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

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Interior Chaparral

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Madrean Pine-Oak

Schussman, Heather and Dave Gori. 2006. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Madrean Pine-Oak of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 35 pp.

Mixed Conifer

Smith, Ed. 2006. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Mixed Conifer of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 31 pp.

Ponderosa Pine

Smith, Ed. 2006. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Ponderosa Pine of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 43 pp.

Spruce-Fir

Smith, Ed. 2006. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Spruce-Fir of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 37 pp.

Aspen

Smith, Ed. 2006. Historical Range of Variation for Aspen of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 21 pp.

Alpine Tundra

Smith, Ed. 2006. Historical Range of Variation for Alpine Tundra of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 15 pp.

Montane Grassland

Smith, Ed. 2007. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Montane Grassland of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 43 pp.

Pinyon-Juniper

Gori, Dave and Joanna Bate. 2007. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Pinyon-Juniper of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 141 pp.

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Guy R. McPherson, Professor – Dept. of Natural Resources, Univ. of Arizona
Jennifer Ruyle, Forest Planner – United States Forest Service, Coronado National Forest
Wayne A. Robbie, Supervisory Soil Scientist – Regional Office, USFS, Region 3

<u>Madrean Encinal:</u>
External reviews have been solicited but have not been received at this time.

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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.

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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.

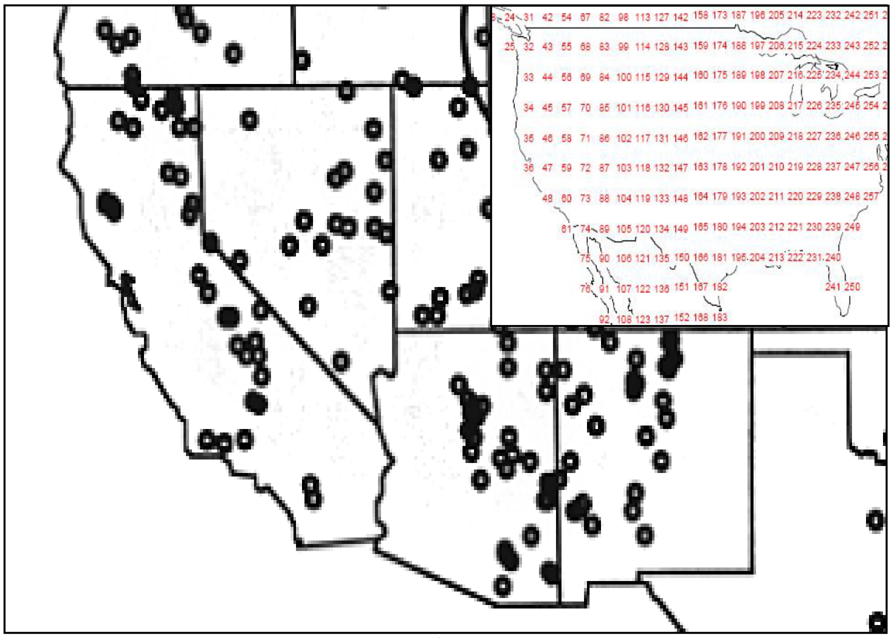
	88	**89**	**104**	**105**	**119**	**120**	**133**	**134**
# of Tree Chronologies	8-9	5-9	8-9	5-9	9	6-9	8-9	5-9

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.

