifer with aspen vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

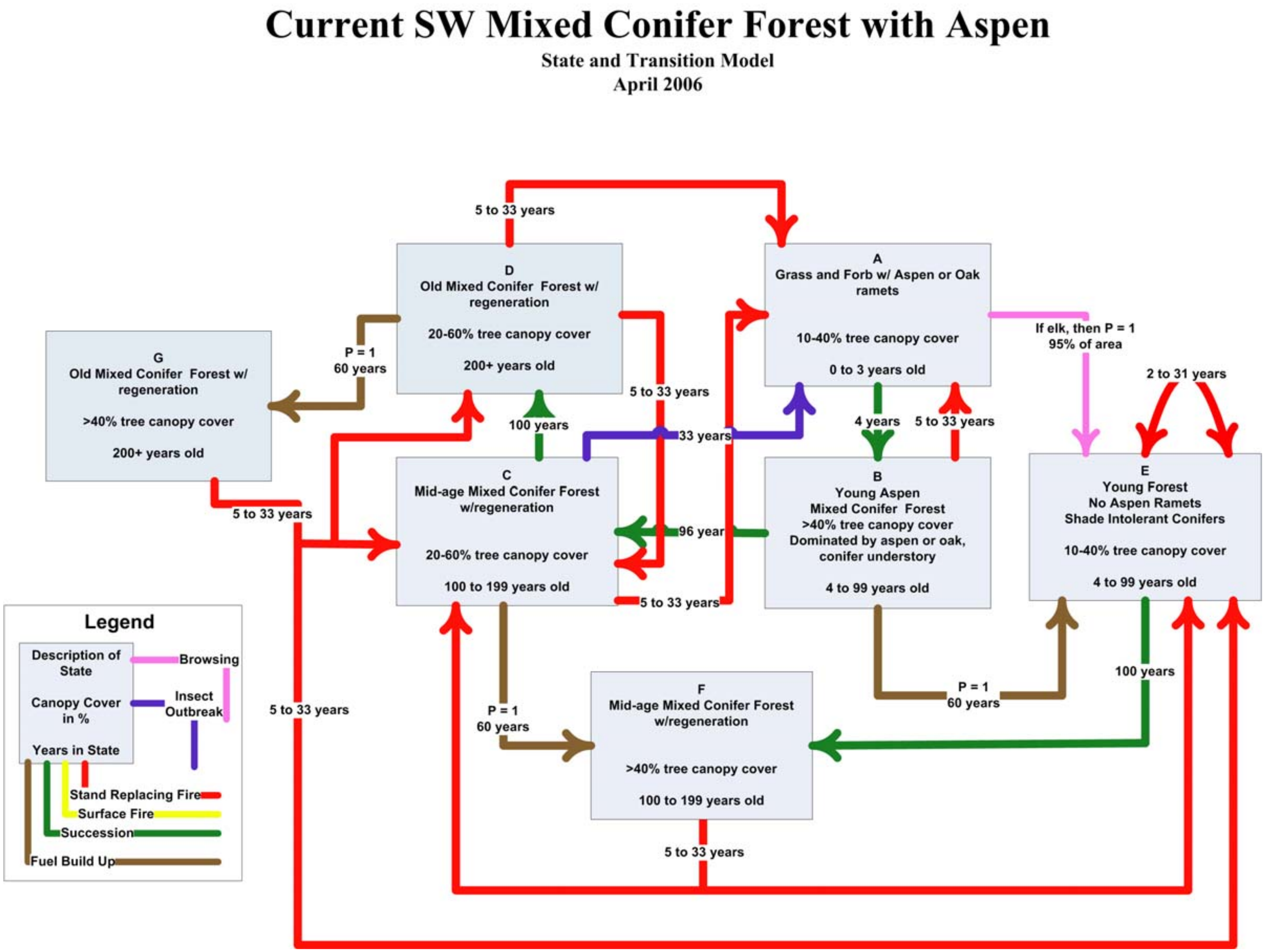


Figure 17-2. Conceptual Current state and transition model for the mixed conifer with aspen vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation. Dashed outlines represent states which may have been uncharacteristic for the historic period.

Model Parameters

In Tables 17-1 and 17-2 below, we describe the parameters included or not included within the Historic and Current VDDT models, as well as the sources of information and any assumptions used to create model parameters.

Table 17-1. Identification of Historic transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Aspen regeneration A to B	0-3 years	Patton and Avant 1970, Rolf 2001, Jones 1975,	Aspen ramet formation is vigorous for 1-5 years after disturbance, after which time it decreases.
Plant Growth B to C	95 years from young to mid-age forest	Jones and DeByle 1985	We assume that the transition from young forest to mid-age forest takes approximately 95 years, as aspens self-thin, conifer trees in the understory grow into the canopy and begin to overtop aspen, and vigor of aspen declines.
Plant Growth C to D	100 years from mid-age forest to old/mature forest	Old growth characteristics begin developing at around 200 years (Mehl 1992).	We assume that transition from mid-age to mature/old forest takes approximately 100 years, with multiple conifer regeneration events occurring, aspen decline is nearly complete, and stand structure has been opened by multiple cycles of surface and stand replacing fires.
Surface Fire	5 to 33 years between fires	Swetnam and Baisan 1996, Touchan and others 1996, Fule and others 2003, Heinlein and others 2005, Swetnam and others 2005..	These data are based on direct evidence (fire scar data). We modeled the endpoints of the range (5,33), and we used the average of large fires (>=25% scarred trees), using 15 years as the average for modeling purposes. Surface fire is assumed to maintain a given class or state, rather than cause its transition to another state.
Stand Replacing Fire	5 to 33 years between fires.	Stand replacing fire was reported to be rare and small in area (<250 acres) prior to 1880 (Grissino-Mayer and others 1995, Allen and others 2002, Fule and others 2003, Vankat 2006). Small proportion of landscape (1 to 10%) effectively reduces frequency of stand replacing fire from once every 15 years to once every 150 to 1500 years.	Stand replacing fire occurred in small patches that had variable effects on that patch depending on fuels, weather, and site conditions. We assumed that 1 to 10% of the landscape was affected by stand replacing fire, returning the patch to the younger states in equal proportions (Fule and others 2003). E.g., stand replacing fire in D takes equal, low proportions of area to regeneration state A, young forest B, and mid-age forest C.
Insects	33 years between	Swetnam 1987, Lynch and Swetnam 1992, Swetnam and Lynch 1993.	Western spruce budworm outbreaks occurred once every 33 years on average, preferentially during wet periods, and

Transition Type	Transition Frequency or Length	Sources	Assumptions
	outbreaks		primarily on mid-age forests. We assumed that the area affected by defoliations was small (<5% of landscape) but openings would be moved back to the regeneration state (A).

Table 17-2. Identification of Current transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and its effect on vegetation included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Fuel Build Up	Once every year.	Several authors have documented the cessation of surface fires around 1880, which has led to the accumulation of fuels (Covington and Moore 1994, Swetnam and Baisan 1996, Allen and others 2002, Fule and others 2003).	We assume that it would take approximately 60 years of growth since the last surface fire to move from an open canopy state (<30% canopy cover) to a higher canopy state (>30%).
Surface Fire	Not Used in Current model	Covington and Moore 1994, Swetnam and Baisan 1996, Sneed and others 2003, Allen and others 2002, Fule and others 2003.	Based on direct observation, we assume that surface fire has ceased at the scale of this model (~2 million acres). Occasional surface fires do occur, but not at the same scale, and typically enough fuels have accumulated in most areas to quickly transition surface fires to stand replacing fires. Prescribed fire and fire use fires are occurring in some areas at some times, but not within the range of variability for this system.
Stand Replacing Fire	Once every 5 to 33 years	Cessation of surface fires and accumulation of fuels and development of fuel ladders has not yet led to an increase in the frequency of stand replacing fires, but the size of patches affected by these fire events may be increasing (Swetnam and others 2005).	We assumed that stand replacing fires have not changed in frequency, but that they have changed in how much area they affect, and which states they move forests to. We assumed that stand replacing fires currently move 10-20% of the landscape area (double historic values) to equal proportions of less dense or younger forest classes (e.g. G goes to E, C and D; F goes to C and E, but E does not go to B because aspen has been removed by browsing, so E stays at E.)
Insect Outbreak	33 years between outbreaks	Swetnam 1987, Lynch and Swetnam 1992	Western spruce budworm outbreaks occurred once every 33 years on average, preferentially during wet periods, and primarily on mid-age forests. We assumed that the area

