

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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Aspen

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Chapter 1 - Historical Range of Variation for Potential Natural Vegetation Types of the Southwest

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 PNVNT description, we have identified reference sites that were used for developing its HRV.

1.2 Methods Used in Determining HRV

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

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

fluctuation, competition, and insect outbreak. Every year, fire, silvicultural practices, and decomposition remove more of the available record.

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

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

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

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

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

Photographic Archive	Location of Archive	Contact Person	Repeat Photographs Collected	PNVTs for which photographs were obtained for	Additional Comments
Apache-Sitgreaves National Forest	Springerville, AZ	Bob Dyson	No	aspen, interior chaparral, mixed conifer, montane grasslands, pinyon-juniper, riparian, spruce-fir	The photographs came from the A-S historic archives, and were sent on a CD. The CD included about 500 photographs, although none of the photographs have information regarding dates taken or the specific locations of the photographs.
Carson National Forest	Taos, NM	Bill Westbury and Dave Johnson	No	aspen, mixed conifer, montane grassland, riparian, spruce-fir	

Coronado National Forest	Tucson, AZ	Bill Gillespie and Geoff Soroka	No	aspen, interior chaparral, Madrean encinal, Madrean pin-oak, mixed conifer, pinyon-juniper, semi-desert grasslands	Two sources were used. One was from Bill Gillespie, and included only historical photos. The other source was from Geoff Soroka, where most photos were taken in part to ground-truth the mid-scale vegetation mapping effort.
Ecological Restoration Institute	Northern Arizona University	Dennis Lund	No	aspen, mixed conifer, pinyon-juniper, ponderosa pine	photos from Dennis's collection from national and local USFS archives
Gila National Forest	Silver City, NM	Reese Lolly	No	interior chaparral, mixed conifer, pinyon-juniper, ponderosa pine	
<i>'Historic increases in woody vegetation in Lincoln County, New Mexico'</i> 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	
<i>The changing mile revisited</i> 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 " <i>Wood plenty, grass good, water none</i> " 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 PNV 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 PNVs 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 Kipfmüller 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 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
R² (%)	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.

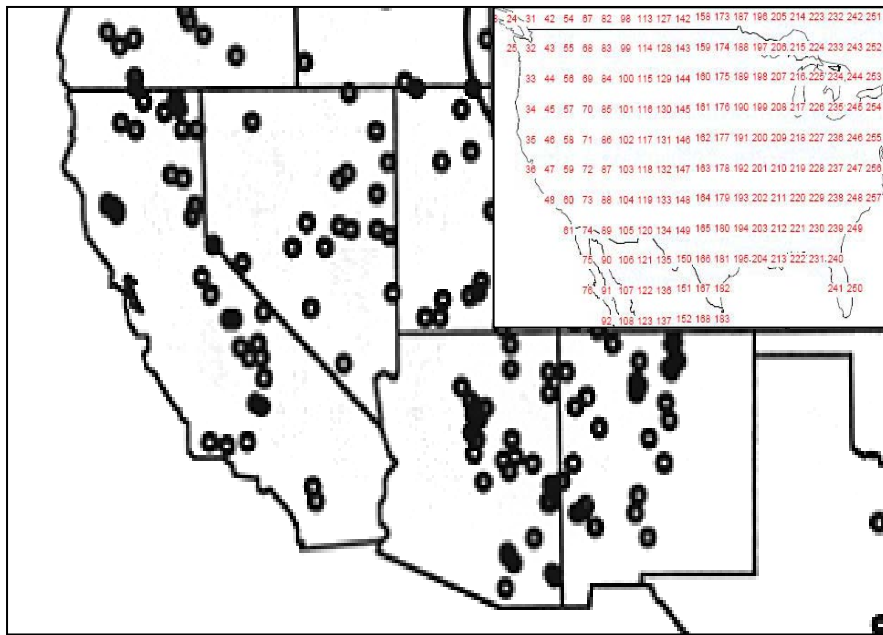
	88	89	104	105	119	120	133	134
# of Tree Chronologies	8-9	5-9	8-9	5-9	9	6-9	8-9	5-9

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).

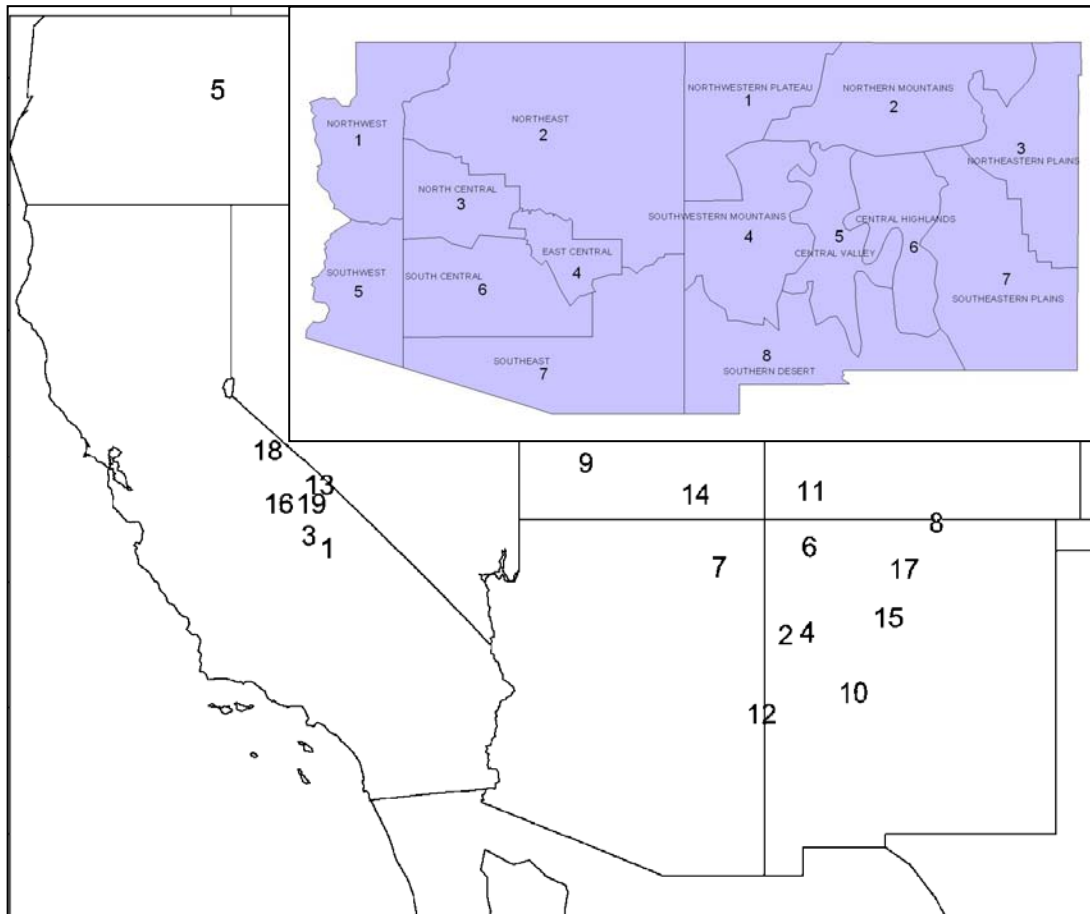
Results - A comparison of the percent of dry and wet winter precipitation years, for the 15 climate divisions that span Arizona and New Mexico, showed a pattern of 19% of the years, between year 1000 and 1988, classified as severe drought or extremely wet years, 11% classified as drought years, 8% classified 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 or summer PDSI data sets (Figure 1-4, Figure 1-5, Appendix 2 - Tables 2.1 and 2.2 and Figures 2.1 to 2.23). Specifically, there were on average 52 drought events, 41 wet events, and 85 normal events identified for the winter precipitation data and 71 drought events, 54 wet events, and 104 normal events identified for the summer PDSI data set. In contrast, the range of the length of events does exhibit some regional variability with winter precipitation events ranging between 9 and 26 years for the longest drought events, between 14 and 23 years for the longest wet events, and between 19 and 40 years for the longest normal events. This level of variability is also seen in the summer PDSI data set with between 19 and 25 years for the longest drought event, between 8 and 17 years for the longest wet events, and between 14 and 23 years for the longest normal events (Appendix 2 - Table 2.1 and Figures 2.1 – 2.23). The timing of the events identified is fairly consistent across the entire Southwest (ie all climate divisions and PDSI grid point locations document drought and wet events occurring in roughly the same years even though the magnitude of those events varies regionally).