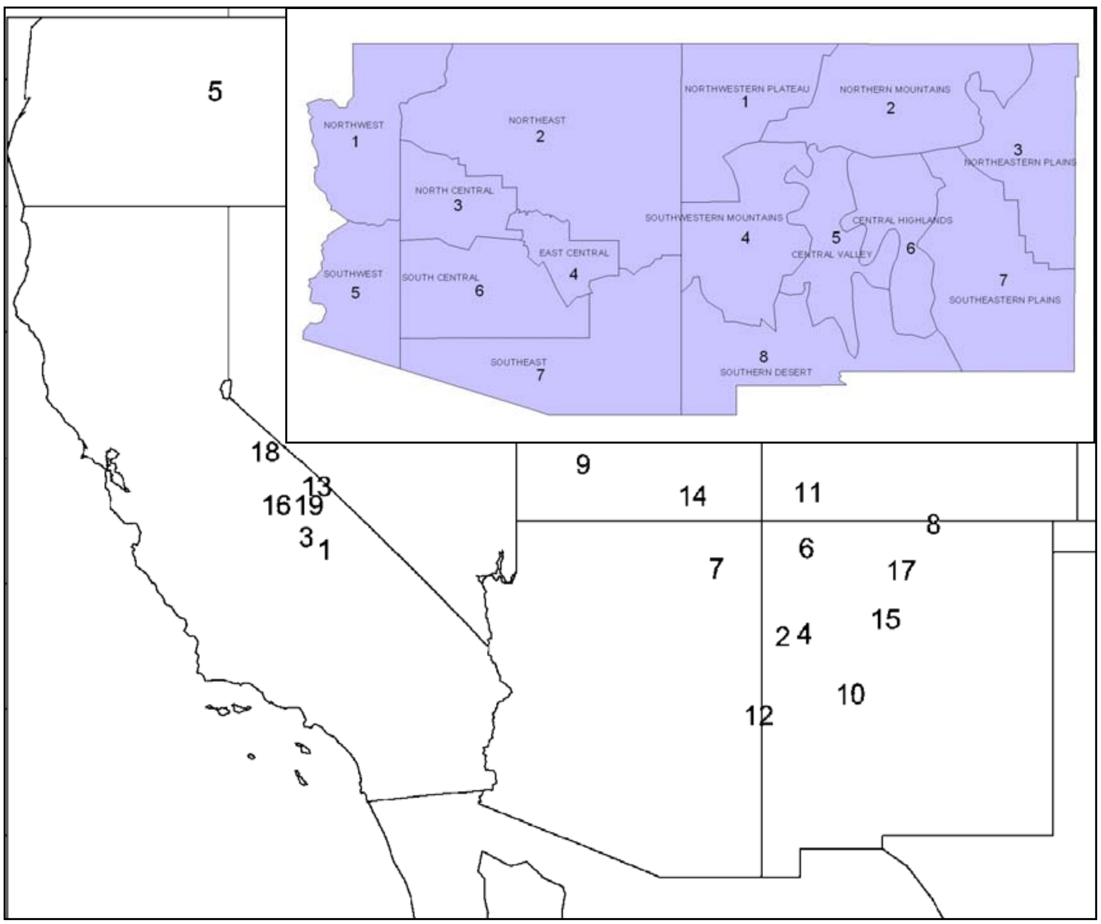


Figure 1-1. Identification of tree chronology locations for both the PDSI (1a taken from Cook and others 1999) and winter precipitation (1b taken from Ni and others 2002) data sets, as well as PDSI grid point locations and climate division boundaries.

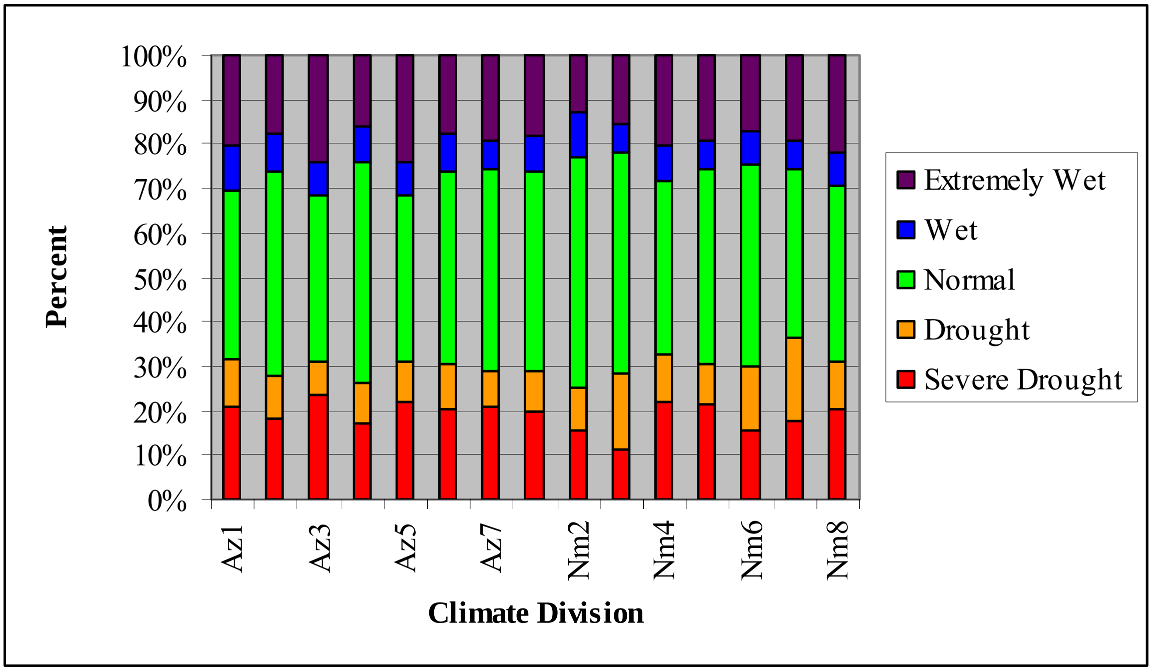


Figure 1-2. Comparison of the percent of years in all year types for all climate divisions in the Southwest.