			affected by defoliations was small (<5% of landscape) but openings would be moved back to the regeneration state (A).
Browsing	Once every year in presence of elk over 95% of landscape	If elk are present, they browse aspen until it does not produce ramets within 2-5 years (Bailey and Whitham 2002, Rolf unpublished report).	We assumed that elk have moved aspen recruitment dynamics into a new trajectory, where regeneration by conifers occurs not under the shade of aspens, but in full sun, preferentially selecting for shade intolerant species such as ponderosa pine. Since aspen are short-lived trees, this process may not be reversible unless mature aspen is still extant.
Plant Growth E to F	95 years from young to mid-age forest	Jones and DeByle 1985	We assume that the transition from young forest to mid-age forest takes approximately 95 years, as conifer trees in the understory grow into the canopy. Tree composition may be different from historic based on above effects of browse pressure and lack of surface fires.
Plant Growth F to G	100 years from mid-age forest to old/mature forest	Old growth characteristics begin developing around 200 years (Mehl 1992).	We assume that transition from mid-age to mature/old forest takes approximately 100 years, with multiple conifer regeneration events occurring, aspen decline is nearly complete, and stand structure has closed due to infill by regeneration and growth of mid-age trees.
Silvicultural Activities	Highly variable through time and across space, thus not included in the model.	Mixed conifer forests have been logged and thinned since the 1850s to 1880s, with silvicultural prescriptions ranging from clear-cutting to thinning of pole and smaller trees (Bahre 1985).	We assume that the model overestimates the proportion of the current landscape in the Old Forest open and closed classes (D and G) due to the loss of many of the larger trees to timber harvest that is not accounted for in the model.

17.3 Results – Results of the Historic mixed conifer model indicate a small amount of variability in the 900-year average for each state based on the fire interval range (Table 17-3). The two least frequent FRIs (FRI=15 and 33) predicted that the largest portion of the landscape (46% to 71%) would be in the open Mature/Old Forest (State D), whereas the most frequently burned landscape moved about 20% of the landscape to this oldest class. The most frequently burned landscape had the greatest proportion of its landscape in the Young Forest class (B: 44 to 52%). The least frequently burned landscape (FRI=33) had only about 10% of the landscape in the Young Forest class (B), and less than 1% in the Grass/Forb regeneration state (A). The amount of Grass/Forb regeneration state (A) increased with the frequency of fire, with the most frequently burned landscape model producing less than 1% to almost 5% of the landscape in state A. The amount of Mid-Age forest varied the least among the three models, with a range from 18% to 33% of the landscape in this class following 900 years of growth, insects and fire.

The Current mixed conifer with aspen model, which was run for 120 years following the development of Historic conditions, had very different results from the Historic model (Table 17-4). Old forest open (State D) has been reduced by about half for all three fire interval models, and Aspen Regeneration (A) has been reduced by about 75% for the two least frequently burned landscapes, and by about 90% for the most frequently burned landscape. With an FRI of 33 years, slightly less than a third of the landscape is in the Old Forest Closed state (G), slightly more than a third is in the Old Forest Open (D), and around 10% is in C, E, and F. Also, with an FRI=33, the two youngest historic classes are not well represented (A<1%, B<3%), and over 50% of the landscape is in states that are not characteristic based on the Historic Model (States E, F, and G). With the average FRI (15 years), 56% of the landscape is uncharacteristic, and for the minimum FRI (5 years), almost 70% of the landscape is uncharacteristic.

Table 12-3. Results for the Historic mixed conifer with aspen VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (15 years), maximum (33 years), and minimum (5 years) of the estimated fire return interval range.

Fire Return Interval (FRI) Modeled	**Model Output % Area Modeled**	**Grass/Forb with aspen or oak ramets A Open**	**Young Forest B Closed**	**Mid-Age Forest w/ regen C Mixed-Open**	**Mature/Old Forest w/ regen D Open**
Every 33 years	Average	0.27	11.23	20.28	68.23
	Minimum	0.03	9.56	18.22	65.59
	Maximum	0.60	13.18	22.57	70.65
	Standard Deviation	0.17	1.06	1.28	1.55
Every 15 years	Average	0.66	21.18	29.07	49.09
	Minimum	0.18	18.39	26.87	45.75
	Maximum	1.27	23.75	31.48	52.71
	Standard Deviation	0.33	1.67	1.36	2.11
Every 5 years	Average	2.61	47.99	30.31	19.09
	Minimum	0.79	44.11	27.28	16.34
	Maximum	4.69	51.92	33.38	21.53
	Standard Deviation	1.19	2.39	1.85	1.55

Table 17-4. Results of the Current mixed conifer with aspen forest VDDT model, reported as the 120 year end value for average, minimum, maximum, and average standard deviation of the percent of the modeled landscape in each state.

Fire Return Interval Modeled	**Model Output % Area Modeled**	**Grass/Forb with aspen or oak ramets A Open**	**Young Forest B Closed**	**Mid-Age Forest C Mixed-Open**	**Mature / Old Forest D Open**	**Young Forest E Open**	**Mid-age Forest F Closed**	**Mature / Old Forest G Closed**
Every 33 years	Average	0.07	1.88	9.35	36.82	11.25	11.33	29.30
	Minimum	0.03	1.61	8.75	35.92	9.97	10.47	28.19
	Maximum	0.15	2.21	10.32	37.71	12.69	12.35	30.56
	Standard Deviation	0.04	0.18	0.45	0.56	0.85	0.59	0.73
Every 15 years	Average	0.10	3.54	12.70	27.05	21.95	15.79	18.18
	Minimum	0.04	3.08	11.92	25.82	20.28	14.51	17.10
	Maximum	0.23	3.93	14.02	28.10	23.49	16.93	18.95
	Standard Deviation	0.06	0.25	0.63	0.76	0.97	0.72	0.57
Every 5 years	Average	0.29	5.99	15.69	9.16	46.79	18.20	3.88
	Minimum	0.16	5.62	14.55	8.15	43.55	16.32	2.92
	Maximum	0.48	6.45	16.89	10.24	49.78	19.94	5.04
	Standard Deviation	0.11	0.25	0.73	0.63	1.86	1.10	0.64

17.4 Discussion – These modeled scenarios implicate the importance of frequent surface fire in maintaining both ends of the age spectrum for mixed conifer with aspen forests. The reduction by half of the open, Mature/Old forest (State D) may have a large impact on wildlife species that depend on old trees and forests. Replacement of these older, more open-canopied forests with denser, closed-canopied forests may also affect the abundance and diversity of shrubs, understory grasses and forbs. Also, the loss of early aspen regeneration and subsequent young and mid-aged forest with a strong aspen component is important due to the potential decline of species of wildlife that utilize aspen. Typically, we think of aspen decline being caused by overtopping and succession by conifers (Shaw 2005). In the absence of disturbance such as fire, few young aspen ramets are produced, and aspen regenerates in very small patches. However, under the scenario presented in the current model, lack of surface fire is compounded by the addition of heavy browsing by an introduced ungulate, Rocky Mountain elk, which hastens the decline of aspen by reducing young ramets across the landscape, and changes its trajectory along a hypothetically irreversible pathway.

Under the current scenario, there is a large increase in the proportion of closed, mid- age to old age forests. In a forest system that has evolved with occasional, small scale stand replacing fire, the elimination of surface fire may lead to increased frequency, and probably more importantly, more widespread spatially stand replacing fire, of higher intensity, covering more area per decade than was experienced in the historic, or prehistoric periods.

17.5 Mixed Conifer Model References

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Chapter 18 - Ponderosa Pine–Bunchgrass Forest Model

18.1 Ponderosa Pine Vegetation Dynamics – Ponderosa pine-bunchgrass forests dominate the mid-elevation forests of Arizona and New Mexico, occurring in a diverse mixture of age-, size-, and canopy cover classes within an uneven-aged forest system. Prior to about 1880, these forests were much more open, with canopy cover rarely exceeding 30% (Pearson 1923, White 1985). Surface fires maintained these open forests with low intensity, widespread fires occurring at a frequency of about 5 to 36 years when the >25% trees scarred filter, or fire rotation values are used (Swetnam and Baisan 1996). Native Americans used fire and were a source of ignition in some places at varying times and seasons, but their contribution to the overall fire frequency is vastly overshadowed by lightning-caused fires (Allen 2002). Propagation of these lightning-initiated fires was supported by a contiguous bed of dense bunchgrasses, forbs and litter in the herbaceous layer. Following fire, and in conjunction with periods of high precipitation, ponderosa pine seedlings established in great numbers, but regeneration has been sporadic and patchy across space and through time (Mast and others 1999). Natural surface fires have been suppressed since about 1880, allowing for the accumulation of large quantities of surface fuels and fuel ladders, facilitating an increase in the frequency, size, and intensity of stand replacing fires (to about once every 100 years -- Savage and Mast 2005).