1a.



1b.

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

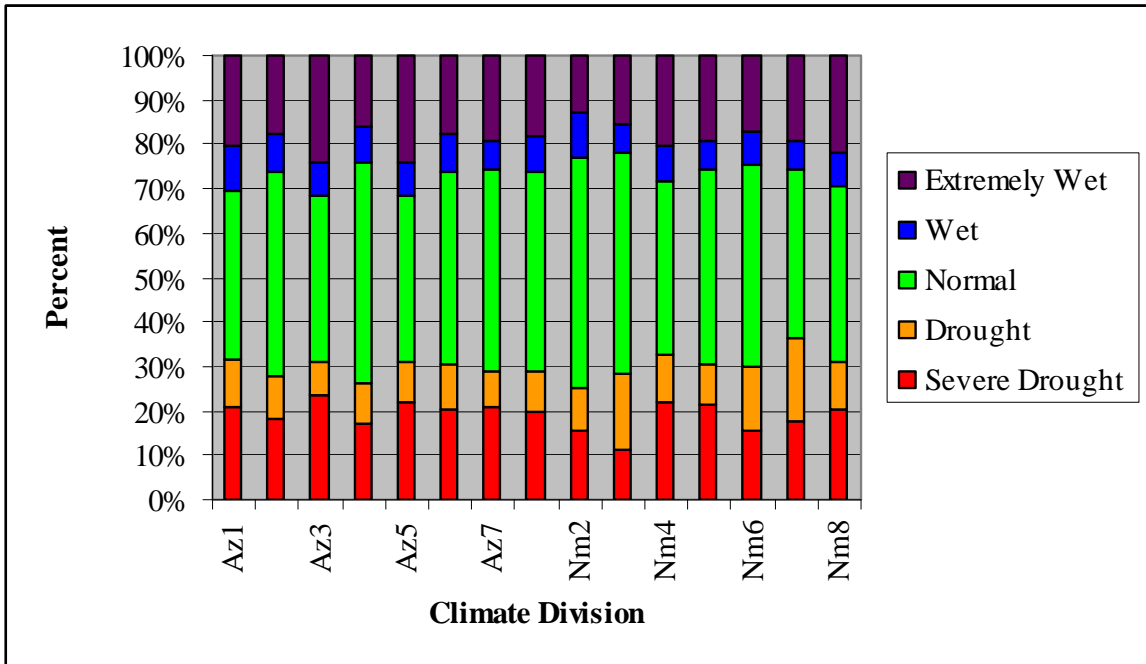


Figure 1-2. Comparison of the percent of years in all year types for all climate divisions in the Southwest.

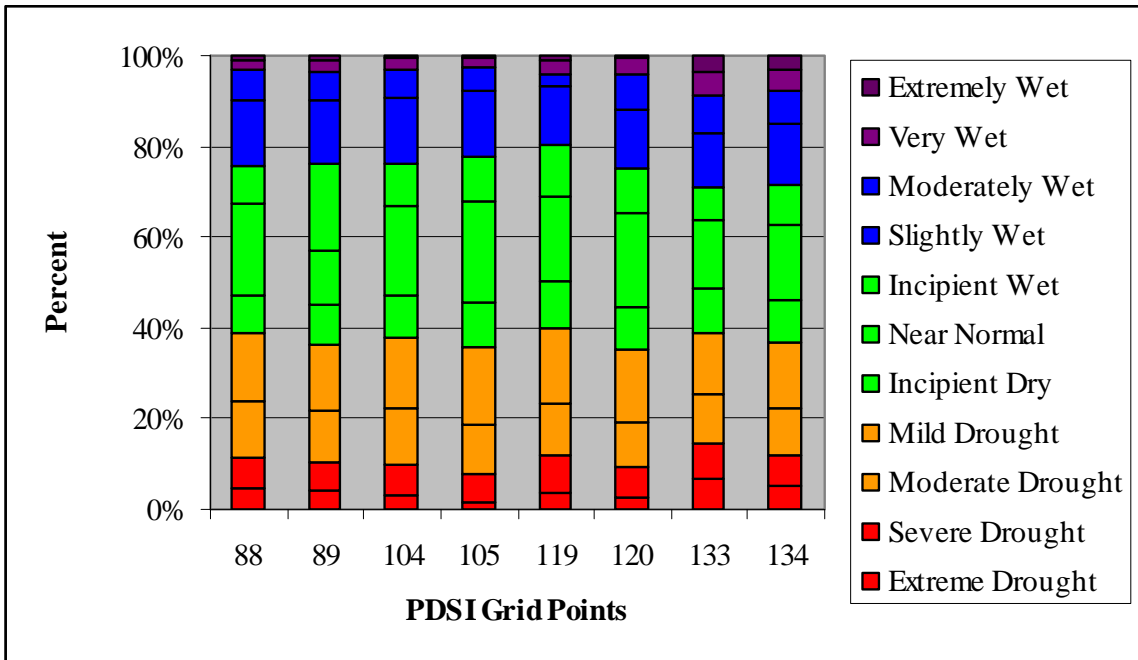


Figure 1-3. Comparison of the percent of years in all year types for all PDSI grid locations in the Southwest.