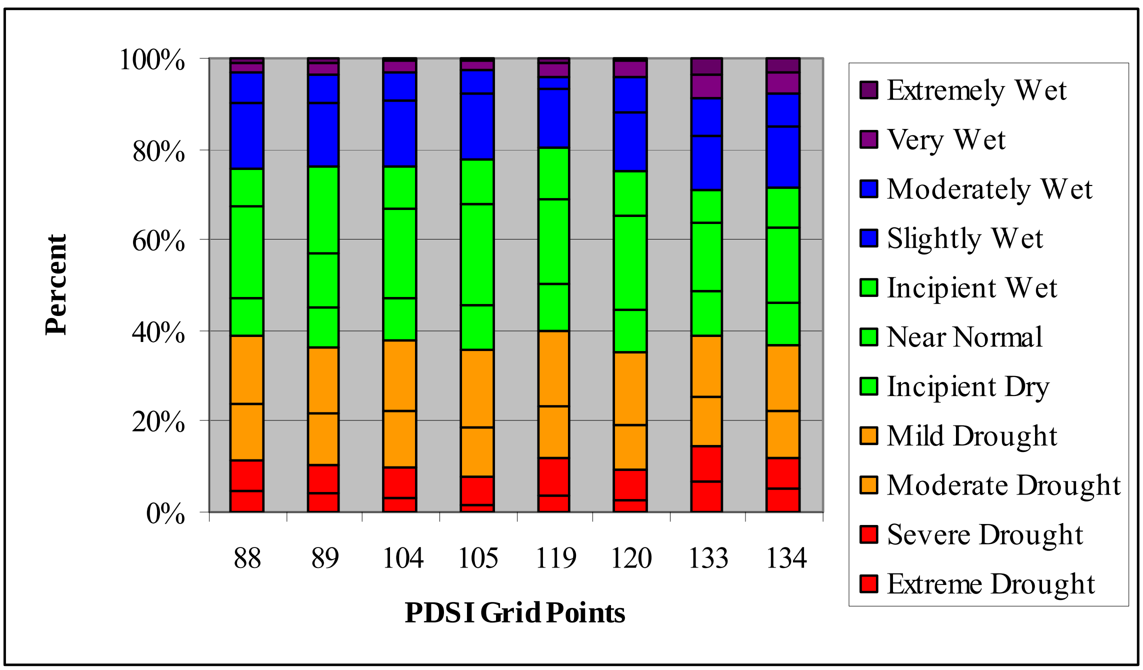


Figure 1-3. Comparison of the percent of years in all year types for all PDSI grid locations in the Southwest.