Grazing animals have been implicated for the removal of surface fuels and the subsequent decrease in surface fire frequency (Savage and Swetnam 1996), but the extent of influence of grazing animals has not been quantified systematically across the Southwest Region. Swetnam and Baisan (1996) determined that climate has influenced fuel production and fuel moisture, thereby affecting the fire regime of ponderosa pine, with large fire years correlated with drought years, especially when preceded by one to three years of higher than average precipitation. Years with fewer fires are correlated with higher precipitation (Swetnam and Betancourt 1990). Various species of bark-beetle insects are endemic to the Southwest, and while historic outbreaks are not well understood, current forests have experienced large-scale irruptions of bark beetles with a frequency of 1-2 outbreaks per century since disruption of the surface fire regime and the resulting increase in density of SW forests (Dahms and Geils 1997, Negron and others 2000).

Vegetation Models - Based on this understanding of vegetation dynamics, we created state and transition models depicting historic (pre-1880) and current (1880 to present) vegetation dynamics within this forest type (Figures 18-1 through 18-2). Additionally, we used information from the state and transition models to develop quantitative Vegetation Dynamics Development Tool (VDDT) models. The VDDT software allows the user to model succession as a series of vegetation states that differ in structure, composition, and cover and to specify the amount of time it takes to move from one vegetation state to another in the absence of disturbance. Various disturbance agents affecting the movement of vegetation between states can then be incorporated (e.g., surface fires, stand-replacing fires, grazing, insect outbreaks). By varying the types and rates of disturbance across the landscape, the effects of different management treatments, such as wildland fire use, fire suppression, prescribed burning, grazing practices, and mechanical fuel treatments, on future vegetation can be investigated. While VDDT models can be used to “game play” with different management scenarios, the models we ran in this analysis only include states and transitions for which there is published information to support their inclusion within the model. We discuss model parameters, output, and analysis below (Tables 18-1 through 18-4).

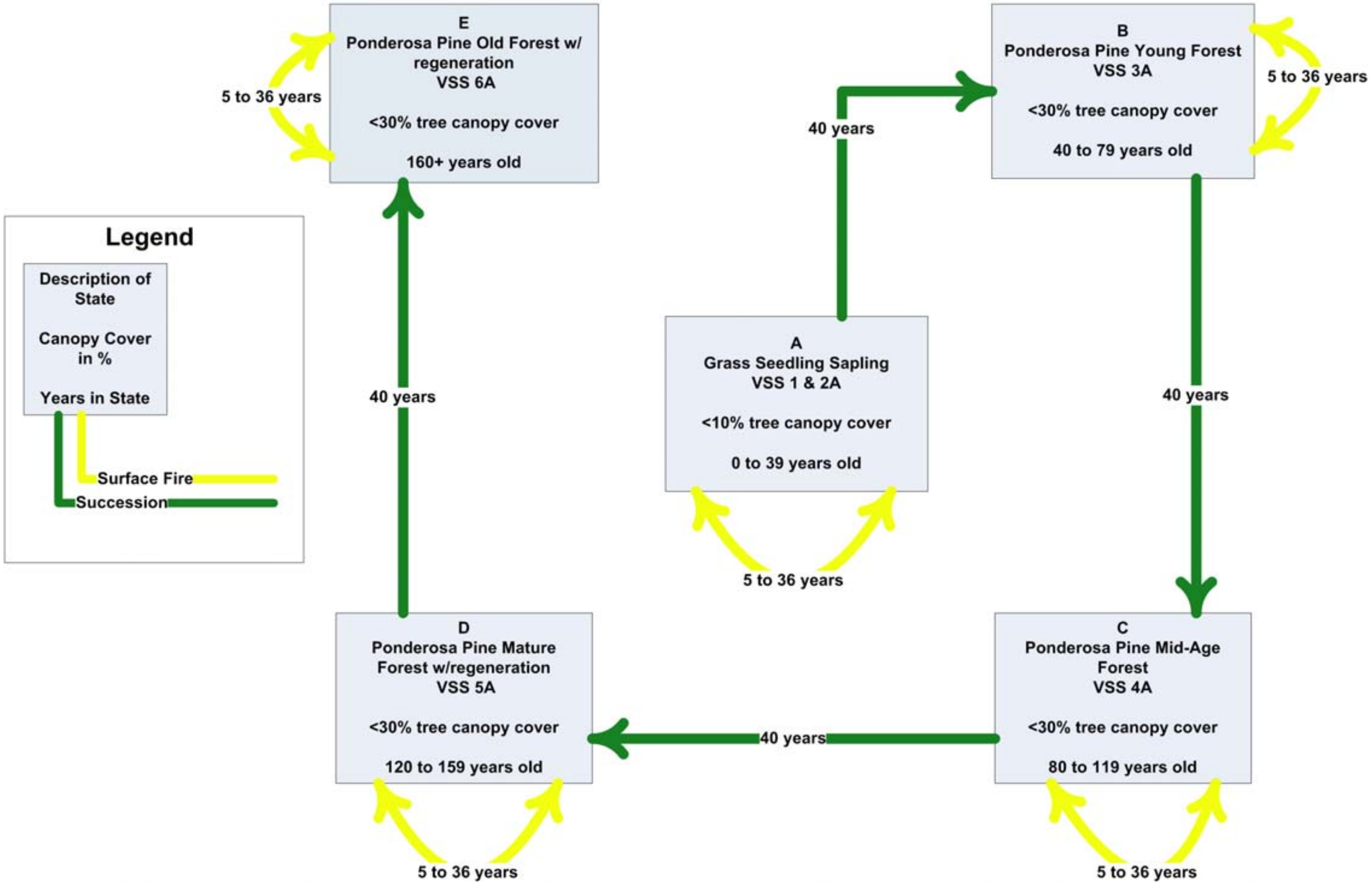

Figure 18-1. Conceptual Historic state and transition model for the ponderosa pine-bunchgrass vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