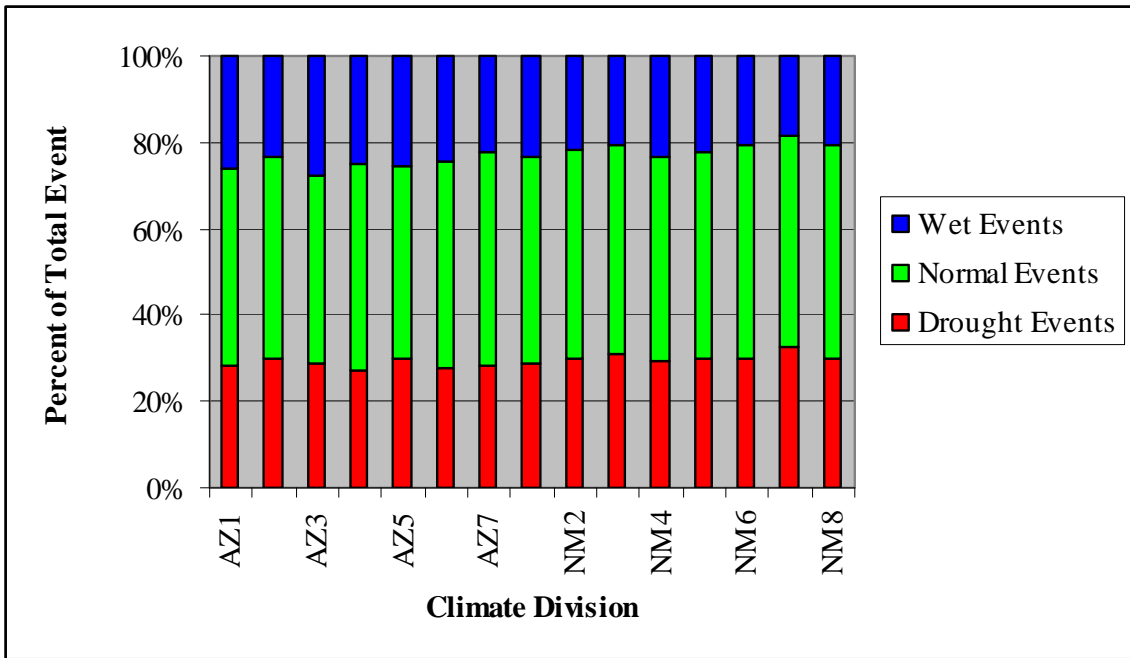


Figure 1-4. Comparison of the percent of events classified as drought, normal, and wet events for all climate divisions in the Southwest.

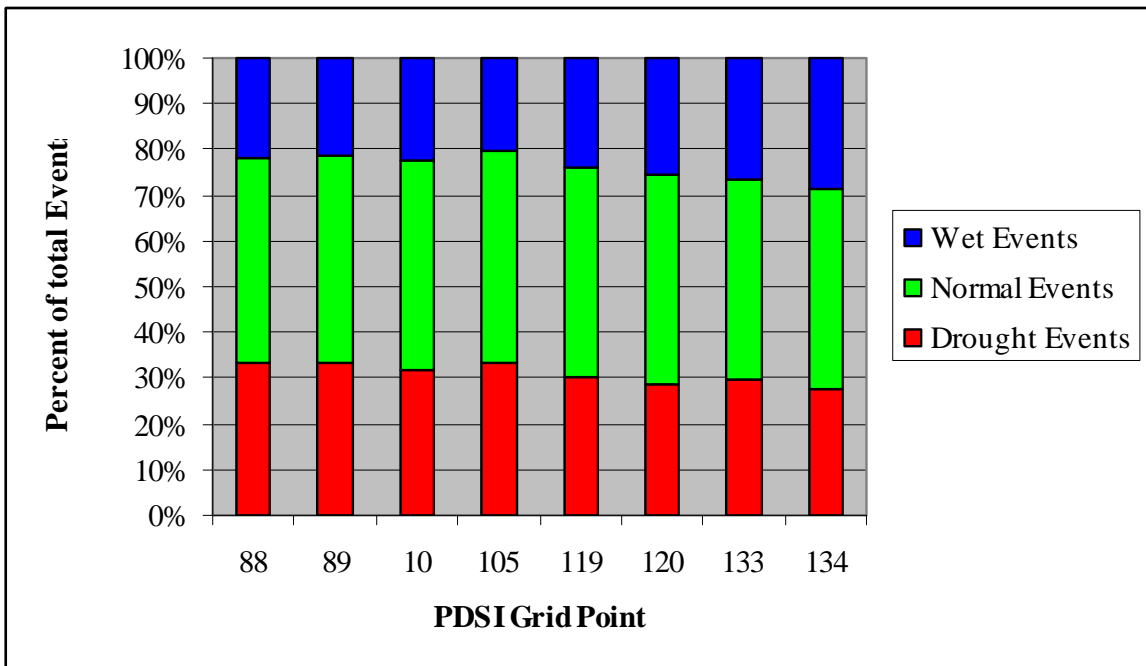


Figure 1-5. Comparison of the percent of events classified as drought, normal, and wet events for all PDSI grid locations in the Southwest.

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

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

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

megadrought has been documented to have coincided with the abandonment of “a dozen” pueblos in New Mexico (Stahle and others 2000).

Comparison of the pattern of dry and wet events for specific climate division with PNVTS 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; IPCC 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 8 - Spruce-Fir Forest

8.1 General Description

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

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

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

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

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

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

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

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

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

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

8.2 Historical Range of Variation of Ecological Processes

Vegetation Dynamics – The spruce-fir forest has been studied very little in the Southwest, probably due to its limited distribution and lack of commercial timber value. Spruce-fir forests are much more extensive in the Rocky Mountains of Colorado, Montana, and Wyoming, and there is a substantial literature base for the more northern occurrences of this forest type. However, most of the central and northern Rocky Mountain spruce-fir

forests probably have a very different evolutionary history including disturbance regimes due to major differences in patch size, soils, latitude, climate, and forest community composition, especially with the inclusion of lodgepole pine (*Pinus contorta*), which acts as an early seral species for the central and northern Rocky Mountain forests, but does not occur in the Southwest. Thus while some of the northern Rocky Mountain studies are included in this report for comparison, generally those studies from outside the Southwest are not considered to be applicable to Southwest spruce-fir forests.

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

Disturbance Processes and Regimes

Climate- See Introduction section of the HRV document.

Fire- Both Engelmann spruce and corkbark fir are fire sensitive due to thin bark at all ages, and hence are unlikely to survive even low intensity fires and provide fire scars for analysis (Veblen and others 1994). Thus, most of the fire regime research has been accomplished using current tree and stand age or stand structure analysis (White and Vankat 1993, Fulé and others 2003), as well as fire scar analysis of adjacent forest types, which for spruce-fir is primarily mixed conifer forests at lower elevation (Grissino-Mayer and others 1995, Baisan 1995, Swetnam and others 2005). For the central Rocky Mountains of Colorado, spruce-fir forests burn as crown fires and at return intervals of centuries (Schoennagle and others 2004), although at lower elevation, some areas have experienced mixed severity and surface fires (Baker and Veblen 1990). Some have suggested that spruce-fir forests in the Southwest experience crown fire, with insufficient time having passed since the last crown fire for these forests to have experienced crown fire in contemporary time, at least for the small patch of spruce-fir forest on Mt. Graham

in the Pinaleno Mountains (Swetnam and others 2005). Grissino-Mayer and others' (1995) analysis of stand structure in the Pinalenos led them to conclude that the fire return interval (FRI) for spruce-fir forest was 300-400 years. However, there is ample evidence to suggest that some spruce-fir forests in Arizona and New Mexico have a mixed severity fire regime that burned with a return interval on the order of decades rather than centuries (Dieterich 1983, Moir 1993, Fulé and others 2003, Vankat 2006).

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

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

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

into portions of this forest from adjacent mixed-conifer, and high severity, small-patch size (individual trees or groups) events probably also occurred in this zone” (Swetnam and others 2005).