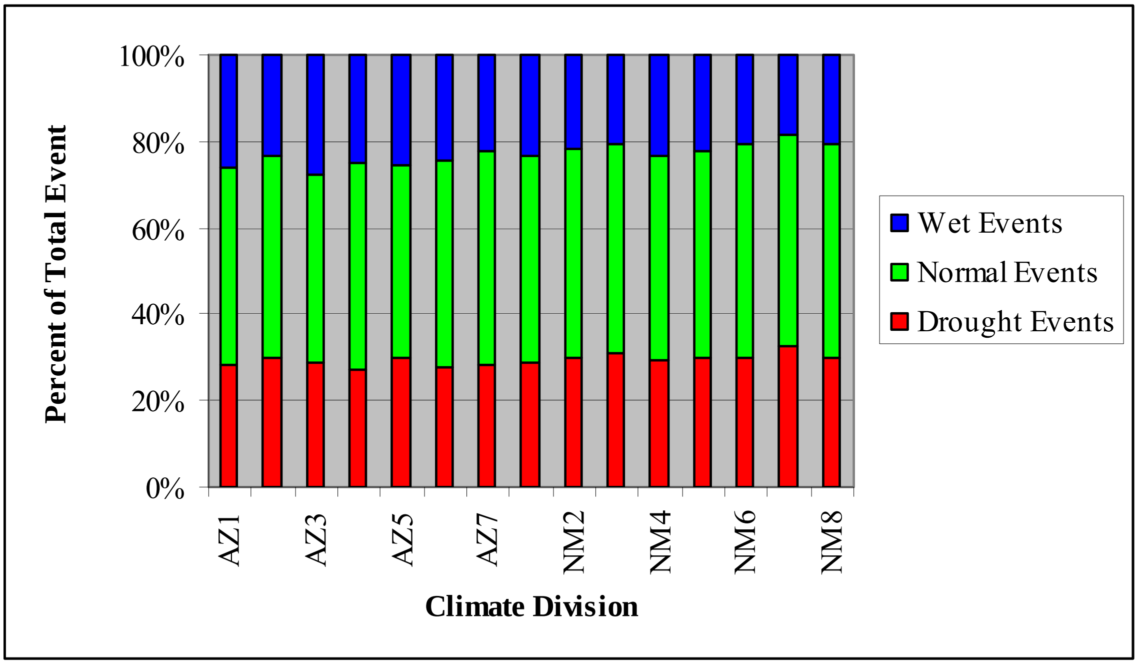


Figure 1-4. Comparison of the percent of events classified as drought, normal, and wet events for all climate divisions in the Southwest.

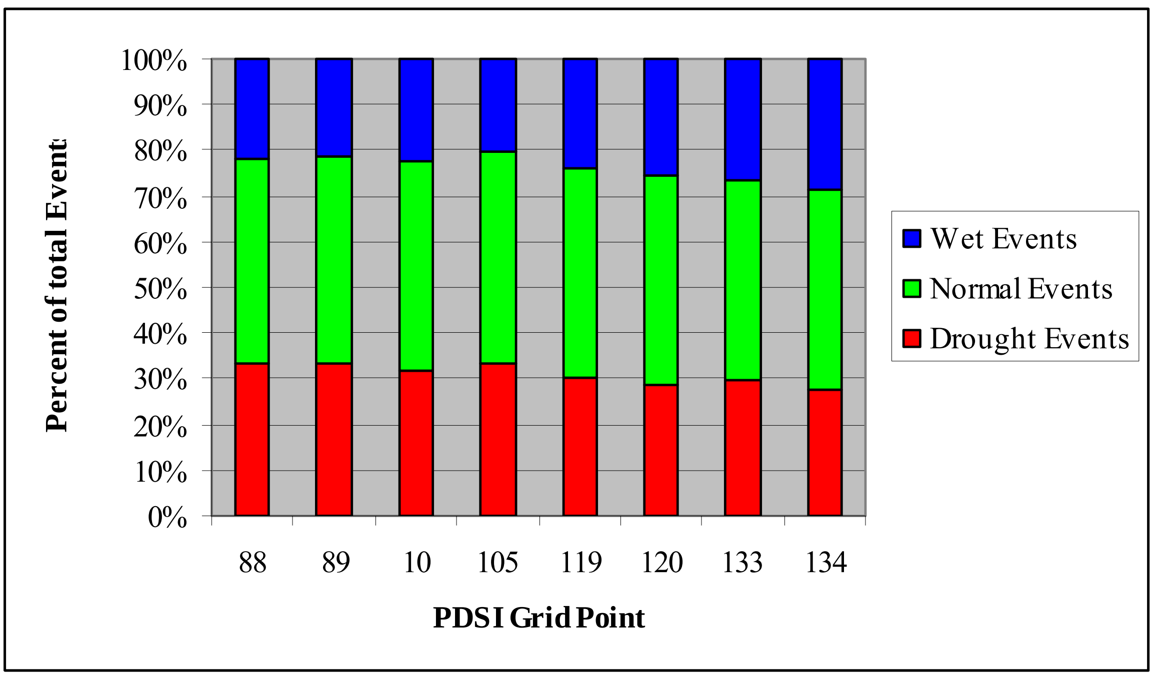


Figure 1-5. Comparison of the percent of events classified as drought, normal, and wet events for all PDSI grid locations in the Southwest.

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 11160, 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 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).