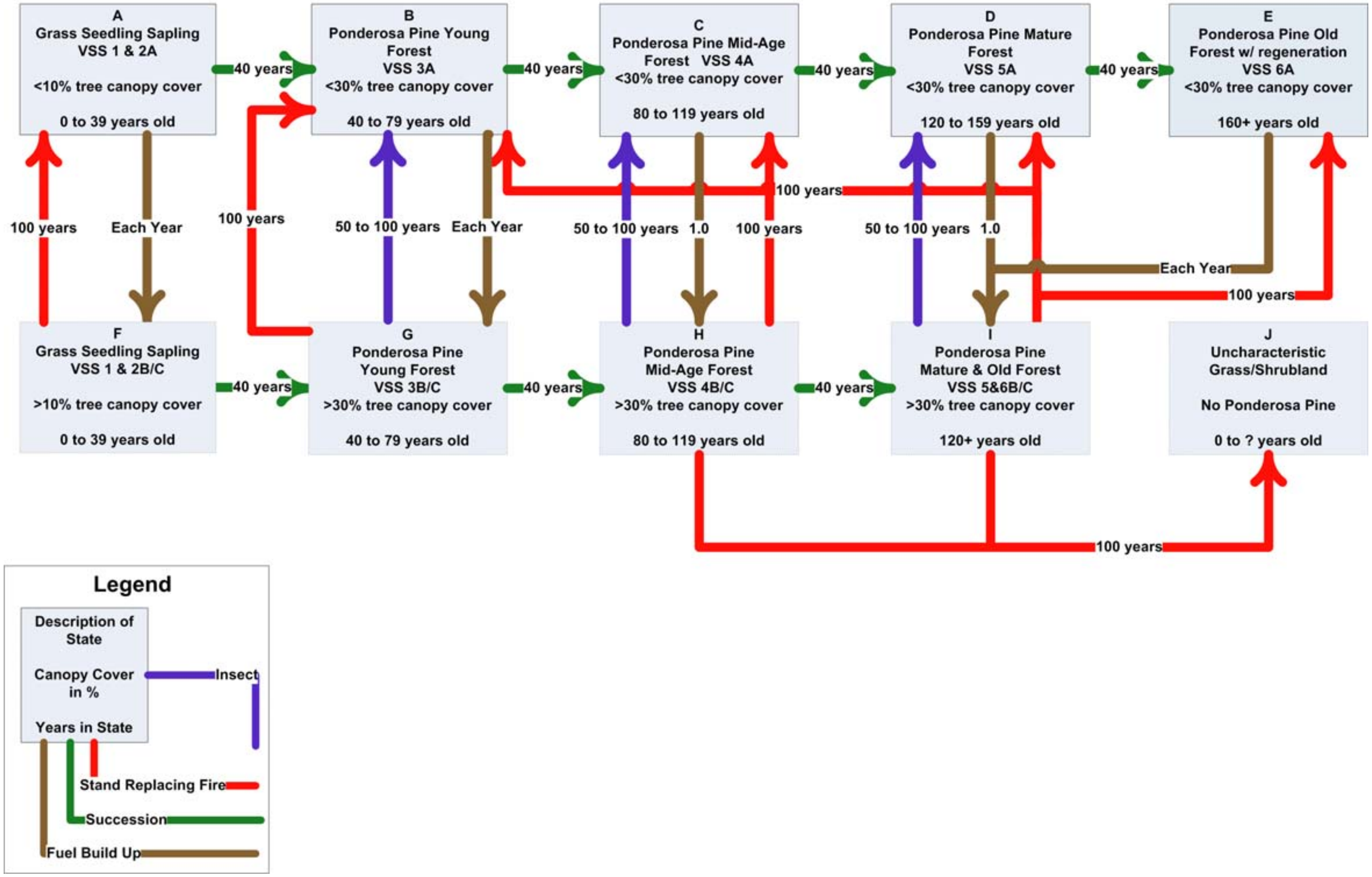

Figure 18-2. Conceptual Current state and transition model for ponderosa pine-bunchgrass native vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation. Dashed outlines represent states which may have been uncharacteristic for the historic period.

18.2 Model Parameters

In Tables 18-1 and 18-2 below, we describe the parameters included or not included within the Historic and Current VDDT models, as well as the sources of information and any assumptions used to create model parameters.

Table 18-1. Identification of Historic transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Regeneration from seed	Unknown, not used in model.	Seed production and seedling recruitment is highly variable in both space and time. Although episodic events have been documented for some areas (Mast and others 1999), there is insufficient information to assign a probability for this transition across the entire region.	Due to the lack of data on seedling recruitment, this transition is not included in the model, and hence the proportion of this seedling/sapling class of vegetation is presumed to be underestimated in the model
Surface Fire	5-36 years	Swetnam and Baisan 1996, Sneed and others 2003, Fule and others 2003.	These data are based on direct evidence (fire scar data). We modeled the endpoints of the range (5.4,36.3), and we averaged the range, using 15.6 years as the average for modeling purposes.
Stand Replacing Fire	Not used in model	Stand replacing fire was reported to be rare and small in area prior to 1880 (Moir and others 1997, Fule and others 2003, Falk 2004), and operating at the individual tree scale rather than on patches (<100 acres).	Stand replacing fire only occurred on individual trees (torching) and at very low frequency (once every 500 to 1,000 years).
Plant Growth	40 years between states	Transitions among model states were taken from silvicultural data summarized by Reynolds and others (1992).	We assume that transition from seedling/sapling to young forest takes approximately 40 years, and from young to mid-age forest, mid-age to mature forest, and mature forest to ld forest also take 40 years.

Table 18-2. Identification of Current transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and its effect on vegetation included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Fuel Build Up	Once every year after 25 years of growth.	Several authors have documented the cessation of surface fires around 1880, which has led to the accumulation of fuels (Covington and Moore 1994, Swetnam and Baisan 1996, Allen and others 2002, Fule and others 2003).	We assume that it would take approximately 25 years of growth since the last surface fire to move from an open canopy state (<30% canopy cover) to a higher canopy state (>30%).
Surface Fire	Not Used in Current model	Covington and Moore 1994, Swetnam and Baisan 1996, Sneed and others 2003, Allen and others 2002, Fule and others 2003.	Based on direct observation, we assume that surface fire has ceased at the scale of this model (9 million acres). Occasional surface fires do occur, but not at the same scale, and typically enough fuels have accumulated in most areas to quickly transition surface fires to stand replacing fires. Prescribed fire and fire use fires are occurring in some areas at some times, but not within the range of variability for this system.
Stand Replacing Fire	Once every 100 years	Cessation of surface fires and accumulation of fuels and development of fuel ladders has led to an increase in the frequency of stand replacing fires (Covington and Moore 1994, Swetnam and Baisan 1996, Covington and others 1997, Allen and others 2002). The effects of stand replacing fire on vegetation has been documented by Savage and Mast (2005).	We based our estimate of fire on fire scar data. Specifically, regional fire scar data shows drastic declines in fires from 1900 to present. Given these data, we estimated a fire occurrence of 1 in the last 100 years. Fires studied by Savage and Mast (2005) give proportions of ponderosa pine forests resulting from stand replacing fires of the 1940s to 1970s. These values were used in modeling the relative proportion of states resulting from stand replacing fire in ponderosa pine forests of the SW.
Insect Outbreak	Once every 50 to 100 years	Bark beetle insect outbreaks have occurred 1 to 2 times over the last century (Dahms and Geils 1997, Negron and others 2000).	Insect-mediated mortality of ponderosa pines is highly variable spatially and temporally, but stands seldom experience 100% mortality, and it is assumed that beetle

			outbreaks take patches from a closed to an open state of the same age.
Plant Growth	40 years	Transitions among model states were taken from silvicultural data summarized by Reynolds and others (1992).	We assume that transition from seedling/sapling to young forest takes approximately 40 years, and transitions from young to mid-age forest, mid-age to mature forest, and mature forest to old forest also take 40 years.
Silvicultural Activities	Highly variable through time and across space, thus not included in the model.	Ponderosa pine forests have been logged and thinned since the 1850s to 1880s, with silvicultural prescriptions ranging from clear-cutting to thinning of pole and smaller trees (Bahre 1985).	We assume that the model overestimates the proportion of the current landscape in the Mature to Old Forest open and closed classes due to the loss of many of the larger trees to timber harvest.

18.3 Results – Results of the Historic ponderosa pine-bunchgrass model indicate a small amount of variability in the 900-year average for each state based on the fire interval range (Table 18-3). All three FRIs predicted that a majority of the landscape (99%) would be in the open Old Forest (State E), with a very small proportion of the landscape in the Mature open forests, State D (<1%). Recall that the states represent uneven-aged stands or patches with the range of ages given representing the maximum age of the stand rather than the absolute range of ages within the patch.

The Current ponderosa pine-bunchgrass model, which was run for 120 years following the Historic conditions, had very different results from the Historic model (Table 18-4). Old forest open (State E) has been reduced 70-90%, while mature/old forest closed (State I) increased from 0% in the Historic model to a range of 53% to 57% in the Current model. The percentage of open states is very low (0 to 1%) in the current model, while there is low abundance of the closed states other than I(1-6%). Uncharacteristic grasslands (J) accumulated up to 16% of the landscape under the Current scenario, compared to 0% under the historic scenario. The minimum and average Fire Rotation values produced identical values for landscape proportion for all classes, indicating that the model is not sensitive to this variability in fire regime.

Table 18-3. Results for the Historic ponderosa pine-bunchgrass VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (15.6 years), maximum (36.3 years), and minimum (5.4 years) of the estimated fire return interval range.

Fire Return Interval or Rotation Modeled	**Model Output**	**Grass/ Seedling & Sapling A Open**	**Young Forest B Open**	**Mid-Age Forest C Open**	**Mature Forest D Open**	**Old Forest E Open**	**Grass/ Seedling/ Sapling F Closed**	**Young Forest G Closed**	**Mid-Age Forest H Closed**	**Mature /Old Forest I Closed**	**Unchar. Grassland J Open**
Every 36.3 years	Average	0.0	0.0	0.1	0.8	99.2	0.0	0.0	0.0	0.0	0.0
	Minimum	0.0	0.0	0.0	0.7	99.1	0.0	0.0	0.0	0.0	0.0
	Maximum	0.0	0.0	0.1	0.8	99.2	0.0	0.0	0.0	0.0	0.0
	Standard Deviation	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Every 15.6 years	Average	0.0	0.0	0.1	0.8	99.2	0.0	0.0	0.0	0.0	0.0
	Minimum	0.0	0.0	0.0	0.7	99.1	0.0	0.0	0.0	0.0	0.0
	Maximum	0.0	0.0	0.1	0.8	99.2	0.0	0.0	0.0	0.0	0.0
	Standard Deviation	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Every 5.4 years	Average	0.0	0.0	0.1	0.8	99.2	0.0	0.0	0.0	0.0	0.0
	Minimum	0.0	0.0	0.0	0.7	99.1	0.0	0.0	0.0	0.0	0.0
	Maximum	0.0	0.0	0.1	0.8	99.2	0.0	0.0	0.0	0.0	0.0
	Standard Deviation	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Table 18-4. Results of the Current ponderosa pine-bunchgrass forest VDDT model, reported as the 120 year end value for average, minimum, maximum, and average standard deviation of the percent of the modeled landscape in each state.