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

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

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

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

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

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

Windthrow – Wind has been documented as an important ecological factor in spruce-fir forests of Colorado, where windthrow or blow-down (trees knocked over by wind events)

may affect up to 92% of old-growth areas (Alexander 1987, Veblen and others 1991). As mentioned above, spruce beetle populations may be maintained in small windthrow areas (Veblen and others 1992). We found no such similar studies for the Southwest.

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



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



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

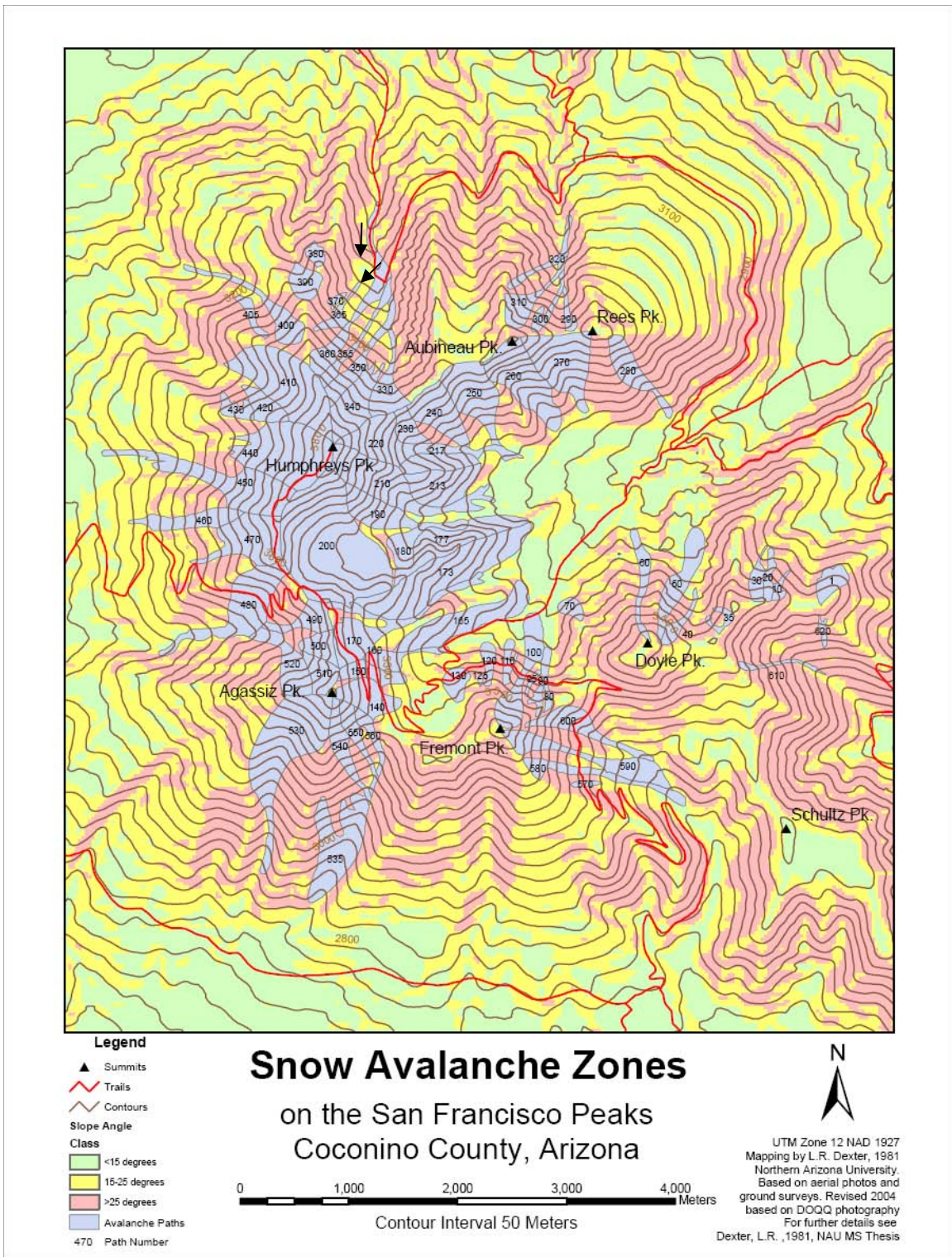


Figure 8-3. Map of snow avalanches (in purple) on San Francisco Peaks near Flagstaff, AZ. Most of these avalanches originate in tundra above treeline, pass through the spruce-fir forest and into the mixed conifer forest. Arrows indicate approximate location and direction taken for photos in Figures 8-1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

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

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

8.3 Historical Range of Variation of Vegetation Composition and Structure

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

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

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

Table 8-2. Historic forest structure reconstructed for two sites (GCNP=Grand Canyon National Park in 1880, SFPA=San Francisco Peaks in 1876) in Arizona. Basal area (BA) is expressed both in square ft. per acre (ft²/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 engelmannii*)+blue spruce (*Picea pungens*), PIPO=ponderosa pine (*Pinus ponderosa*), POTR=aspen (*Populus tremuloides*), PSME=Douglas-fir (*Pseudotsuga menziesii*), RONE=New Mexican locust (*Robinia neomexicana*), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (*Pinus aristata*), PIFL=limber pine (*Pinus flexilis*).

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

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

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

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

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

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

Structure Class (Size Class)-

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

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

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

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

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

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

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

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

8.4 Anthropogenic Disturbance (or Disturbance Exclusion)

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

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

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

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

Mining –

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

Fire Management –

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

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

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

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

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

8.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

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

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

Table 8-3. Current forest structure determined for two sites (GCNP=Grand Canyon National Park, SFPA=San Francisco Peaks) in Arizona. Basal area (BA) is expressed both in square ft. per acre (ft²/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 engelmannii*)+blue spruce (*Picea pungens*), PIPO=ponderosa pine (*Pinus ponderosa*), POTR=aspen (*Populus tremuloides*), PSME=Douglas-fir (*Pseudotsuga menziesii*), RONE=New Mexican locust (*Robinia neomexicana*), ABIES=white fir+corkbark fir, PIAR=bristlecone pine (*Pinus aristata*), PIFL=limber pine (*Pinus flexilis*). Trees are defined as stems having dbh > 1 inch, and regeneration as stems having dbh ≤ 1 inch.

Fulé and others (2003) censused spruce-fir forests at Grand Canyon National Park’s north rim and determined a current-era basal area of 121.1 ft²/acre (standard error=7.8), a tree density of 383.0 trees/acre (s.e.=40.12) for trees >1 inch (dbh), and a tree density of 3066.0 trees/acre for trees < 1 inch dbh. Cocke and others (2005) censused spruce-fir forests on the San Francisco Peaks and determined current-era basal area of 251.7 ft²/acre (s.e.=18.3), a tree density of 340.2 trees/acre (s.e.=55.9) for trees >1 inch (dbh), and a tree density of 309.1 trees/acre for trees < 1 inch dbh. (Table 9-3).

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

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

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

and Abolt (1997) to determine current patch structure, structure class, age structure, and patch dispersion.