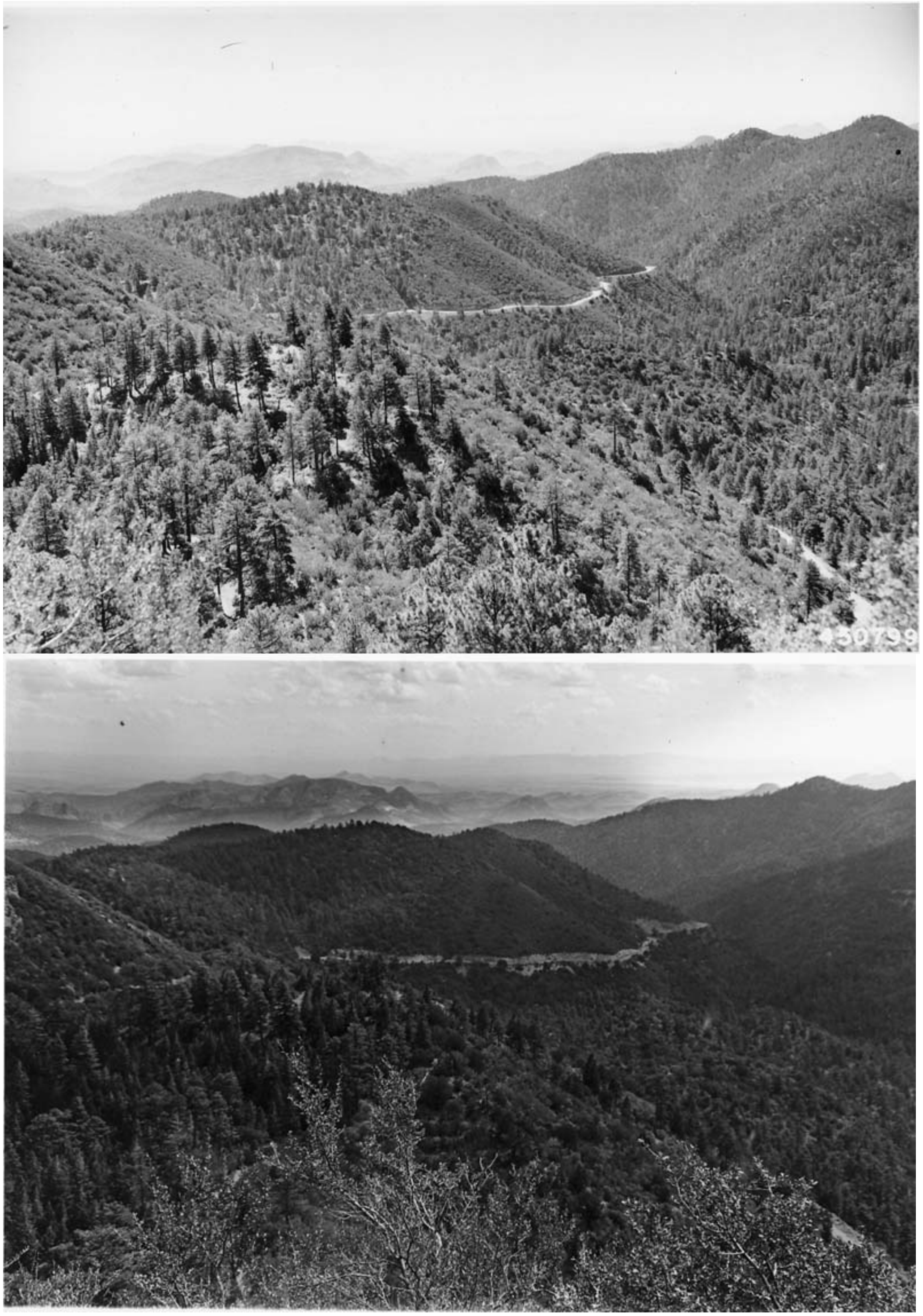

Figure 6-2. Photographic comparison from the Gila National Forest, top photo from 1948, and bottom photo from 1984. These images depict an increase in tree density, especially in the understory over 36 years. Photos courtesy USDA Forest Service Region 3 Office.[Top image15a, #450799. Looking east from 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 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.