Fire Return Interval or Rotation Modeled	**Model Output by Class or State**	**Grass/ Seedling & Sapling A Open**	**Young Forest B Open**	**Mid-Age Forest C Open**	**Mature Forest D Open**	**Old Forest E Open**	**Grass/ Seedling/Sapling F Closed**	**Young Forest G Closed**	**Mid-Age Forest H Closed**	**Mature /Old Forest I Closed**	**Unchar. Grassland J Open**
Every 36.3 years	Average	0.1	0.2	0.2	0.5	23.2	3.8	5.5	4.5	52.7	9.4
	Minimum	0.0	0.0	0.0	0.2	23.1	2.5	4.3	3.7	49.1	7.6
	Maximum	0.4	0.6	0.5	0.8	23.3	5.2	7.1	5.5	57.1	11.7
	Standard Deviation	0.1	0.2	0.1	0.2	0.1	0.8	0.8	0.5	2.3	1.1
Every 15.6 years	Average	0.2	0.3	0.2	0.6	7.8	5.1	8.0	6.4	57.2	14.3
	Minimum	0.0	0.0	0.0	0.2	7.7	3.8	6.5	5.4	53.6	12.3
	Maximum	0.5	0.8	0.5	1.0	8.0	6.5	9.6	7.8	61.4	15.7
	Standard Deviation	0.2	0.2	0.2	0.2	0.1	0.8	0.9	0.7	2.3	1.0
Every 5.4 years	Average	0.2	0.3	0.2	0.6	7.8	5.1	8.0	6.4	57.2	14.3
	Minimum	0.0	0.0	0.0	0.2	7.7	3.8	6.5	5.4	53.6	12.3
	Maximum	0.5	0.8	0.5	1.0	8.0	6.5	9.6	7.8	61.4	15.7
	Standard Deviation	0.2	0.2	0.2	0.2	0.1	0.8	0.9	0.7	2.3	1.0

18.4 Discussion – These modeled scenarios implicate the importance of frequent surface fire in maintaining open ponderosa pine-bunchgrass ecosystems. When comparing the Historic versus the Current models, the increase in proportion of the landscape that is closed, and susceptible to stand replacing fires that result in uncharacteristic grasslands that are not forested, and may have a low probability of becoming reforested without costly intervention, is readily apparent. When comparing the model outputs to existing conditions, it is likely that the model overestimates all classes of mature and old forest, which in reality have been reduced as a result of timber harvest (Bahre 1985), and underestimate the seedling and sapling class abundance due to the lack of a regeneration transition that would increase these class abundances (States A and F). The abundance of these model states could be refined through a careful assessment of forthcoming datasets that will quantify their current abundance in the mid-scale vegetation analysis.

18.5 Ponderosa Pine Model References

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Chapter 19 - Spruce-Fir Forest Model

19.1 Vegetation Dynamics – Spruce-fir forests dominate the high elevation forests of Arizona and New Mexico, occurring in small mountain- and plateau-top forests throughout the region. The severe and cold environment found in high elevations generally reduces forest productivity, and slows succession. Most areas require hundreds of years to move from early successional stages to later, more mature stands (Moir 1993). In the central Rocky Mountains of Colorado, vegetation dynamics are more influenced by the type of disturbance than the spatial scale of the disturbance (Veblen 1986). Early seral species that establish after major disturbances such as fire, windthrow, avalanche, or insect outbreak are variable and include aspen, Douglas-fir, bristlecone pine, and white fir, as well as the dominant Engelmann spruce and corkbark fir. Disturbance does not recur for a period of 70 to 100 years (or more) due to lack of sufficient fuels (for fire), or biomass (for insects, windthrow or avalanche) (Veblen and others 1994, Vankat 2006). Aspen is an important component of some of the Habitat Types for 50 to 150 years but declines rapidly in density and canopy dominance as the coniferous canopy cover increases (Moir 1993). Without disturbance, in the Rocky Mountains of Colorado, Engelmann spruce slowly increases in dominance in the canopy or overstory, while corkbark fir increases in abundance in the understory (Aplet and others 1988). The spruce-fir forest continues to grow and develop, but is increasingly susceptible to disturbance events. The average longevities for tree species in the Southwest and southern to central Rocky Mountains was reported to be 300-350 years for corkbark fir, and 500 to 600 years for Engelmann spruce (Alexander 1987, Moir 1993). Currently, many spruce-fir forests on the north rim of Grand Canyon National Park (Vankat 2006) and the Pinaleno Mountains of southeastern Arizona (Koprowski and others 2005) are experiencing high mortality in older canopy trees due to a combination of drought and insects, especially of Engelmann spruce.

Both Engelmann spruce and corkbark fir are fire sensitive due to thin bark at all ages, and hence are unlikely to survive even low intensity fires and provide fire scars for analysis (Veblen and others 1994). Thus, most of the fire regime research has been accomplished using current tree and stand age or stand structure analysis (White and Vankat 1993, Fulé and others 2003), as well as fire scar analysis of adjacent forest types, which for spruce-fir is primarily mixed conifer forests at lower elevation (Grissino-Mayer and others 1995, Baisan and Swetnam 1995, Swetnam and others 2005). For the central Rocky Mountains of Colorado, spruce-fir forests burn as crown fires and at return intervals of centuries (Schoennagle and others 2004), although at lower elevation, some areas have experienced mixed severity and surface fires (Baker and Veblen 1990). Some have suggested that spruce-fir forests in the Southwest experience crown fire, with insufficient time having passed since the last crown fire for these forests to have experienced crown fire in contemporary time, at least for the small patch of spruce-fir forest on Mt. Graham in the Pinaleno Mountains (Swetnam and others 2005). Grissino-Mayer and others' (1995) analysis of stand structure in the Pinalenos led them to conclude that the fire return interval (FRI) for spruce-fir forest was 300-400 years. However, there is ample evidence to suggest that some spruce-fir forests in Arizona and New Mexico have a mixed severity fire regime that burned with a return interval on the order of decades rather than centuries (Dieterich 1983, Moir 1993, Fulé and others 2003, Vankat 2006).

At Grand Canyon National Park on the North Rim, Fulé and others (2003) reported that tree densities and fire-initiated/non-fire-initiated groups were very patchily distributed, and thus stands created by fire could not be discerned from analysis of satellite imagery. This diverse forest structure suggests a combination of a surface fire regime as well as crown-fire initiated groups or stands. They also reported that the mean fire interval (MFI) for spruce-fir was 8.8 years, 8.0 for 10% trees scarred, and 31.0 for 25% scarring (larger fires) for the period 1700 to 1879 (Fulé and others 2003). In addition, they reported a median fire return interval of 7 years, with minimum of 2 years and maximum of 32 years; a standard deviation of 7.7 years, and a Weibull median probability interval (WMPI) of 7.2 years. They found that upper elevation fire dates often coincided with fire dates for lower elevation forest types, suggesting that pre-1880 fires may have been very large (Fulé and others 2003). Most of the fires they studied occurred in summer, and large fire dates occurred in dry years that followed several wet years (Fulé and others 2003). Vankat (2006) asserted that the mixed severity fire regime of the North Rim spruce-fir forest might be a function of the relatively low elevation, but might also be a product of the Southwest's climate. Fulé hypothesized (2006 personal communication) from the 2003 study that south and west facing slopes were dominated by surface fires captured in pine fire scars, whereas north and east facing slopes only experienced stand replacing fires at much longer intervals. These two forest disturbance types are in close enough proximity that when viewed together as a landscape, a mixed severity fire regime is apparent.

Shoennagel and others (2004) hypothesized that the main variables controlling fires in the central Rocky Mountains of Colorado shift from climate at higher elevation, crown-fire prone mesic sites to a combination of climate and fuel related variables at drier mid-elevation, mixed fire regime sites. Fulé and others (2003) did caution that the mixed surface and crown fire regimes "appeared not to be stable over the temporal and spatial scales of this study." This suggests that the return interval itself may be dynamic, and the 279-year period for which they collected data may not have been sufficient to bracket the historical range of variability for fire return intervals for that site.