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

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

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

Density - For the GCNP North Rim site, the spruce-fir forest has increased from 60.6 (se=6.0) trees/acre to 383.0 (se=40.1) trees/acre, not counting current regeneration, which is a 532% increase in tree density from pre-settlement times to present. Similarly, the basal area increased from 42.3 (se=5.2) ft²/ac to 121.1 (se=7.8) ft²/ac, representing a 187% increase in basal area (Fulé and others 2003). For the San Francisco Peaks study, Cocke and others (2005) reported an increase in tree density from 106.4 (se=16.5) trees/ac to 340.2 (se=55.9) trees/ac, for a 220% increase, and basal area increased from 67.4 (se=14.3) ft²/ac to 251.7 (se=18.3) ft²/ac, for a 273% increase in current basal area over pre-settlement basal area.

Age Structure - We found no studies that documented the effects of human disturbance on the age structure of spruce-fir forests. One study (Popp and others 1992) delineated minimum criteria for the structural attributes used to determine old-growth and reported that spruce-fir stands should have living trees of 15 trees/ac of 12 inch dbh, or 20 trees/ac of 10 inch dbh (low end), up to 30 trees/ac of 14 inch dbh or 25 trees/ac of 16 inch dbh, with an age of 140 to 170 years. The range of snag (standing dead tree) density requirements is 3 trees/ac of 12 inch dbh at 20 ft tall, to 4 trees/ac of 16 inch dbh at 30 ft tall. They recommended that downed material should be 5 pieces/ac of 12 inch dbh and 16 ft long. They also recommended a blend of single-storied and multi-storied canopies, a range of basal areas of 120 to 140 ft²/ac, and canopy cover of 60 to 70%.

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

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

Synthesis – The overstory density at both the San Francisco Peaks and the Grand Canyon National Park sites has increased 220-530% in trees per acre over the 120-year period covered by the two studies. Similarly, the basal area has increased 190 to 270% in ft²/acre over the same period, with most of the increase in density contributed by fire sensitive species such as white fir, corkbark fir, Engelmann spruce, and aspen. If regeneration (trees<1 inch DBH) is counted, the differences are even more striking, with even higher densities in the current time period (610% to 5,586% increase in trees per

acre). Unfortunately, data are not yet available for concomitant changes in age- or size-class structural changes, nor for canopy cover changes, although the increase in density probably has also increased canopy cover values. The spruce-fir forest on top of the Pinalenos is undergoing massive die-off of mature trees, primarily due to drought, high density of trees and competition, and insect outbreak. With continued drought and fire suppression, more insect outbreaks are likely to occur in other portions of spruce-fir forest throughout the Southwest. The limited spatial distribution, inaccessibility, and high fuel loads of this important, mixed-fire regime adapted forest present difficult challenges for its management.

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Chapter 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 $\geq 60\%$ canopy cover, or in terms of co-dominant species or genera when 2 or more species or genera account for $\geq 80\%$ canopy cover together with each individually having $\geq 20\%$ canopy cover. Life forms are classified as tree if tree canopy cover is $\geq 10\%$, shrub if shrub canopy cover is $\geq 10\%$, and herbaceous if herbaceous canopy cover is $\geq 10\%$ 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.

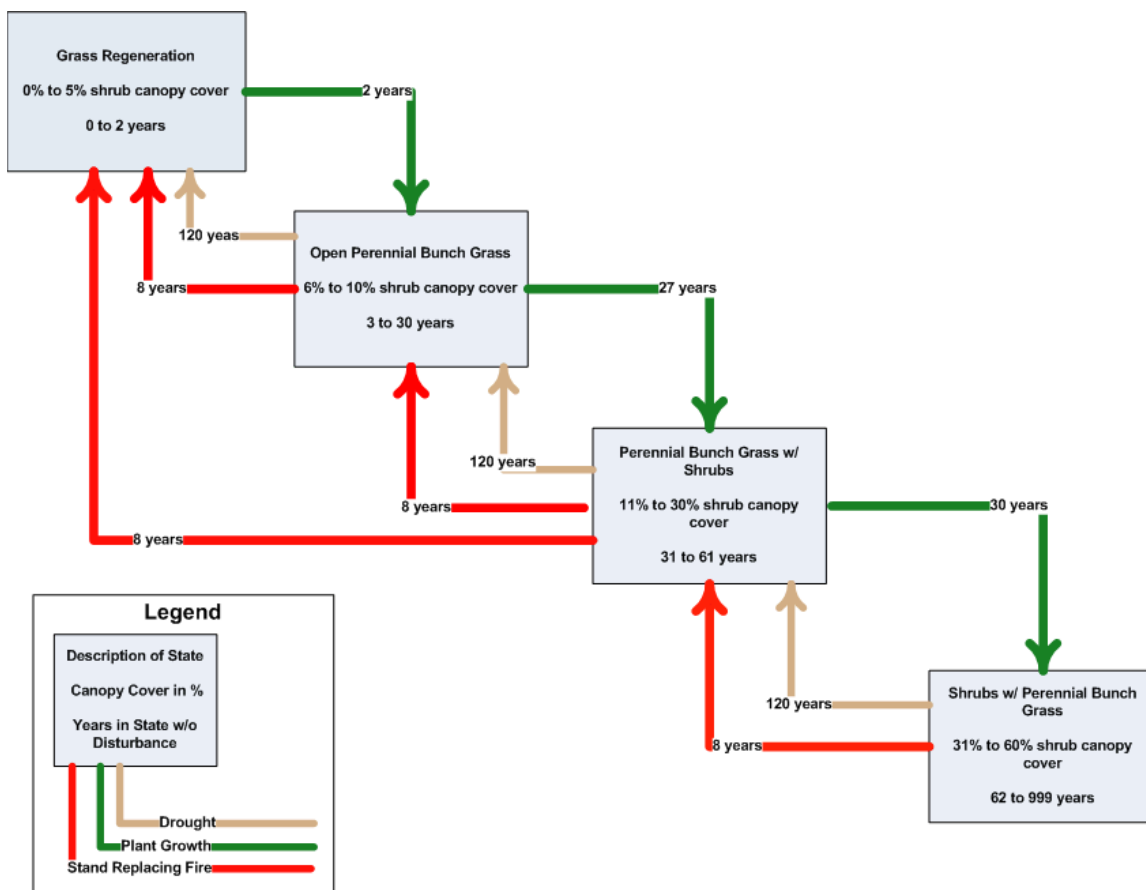


Figure 13-1. Simple grassland model used in sensitivity testing of VDDT software

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.