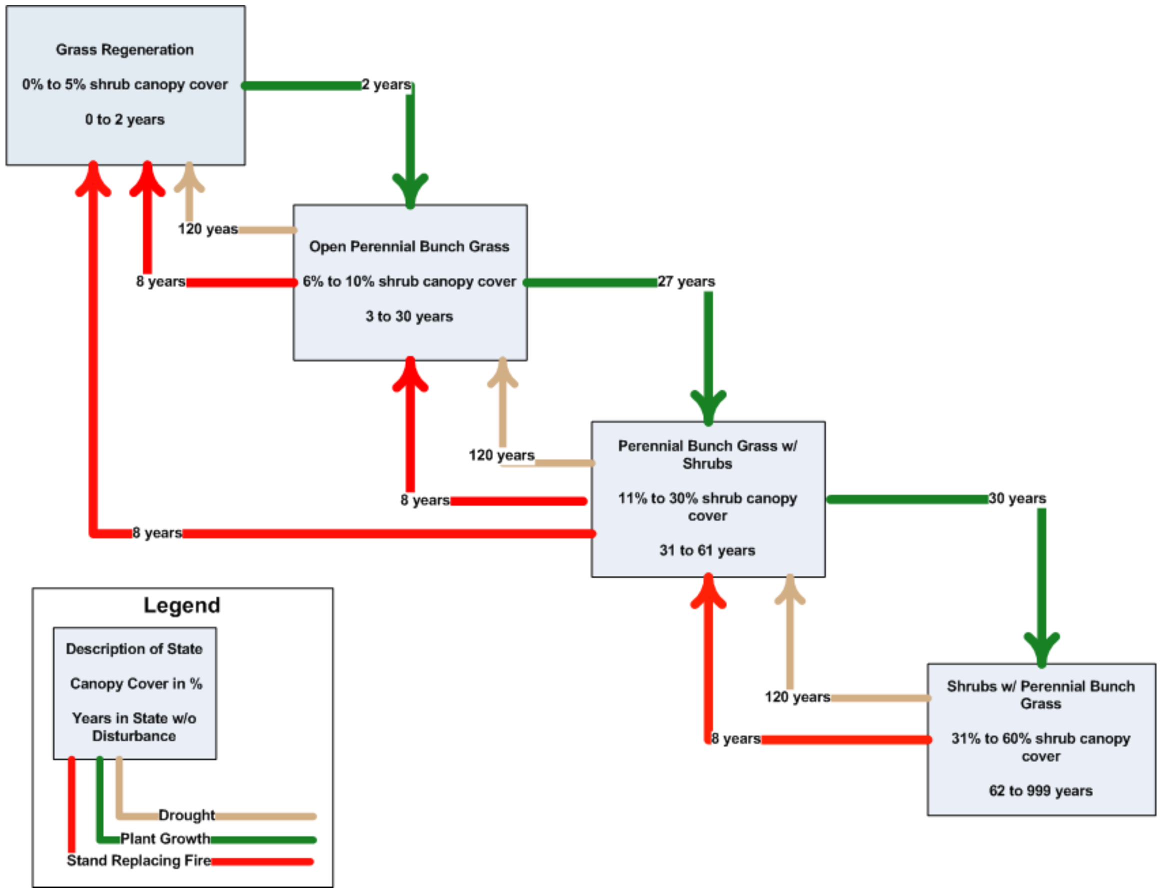


Figure 13-1. Simple grassland model used in sensitivity testing of VDDT software

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

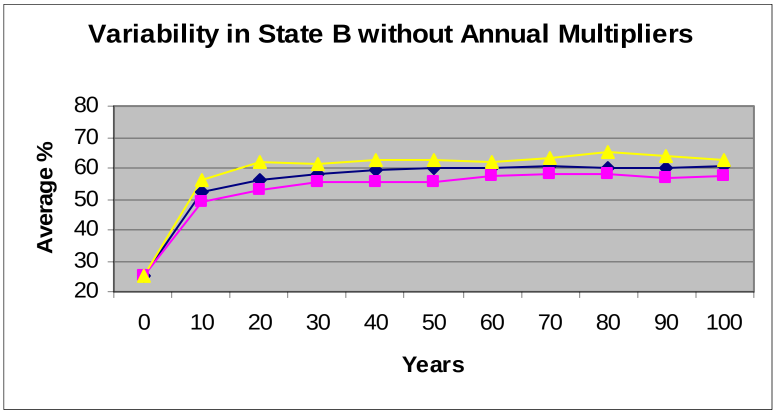
We ran the historic models for 1000 years, as this temporal span corresponds with the widest frame of reference offered by the scientific literature. Additionally, 1000 year long runs allowed for infrequent transitions, such as stand replacing fires in the spruce fir PNVT and extreme drought events in 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.