In a study of spruce-fir forests on Mt. Graham in the Pinaleno Mountains of SE Arizona, Swetnam and others (2005) studied age structure of existing stands, cross-dated with fire-scar dates from downslope mixed conifer trees, and determined that a widespread and severe fire occurred in 1685, and that it was a crown fire. Engelmann spruce and corkbark fir have been co-dominants since 1685, and there was a large pulse of corkbark fir recruitment in the mid-1800s (Grissino-Mayer and others 1995, Swetnam and others 2005). They also found that the growth rates of Douglas-fir trees that survived the 1685 widespread fire at the spruce-fir/mixed conifer forest ecotonal boundary were slow growing after the fire, indicating that they had been damaged by a severe crown fire. They also determined from relative ring width that Engelmann spruce and corkbark fir that established after the fire (i.e., during the 1690s and early 1700s) had rapid growth rates, probably indicating an open stand condition. Tree-age structure and fire-scar evidence also pointed to varying periods of tree recruitment after the 1685 burn, possibly indicating trees' response to a combination of climatic variation and fire events (Swetnam and others 2005). They concluded that the dominant FRI for spruce-fir forest on Mt Graham is 150+ years, with large, high severity crown fire behavior. But, they conceded that "occasional surface and ground fires crept into portions of this forest from adjacent mixed-conifer, and high severity, small-patch size (individual trees or groups) events probably also occurred in this zone" (Swetnam and others 2005).

Vankat (2006) concluded that the mixed severity fire regime of spruce-fir forests in the Southwest may follow a moisture and/or elevation gradient similar to the difference between southern and central Rocky Mountain fire regimes. Swetnam and others (2005) also postulated the possibility that historically, the long term stability of the spruce-fir forest was somewhat protected by the more frequent fire regime in the adjacent and down-slope mixed conifer forests. Frequent fires in the mixed conifer forests maintained more open stands, with low woody fuel accumulations, grassy understories, and higher crown base heights, which may have combined to produce lower intensity surface fires that would be less likely to transition into crown fires upon spread into the spruce-fir forests upslope. Swetnam and others (2005) have observed such fire behavior, and assert that the historic fire regime may have been responsible for maintaining a more patchy vegetation and fuel mosaic that was less conducive to crown fire. Based on Vankat (2006), Fulé and others (2003), and Swetnam and others (2005), we hypothesize that lower elevation (or latitude or moisture) spruce-fir forests may have more frequent, less severe fires more similar to surface fire regimes, and higher elevation/latitude/moisture forests may have infrequent, more severe crown fires, and some areas have both.

Historically, spruce beetles (*Dendroctonus rufipennis*) have had the greatest impact on spruce-fir forests of the central and southern Rocky Mountains, and in the Southwest (Alexander 1987). Historic photographs and tree-ring analysis indicate that there have been six major outbreaks since the middle 1800s, and between 1850 and 1880 a very large-scale outbreak affected forests from northern New Mexico to northern Colorado (Baker and Veblen 1990). These outbreaks are considered part of the natural variability of spruce-fir forests, and spruce beetles likely persist in small windthrow areas (Veblen and others 1991) and in live trees (Veblen and others 1994). Small to large predators such as nematodes to woodpeckers maintain populations at low levels, until conditions favor an outbreak (Alexander 1987). Factors influencing outbreaks include large diameter Engelmann spruce or canopy dominance, slowed growth, mild winters, and well-drained creek bottom sites (Veblen and others 1994, Bebi and others 2003).

19.2 Vegetation Models - Based on this understanding of vegetation dynamics, we created state and transition models depicting historic (pre-1880) and current (1880 to present) vegetation dynamics within this forest type (Figures 19-1 through 19-3). Additionally, we used information from the state and transition models to develop quantitative Vegetation Dynamics Development Tool (VDDT) models. The VDDT software allows the user to model succession as a series of vegetation states that differ in structure, composition, and cover and to specify the amount of time it takes to move from one vegetation state to another in the absence of disturbance. Various disturbance agents affecting the movement of vegetation between states can then be incorporated (e.g., surface fires, stand-replacing fires, grazing, insect outbreaks). By varying the types and rates of disturbance across the landscape, the effects of different management treatments, such as wildland fire use, fire suppression, prescribed burning, grazing practices, and mechanical fuel treatments, on future vegetation can be investigated. While VDDT models can be used to "game play" with different management scenarios, the models we ran in this analysis only include states and transitions for which there is published information to support their inclusion within the model. We discuss model parameters, output, and analysis below (Tables 19-1 through 19-5).

Historic Southwest Spruce-Fir Forest High Elevation

State and Transition Model

May 2006

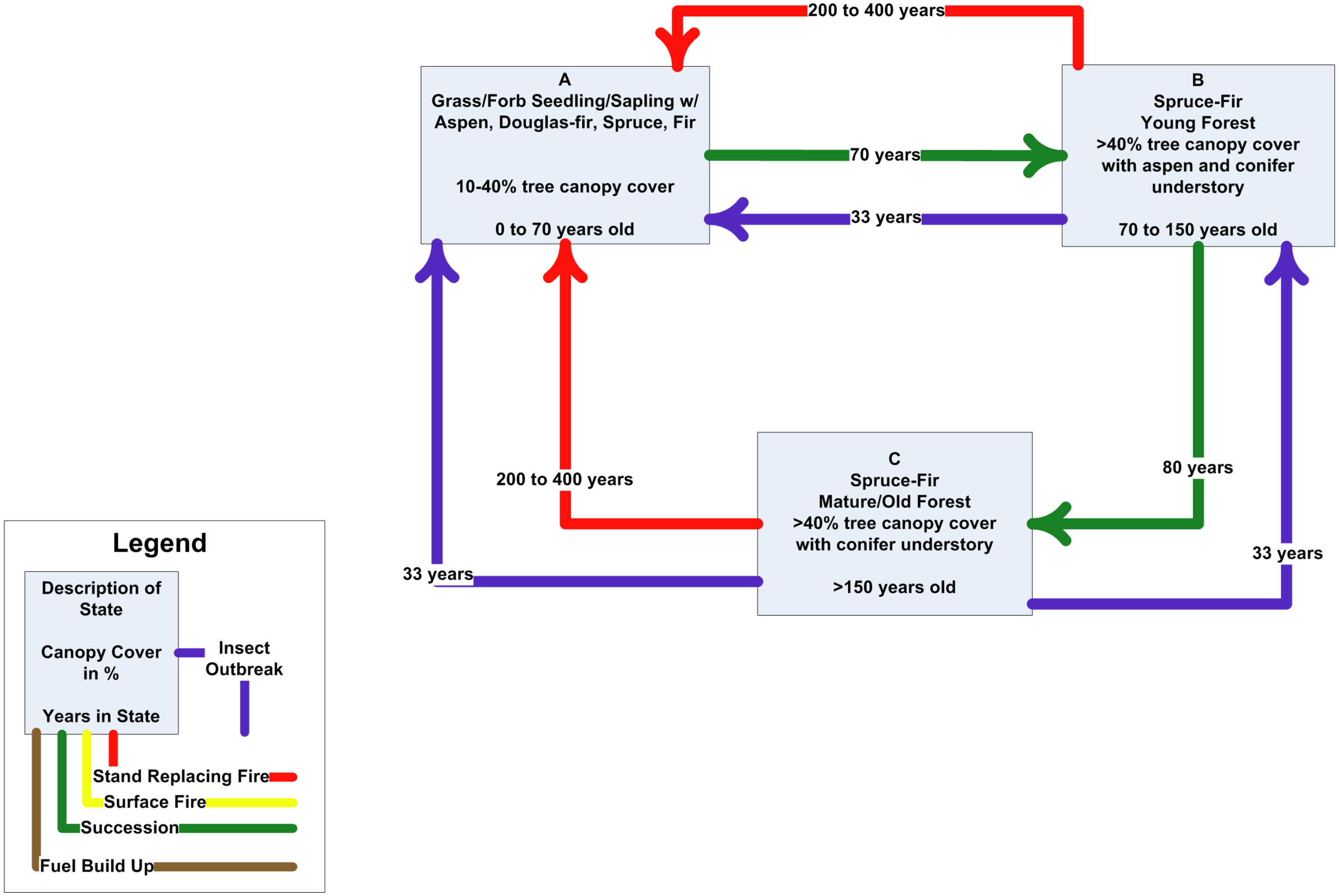

Figure 19-1. Conceptual Historic state and transition model for the spruce-fir forest high elevation vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Historic Southwest Spruce-Fir Forest Low Elevation