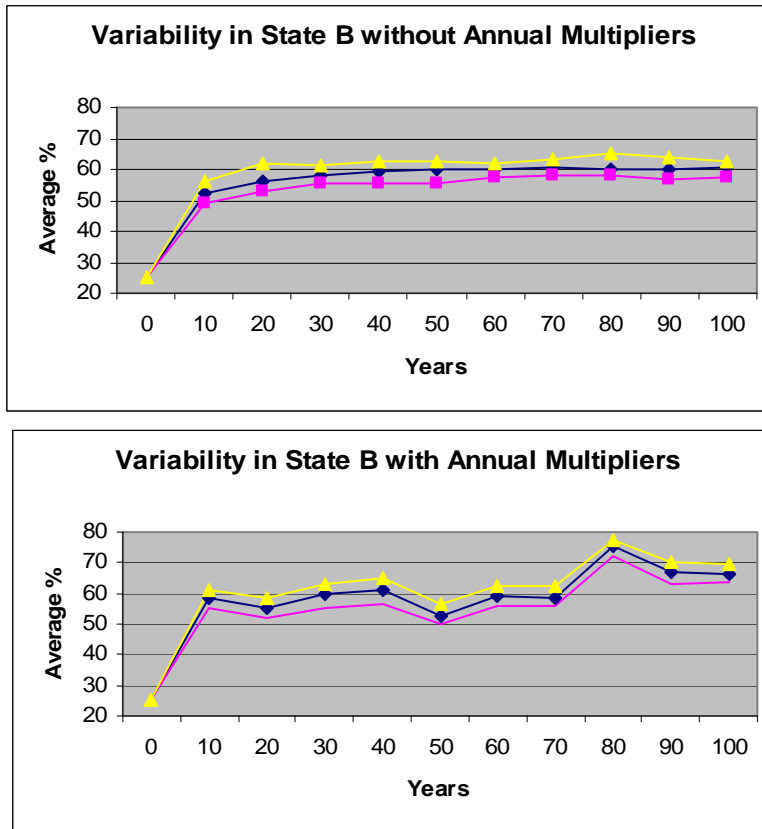


Figure 13-2. Comparison of year to year variability in state B of the simple grassland VDDT model with and without the use of annual multipliers. Maximum values in yellow, average values in blue, and minimum values in pink.

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 between 1709 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 19 - Spruce-Fir Forest Model

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

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

At Grand Canyon National Park on the North Rim, Fulé and others (2003) reported that tree densities and fire-initiated/non-fire-initiated groups were very patchily distributed, and thus stands created by fire could not be discerned from analysis of satellite imagery. This diverse forest structure suggests a combination of a surface fire regime as well as crown-fire initiated groups or stands. They also reported that the mean fire interval (MFI) for spruce-fir was 8.8 years, 8.0 for 10% trees scarred, and 31.0 for 25% scarring (larger fires) for the period 1700 to 1879 (Fulé and others 2003). In addition, they reported a median fire return interval of 7 years, with minimum of 2 years and maximum of 32 years; a standard deviation of 7.7 years, and a Weibull median probability interval (WMPI) of 7.2 years. They found that upper elevation fire dates often coincided with fire dates for lower elevation forest types, suggesting that pre-1880 fires may have been very large (Fulé and others 2003). Most of the fires they studied occurred in summer, and large fire dates occurred in dry years that followed several wet years (Fulé and others 2003). Vankat (2006) asserted that the mixed severity fire regime of the North Rim spruce-fir forest might be a function of the relatively low elevation, but might also be a product of the Southwest's climate. Fulé hypothesized (2006 personal communication) from the 2003 study that south and west facing slopes were dominated by surface fires captured in pine fire scars, whereas north and east facing slopes only experienced stand replacing fires at much longer intervals. These two forest disturbance types are in close enough proximity that when viewed together as a landscape, a mixed severity fire regime is apparent.

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

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

mixed-conifer, and high severity, small-patch size (individual trees or groups) events probably also occurred in this zone” (Swetnam and others 2005).

Vankat (2006) concluded that the mixed severity fire regime of spruce-fir forests in the Southwest may follow a moisture and/or elevation gradient similar to the difference between southern and central Rocky Mountain fire regimes. Swetnam and others (2005) also postulated the possibility that historically, the long term stability of the spruce-fir forest was somewhat protected by the more frequent fire regime in the adjacent and down-slope mixed conifer forests. Frequent fires in the mixed conifer forests maintained more open stands, with low woody fuel accumulations, grassy understories, and higher crown base heights, which may have combined to produce lower intensity surface fires that would be less likely to transition into crown fires upon spread into the spruce-fir forests upslope. Swetnam and others (2005) have observed such fire behavior, and assert that the historic fire regime may have been responsible for maintaining a more patchy vegetation and fuel mosaic that was less conducive to crown fire. Based on Vankat (2006), Fulé and others (2003), and Swetnam and others (2005), we hypothesize that lower elevation (or latitude or moisture) spruce-fir forests may have more frequent, less severe fires more similar to surface fire regimes, and higher elevation/latitude/moisture forests may have infrequent, more severe crown fires, and some areas have both.

Historically, spruce beetles (*Dendroctonus rufipennis*) have had the greatest impact on spruce-fir forests of the central and southern Rocky Mountains, and in the Southwest (Alexander 1987). Historic photographs and tree-ring analysis indicate that there have been six major outbreaks since the middle 1800s, and between 1850 and 1880 a very large-scale outbreak affected forests from northern New Mexico to northern Colorado (Baker and Veblen 1990). These outbreaks are considered part of the natural variability of spruce-fir forests, and spruce beetles likely persist in small windthrow areas (Veblen and others 1991) and in live trees (Veblen and others 1994). Small to large predators such as nematodes to woodpeckers maintain populations at low levels, until conditions favor an outbreak (Alexander 1987). Factors influencing outbreaks include large diameter Engelmann spruce or canopy dominance, slowed growth, mild winters, and well-drained creek bottom sites (Veblen and others 1994, Bebi and others 2003).

19.2 Vegetation Models - Based on this understanding of vegetation dynamics, we created state and transition models depicting historic (pre-1880) and current (1880 to present) vegetation dynamics within this forest type (Figures 19-1 through 19-3). Additionally, we used information from the state and transition models to develop quantitative Vegetation Dynamics Development Tool (VDDT) models. The VDDT software allows the user to model succession as a series of vegetation states that differ in structure, composition, and cover and to specify the amount of time it takes to move from one vegetation state to another in the absence of disturbance. Various disturbance agents affecting the movement of vegetation between states can then be incorporated (e.g., surface fires, stand-replacing fires, grazing, insect outbreaks). By varying the types and rates of disturbance across the landscape, the effects of different management treatments, such as wildland fire use, fire suppression, prescribed burning, grazing practices, and mechanical fuel treatments, on future vegetation can be investigated. While VDDT models can be used to “game play” with different management scenarios, the models we ran in this analysis only include states and transitions for which there is published

information to support their inclusion within the model. We discuss model parameters, output, and analysis below (Tables 19-1 through 19-5).