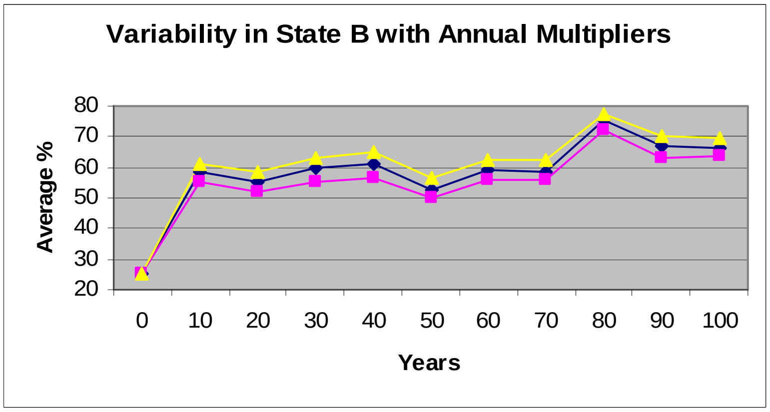


Figure 13-2. Comparison of year to year variability in state B of the simple grassland VDDT model with and without the use of annual multipliers. Maximum values in yellow, average values in blue, and minimum values in pink.

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 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).

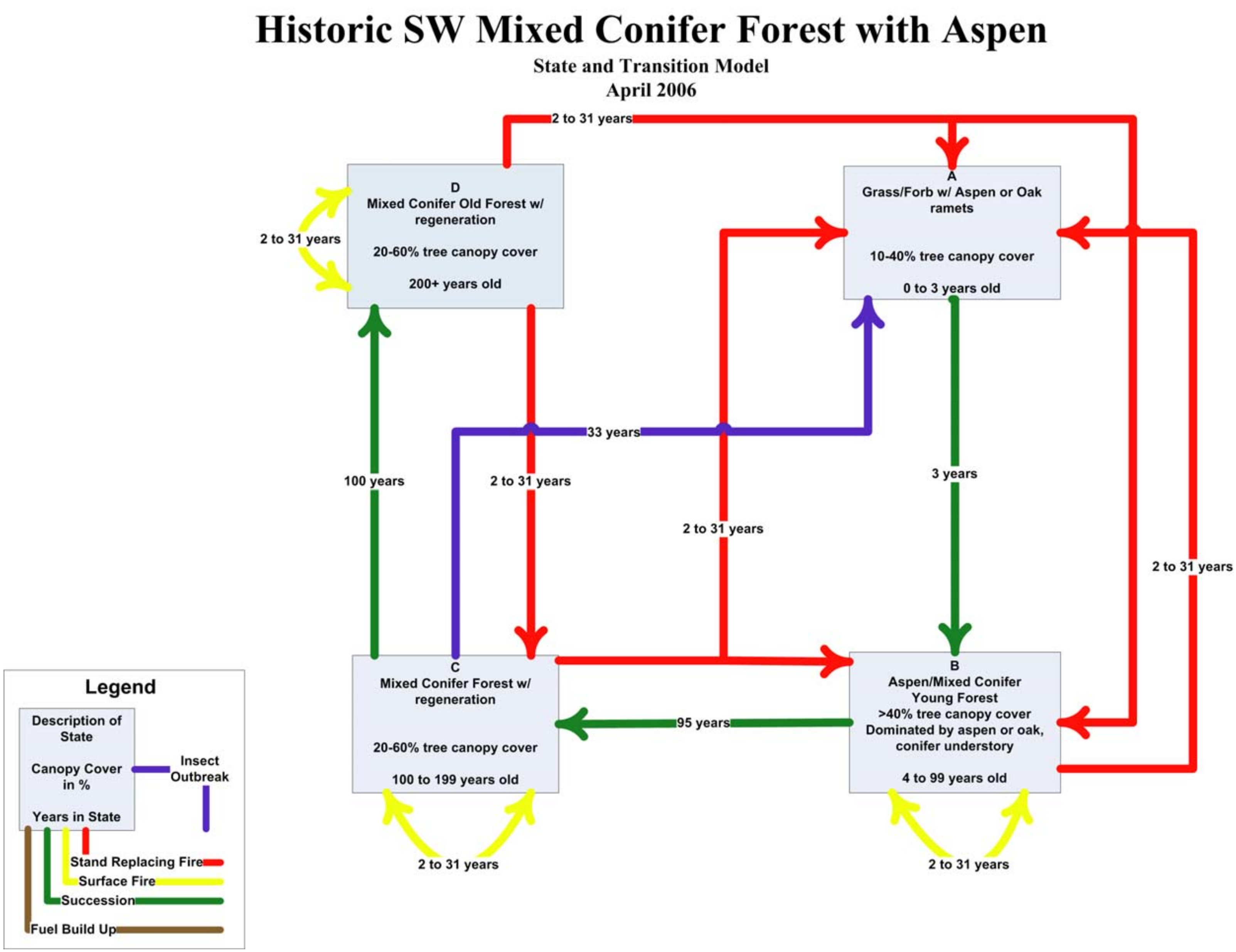

Figure 37-1. Conceptual Historic state and transition model for the mixed conifer with aspen vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