State and Transition Model

May 2006

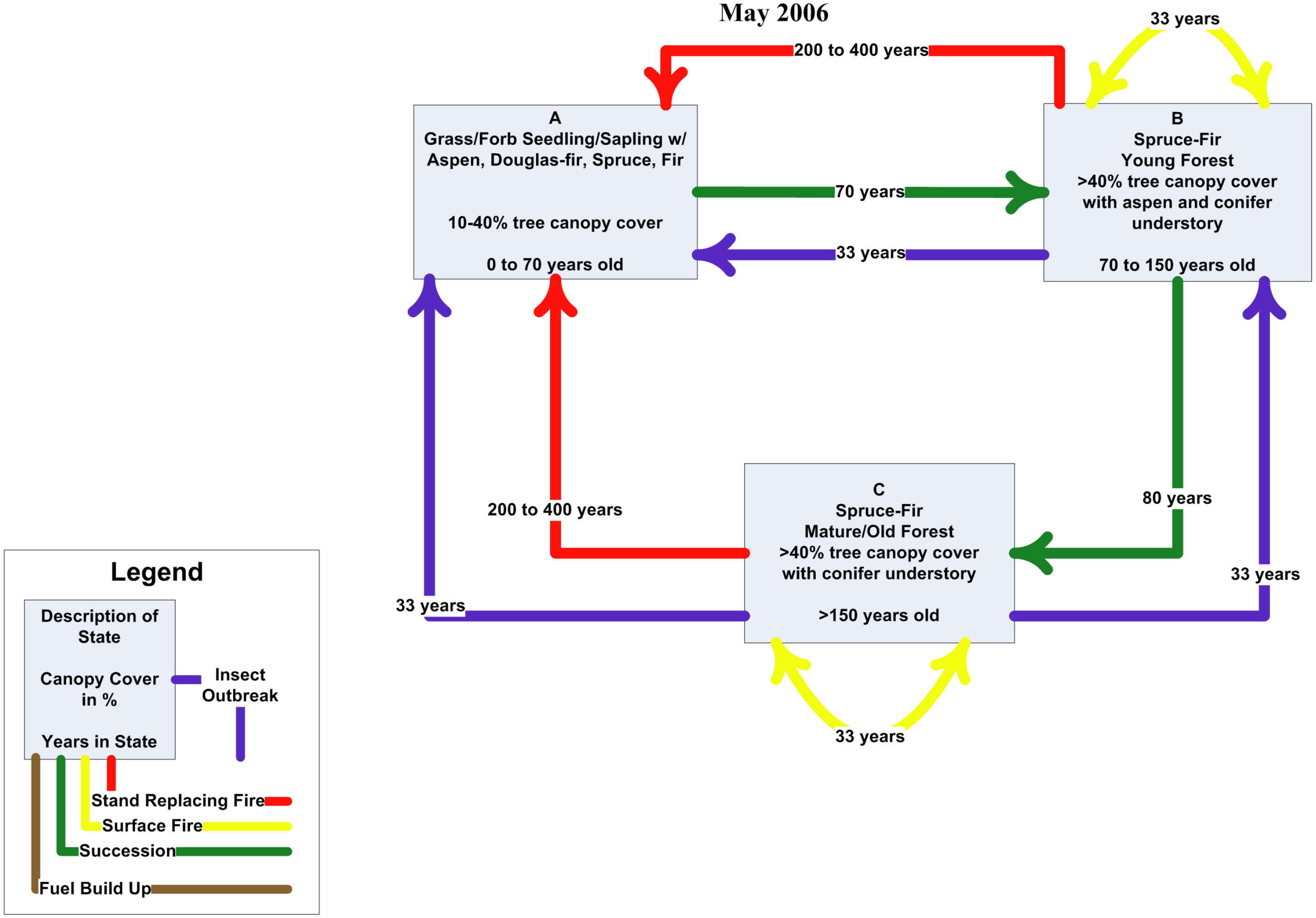

Figure 19-2. Conceptual Historic state and transition model for spruce-fir forest low elevation vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Current Southwest Spruce-Fir Forest High & Low Elevation

State and Transition Model
May 2006

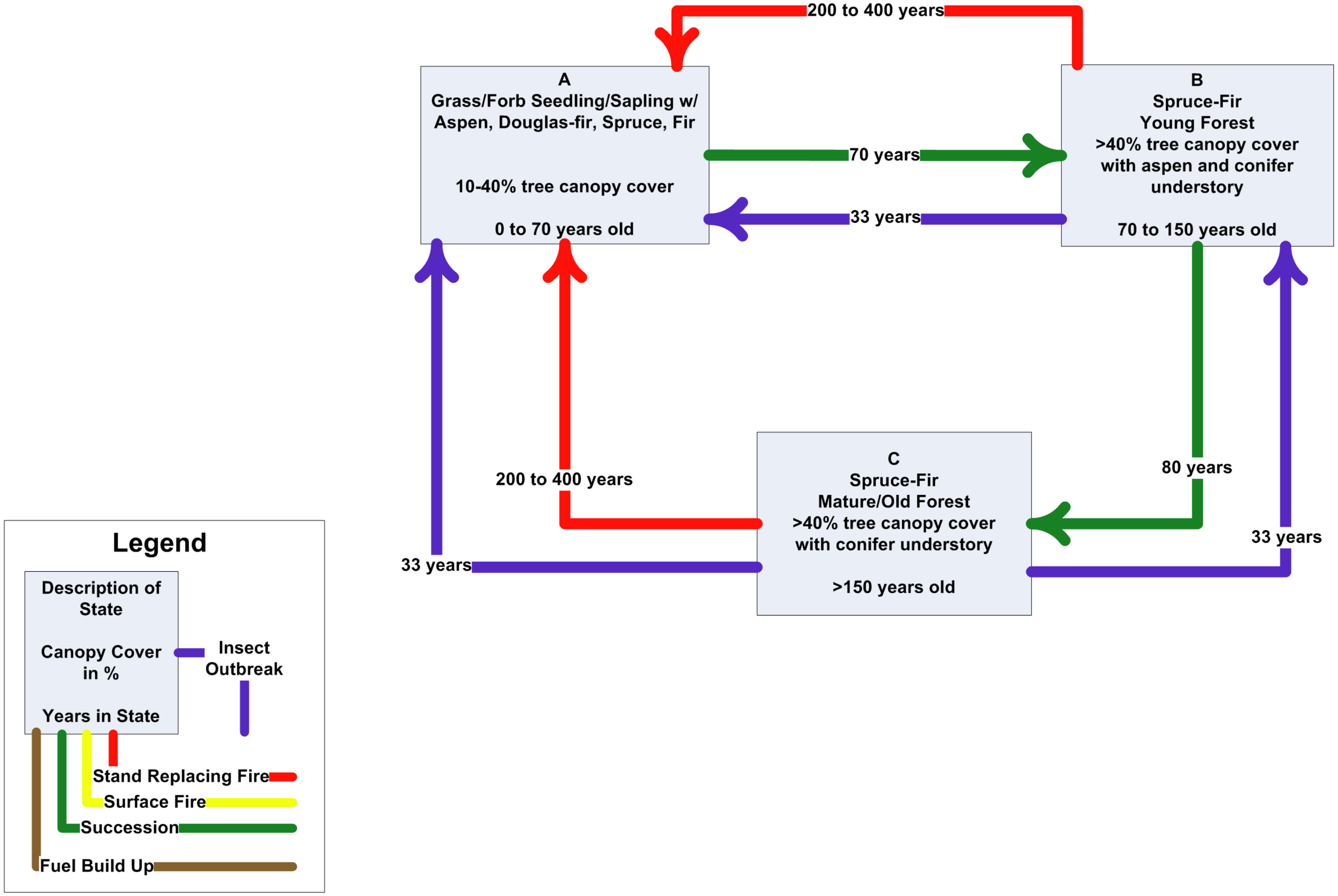

Figure 19-3. Conceptual Current state and transition model for both low and high elevation spruce-fir forest vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Model Parameters

In Table 19-1 we describe the parameters included or not included within the Historic VDDT model for spruce-fir low elevation, as well as the sources of information and any assumptions used to create model parameters. In Table 19-2 we describe the parameters included or not included within the Historic VDDT model for spruce-fir high elevation, as well as the sources of information and any assumptions used to create model parameters. Table 19-2 also describes the parameters for the Current VDDT model for both high and low elevation spruce-fir forest types, since both types are assumed to be within their historic range of variation, with the exception of the cessation of surface fire, which is the only difference between high and low elevation spruce-fir forest.