Historic Southwest Spruce-Fir Forest High Elevation

State and Transition Model

May 2006

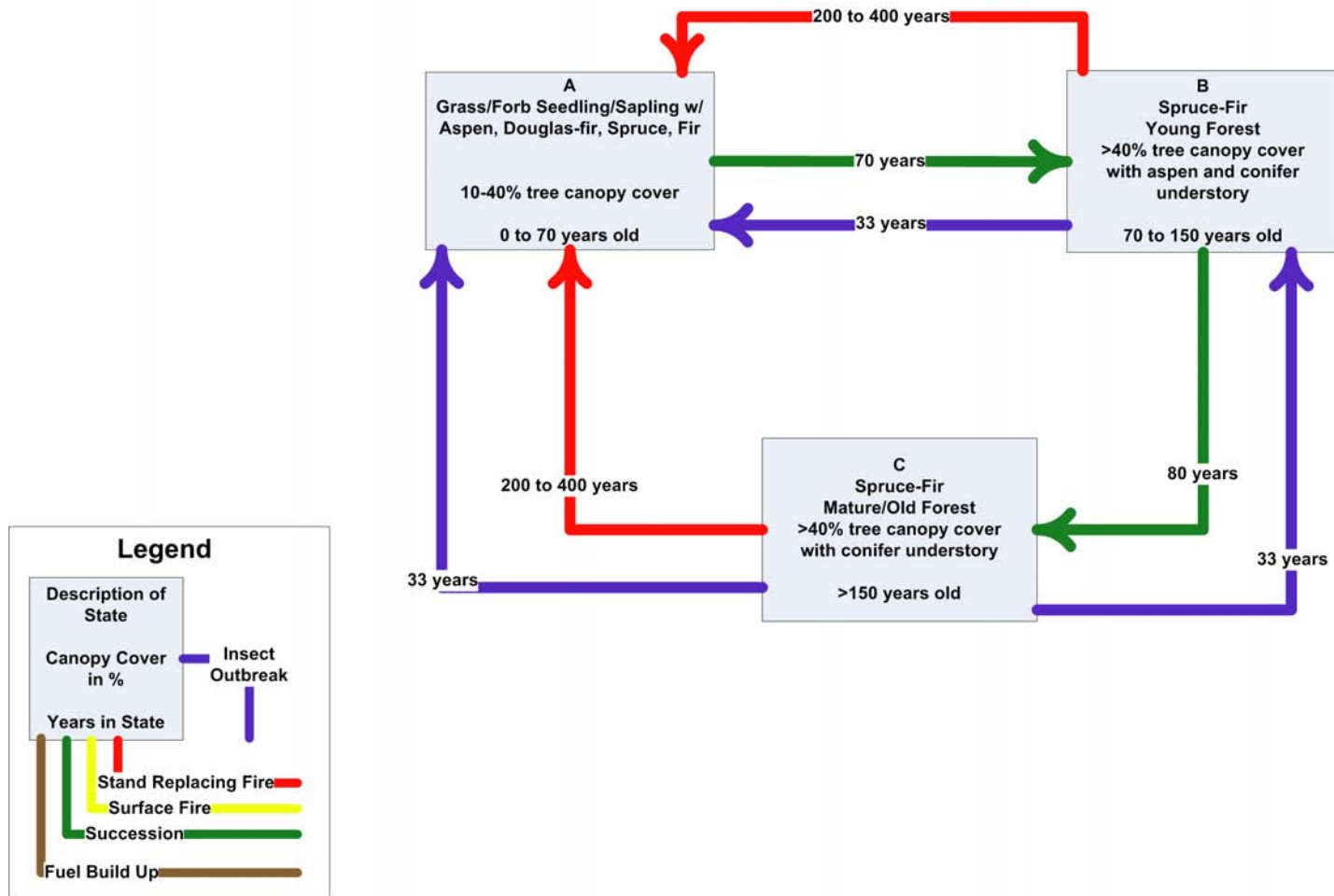


Figure 19-1. Conceptual Historic state and transition model for the spruce-fir forest high elevation vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Historic Southwest Spruce-Fir Forest Low Elevation

State and Transition Model

May 2006

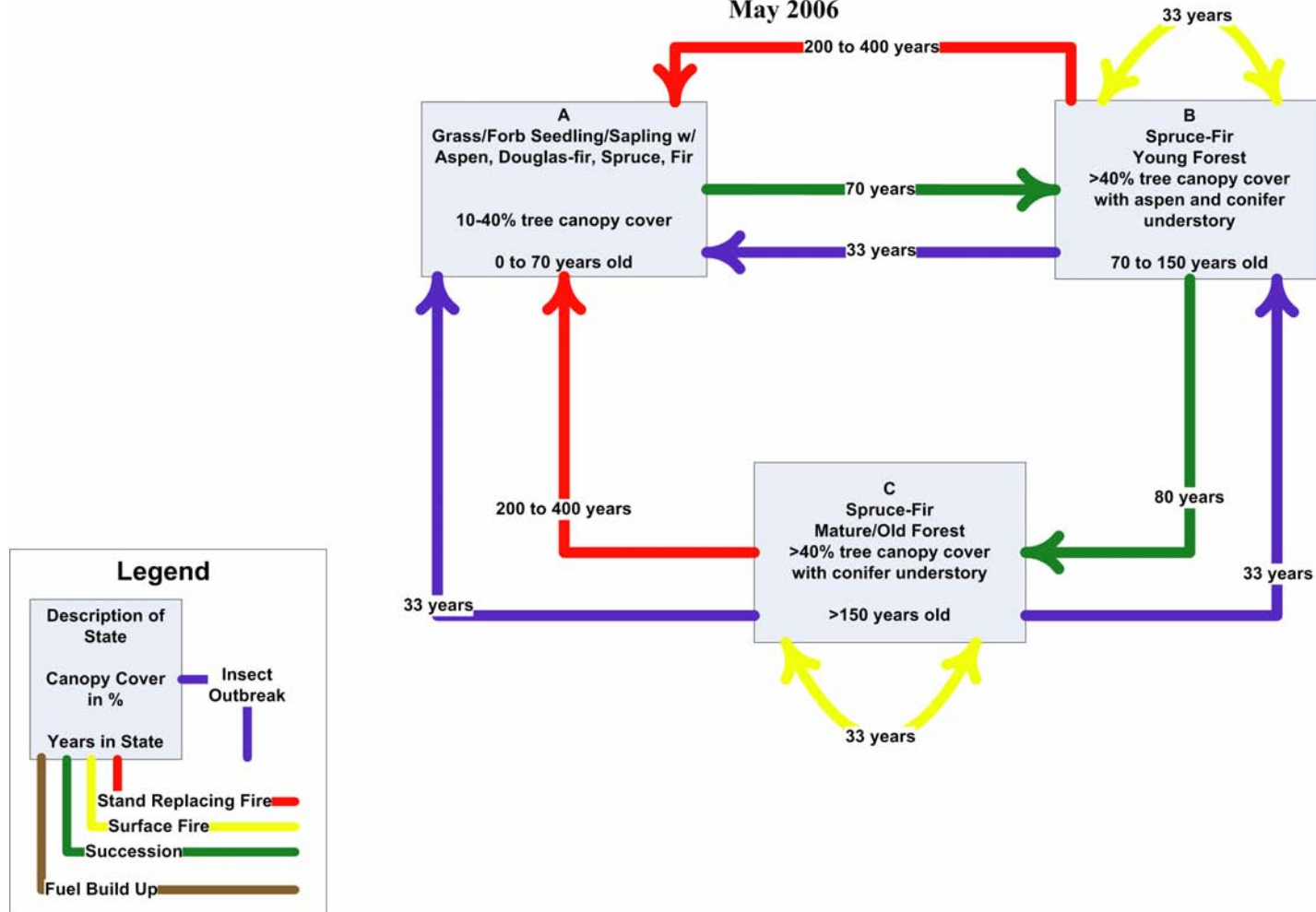


Figure 19-2. Conceptual Historic state and transition model for spruce-fir forest low elevation vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Current Southwest Spruce-Fir Forest High & Low Elevation