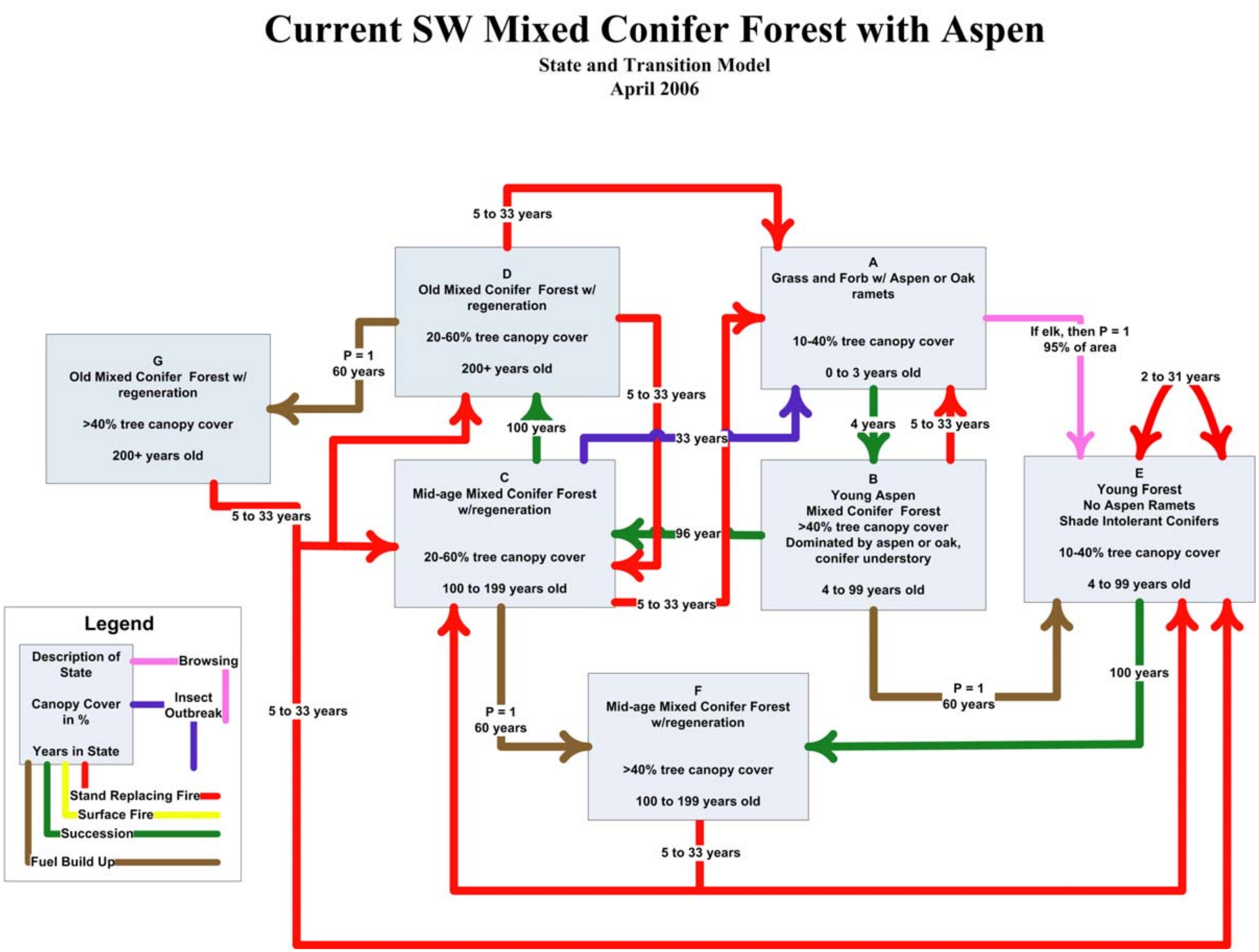

Figure 17-2. Conceptual Current state and transition model for the mixed conifer with aspen vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation. 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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