Table 19-1. Identification of Historic transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the VDDT model for spruce-fir low elevation with mixed severity fire.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Plant Growth	70 to 80 years between states	Transitions among model states were taken from data developed by Veblen and others (1994) and Vankat (2006).	We assume that transition from seedling/sapling to young forest takes approximately 70 years, and from young to old/mature forest takes approximately 80 years.
Regeneration from seed	Unknown, not used in model.	Seed production and seedling recruitment is highly variable in both space and time, and there is insufficient information to assign a probability for this transition across the entire region.	Due to the lack of data on seedling recruitment, this transition is not included in the model, and hence the proportion of this seedling/sapling class of vegetation is presumed to be underestimated in the model
Surface Fire	31 years	Fulé and others 2003.	These data are based on direct evidence (fire scar data). We used the 25% scarring filter (large fires) for modeling purposes. Because there was only one value for surface fire reported in the literature, we did not model minimum, maximum, and mean for surface fires.
Stand Replacing Fire	Once every 200 to 400 years	Stand replacing fire was reported to be pervasive but relatively infrequent (Dieterich 1983, Moir 1993, Fulé and others 2003, Vankat 2006).	Stand replacing fire occurred across all spruce-fir forests, returning the forest to the seedling/sapling stage.
Insect Outbreak	Once every 33 years for young forests; once every 100 years for	Spruce beetle outbreaks have been documented for the northern portion of the range (northern New Mexico, Kaibab Plateau) and for the Pinalenos in southern	We assumed that insect outbreaks occurred at a lower frequency for young forest than for old/mature forest, returning young forest to regeneration state “A”, and returning old/mature forest to equal proportions of

Transition Type	Transition Frequency or Length	Sources	Assumptions
	old/mature forests.	Arizona(Baker and Veblen 1990, Koprowski and others 2005).	regeneration and young forest.
Silvicultural Activities	Not used in model		Spruce-fir forests were too high in elevation, too inaccessible, or of too low value to have been significantly affected by timber harvest.

Table 19-2. Identification of Historic and Current transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the VDDT model for spruce-fir high elevation with stand replacing fire. A major assumption here is that Historic and Current conditions are sufficiently similar for the Spruce-Fir high elevation vegetation type that current forests are within the historic range of variability, because the time elapsed since the last crown fire is within the range of FRI. Current conditions for the spruce-fir wet forest vegetation type were assumed to reflect the current conditions of the dry spruce-fir forest type because the surface fire regime, which is the only difference between the two (from a disturbance standpoint), has been extinguished.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Plant Growth	70 to 80 years between states	Transitions among model states were taken from data developed by Veblen and others (1994) and Vankat (2006).	We assume that transition from seedling/sapling to young forest takes approximately 70 years, and from young to old/mature forest takes approximately 80 years.
Regeneration from seed	Unknown, not used in model.	Seed production and seedling recruitment is highly variable in both space and time, and there is insufficient information to assign a probability for this transition across the entire region.	Due to the lack of data on seedling recruitment, this transition is not included in the model, and hence the proportion of this seedling/sapling class of vegetation is presumed to be underestimated in the model
Stand Replacing Fire	Once every 200 to 400 years	Stand replacing fire was reported to occur in small patches throughout spruce-fir's range, but relatively infrequent (Dieterich 1983, Moir 1993, Fulé and others 2003, Vankat 2006).	Stand replacing fire occurred in small patches across all spruce-fir forests, returning the forest to the seedling/sapling stage.
Insect Outbreak	Once every 33 years for young	Spruce beetle outbreaks have been documented for the northern portion of the range (northern New Mexico, Kaibab	We assumed that insect outbreaks occurred at a lower frequency for young forest than for old/mature forest, returning young forest to regeneration state "A", and returning

	forests; once every 100 years for old/mature forests.	Plateau) and for the Pinalenos in southern Arizona(Baker and Veblen 1990, Koprowski and others 2005).	old/mature forest to equal proportions of regeneration and young forest.
Silvicultural Activities	Not used in model		Spruce-fir forests were too high in elevation, too inaccessible, or of too low value to have been significantly affected by timber harvest.

19.3 Results – Results of the historic spruce-fir models indicate a small amount of variability in the 900-year average for each state based on the fire interval range (Table 19-3 and 19-4). For both the high and low elevation forest types, as stand replacing fire decreases in frequency the proportion of the landscape in the youngest state decreases from 31.4% to 21.1% (low elevation), and from 31.1% to 20.7% for the high elevation model. Conversely, for both models, as stand replacing fire decreases in frequency from once every 200 years to once every 400 years, the proportion of the landscape in the oldest age class (C) increases from 32.1% to 44.5% (low elevation), and 32.5% to 45.0% (high elevation). Although the two different models differ in having a mixed fire regime (low elevation) and a stand replacing only fire regime (high elevation), the model outputs are very similar. Although there are small differences between the two models and among the different fire return intervals, all model outputs portray a landscape that has fairly even proportions of the different model states (total range = 18.8% to 47.2%).

Results for the current spruce-fir low and high elevation models (Table 19-5), which were run for 120 years following development of the Historic conditions, were very similar to results from the Historic models. For example, under a stand replacing fire return interval of 300 years, average landscape proportion of the historic low elevation spruce-fir was 39.8%, for high elevation spruce-fir it was 40.5%, and for the current model for both high and low elevation spruce-fir it was 45.4%.

Table 19-3. Results for the Historic spruce-fir low elevation VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (300 years), maximum (400 years), and minimum (200 years) of the estimated fire return interval range for stand replacing fire.

Stand Replacing Fire Return Interval (FRI) Modeled	Model Output	Grass/ Seedling & Sapling A Open	Young Forest B Open to Closed	Old/Mature Forest C Multi-storied
Every 200 years	Average	31.4	36.6	32.1
	Minimum	29.1	34.1	29.8
	Maximum	33.7	39.1	34.3
	Standard Deviation	1.4	1.5	1.4
Every 300 years	Average	24.9	35.3	39.8
	Minimum	23.1	33.2	37.5
	Maximum	26.9	37.3	42.1
	Standard Deviation	1.2	1.3	1.4
Every 400 years	Average	21.1	34.4	44.5
	Minimum	19.3	32.1	42.2
	Maximum	23.1	36.8	47.1
	Standard Deviation	1.2	1.4	1.5

Table 19-4. Results of the Historic spruce-fir high elevation forest VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (300 years), maximum (400 years), and minimum (200 years) of the estimated fire return interval range for stand replacing fire.

Stand Replacing Fire Return Interval (FRI) Modeled	**Model Output**	**Grass/ Seedling & Sapling A Open**	**Young Forest B Open to Closed**	**Old/Mature Forest C Multi-storied**
Every 200 years	Average	31.1	36.4	32.5
	Minimum	28.8	33.9	30.2
	Maximum	33.5	38.8	34.9
	Standard Deviation	1.4	1.5	1.4
Every 300 years	Average	24.5	35.0	40.5
	Minimum	22.5	32.8	38.1
	Maximum	26.5	37.3	42.7
	Standard Deviation	1.2	1.4	1.4
Every 400 years	Average	20.7	34.3	45.0
	Minimum	18.8	32.0	42.6
	Maximum	22.5	36.6	47.2
	Standard Deviation	1.1	1.4	1.4

Table 19-5. Results of the Current spruce-fir high and low elevation forest VDDT model, reported as the 120 year end value for average, minimum, maximum, and average standard deviation of the percent of the modeled landscape in each state.

Stand Replacing Fire Return Interval (FRI) Modeled	**Model Output**	**Grass/ Seedling & Sapling** **A** **Open**	**Young Forest** **B** **Open to Closed**	**Old/Mature Forest** **C** **Multi-storied**
Every 200 years	Average	30.6	36.7	32.7
	Minimum	28.8	35.1	31.5
	Maximum	32.2	38.2	33.8
	Standard Deviation	1.0	0.9	0.7
Every 300 years	Average	21.1	32.5	45.4
	Minimum	19.4	30.9	43.8
	Maximum	22.8	34.6	47.5
	Standard Deviation	1.0	1.1	1.1
Every 400 years	Average	21.1	33.5	45.4
	Minimum	19.4	31.9	43.8
	Maximum	22.8	35.8	47.5
	Standard Deviation	1.0	1.1	1.1

19.4 Discussion – These modeled scenarios indicate that although there may be large differences in canopy and understory composition due to differences in elevation and moisture regimes, the resulting difference in disturbance regime (surface plus canopy versus strictly canopy fire) has a minor effect on the relative proportion of the landscape in the three model states (regeneration, young, and old forest). Also, elimination of a relatively long-rotation surface fire regime (31 years) has had minimal effect on the relative proportion of model states over 120 years. This is in concurrence with other authors' contention that spruce-fir forest conditions in many areas of the Southwest may be within its historic range of variation, and that insufficient time has elapsed since the last crown fire for us to have experienced crown fire in contemporary time (Swetnam and others 2005, Koprowski and others 2005, Vankat 2006).

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