State and Transition Model
May 2006

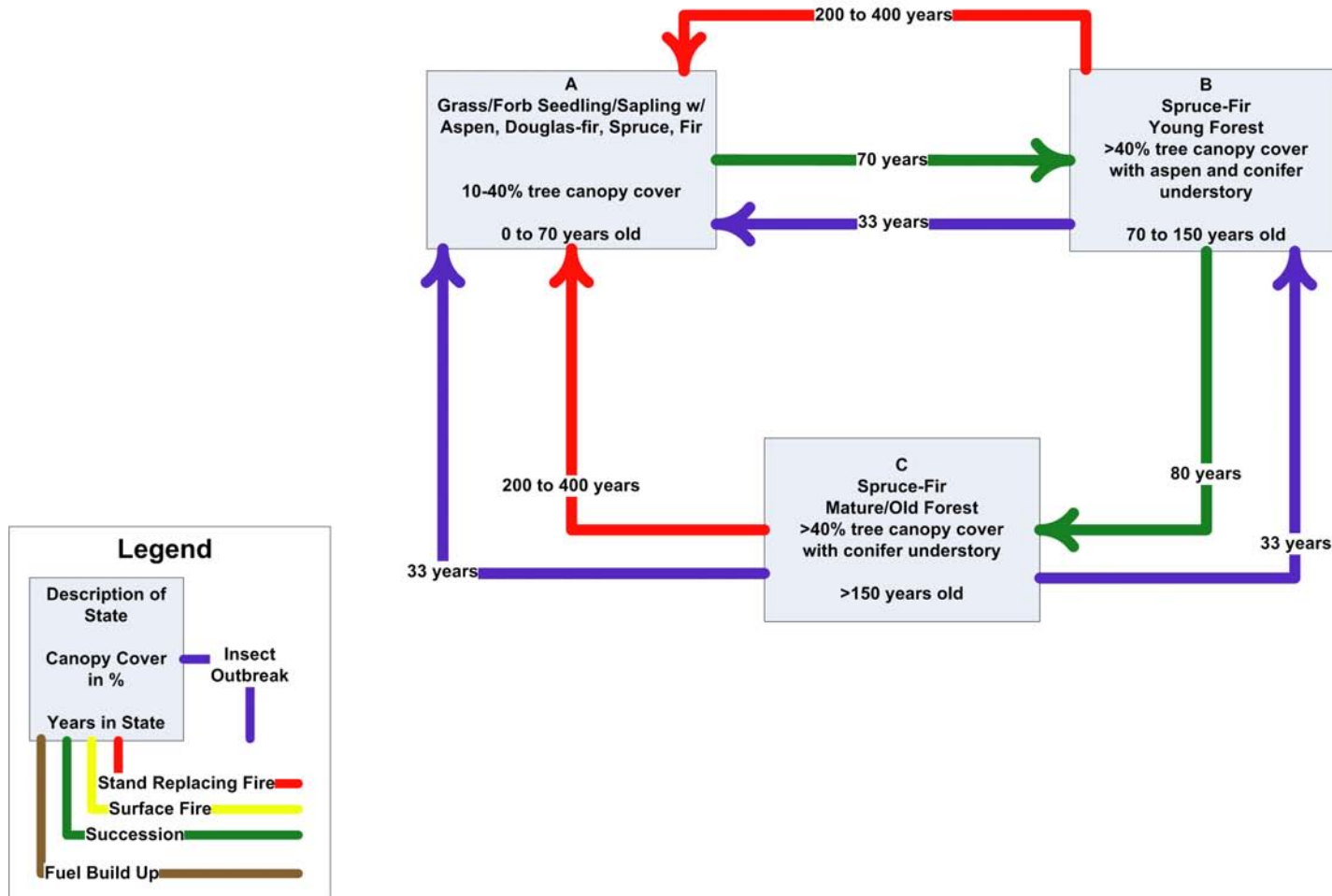


Figure 19-3. Conceptual Current state and transition model for both low and high elevation spruce-fir forest vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Model Parameters

In Table 19-1 we describe the parameters included or not included within the Historic VDDT model for spruce-fir low elevation, as well as the sources of information and any assumptions used to create model parameters. In Table 19-2 we describe the parameters included or not included within the Historic VDDT model for spruce-fir high elevation, as well as the sources of information and any assumptions used to create model parameters. Table 19-2 also describes the parameters for the Current VDDT model for both high and low elevation spruce-fir forest types, since both types are assumed to be within their historic range of variation, with the exception of the cessation of surface fire, which is the only difference between high and low elevation spruce-fir forest.

Table 19-1. Identification of Historic transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the VDDT model for spruce-fir low elevation with mixed severity fire.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Plant Growth	70 to 80 years between states	Transitions among model states were taken from data developed by Veblen and others (1994) and Vankat (2006).	We assume that transition from seedling/sapling to young forest takes approximately 70 years, and from young to old/mature forest takes approximately 80 years.
Regeneration from seed	Unknown, not used in model.	Seed production and seedling recruitment is highly variable in both space and time, and there is insufficient information to assign a probability for this transition across the entire region.	Due to the lack of data on seedling recruitment, this transition is not included in the model, and hence the proportion of this seedling/sapling class of vegetation is presumed to be underestimated in the model
Surface Fire	31 years	Fulé and others 2003.	These data are based on direct evidence (fire scar data). We used the 25% scarring filter (large fires) for modeling purposes. Because there was only one value for surface fire reported in the literature, we did not model minimum, maximum, and mean for surface fires.
Stand Replacing Fire	Once every 200 to 400 years	Stand replacing fire was reported to be pervasive but relatively infrequent (Dieterich 1983, Moir 1993, Fulé and others 2003, Vankat 2006).	Stand replacing fire occurred across all spruce-fir forests, returning the forest to the seedling/sapling stage.
Insect Outbreak	Once every 33 years for young forests; once every 100 years for	Spruce beetle outbreaks have been documented for the northern portion of the range (northern New Mexico, Kaibab Plateau) and for the Pinalenos in southern	We assumed that insect outbreaks occurred at a lower frequency for young forest than for old/mature forest, returning young forest to regeneration state "A", and returning old/mature forest to equal proportions of

Transition Type	Transition Frequency or Length	Sources	Assumptions
	old/mature forests.	Arizona(Baker and Veblen 1990, Koprowski and others 2005).	regeneration and young forest.
Silvicultural Activities	Not used in model		Spruce-fir forests were too high in elevation, too inaccessible, or of too low value to have been significantly affected by timber harvest.

Table 19-2. Identification of Historic and Current transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the VDDT model for spruce-fir high elevation with stand replacing fire. A major assumption here is that Historic and Current conditions are sufficiently similar for the Spruce-Fir high elevation vegetation type that current forests are within the historic range of variability, because the time elapsed since the last crown fire is within the range of FRI. Current conditions for the spruce-fir wet forest vegetation type were assumed to reflect the current conditions of the dry spruce-fir forest type because the surface fire regime, which is the only difference between the two (from a disturbance standpoint), has been extinguished.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Plant Growth	70 to 80 years between states	Transitions among model states were taken from data developed by Veblen and others (1994) and Vankat (2006).	We assume that transition from seedling/sapling to young forest takes approximately 70 years, and from young to old/mature forest takes approximately 80 years.
Regeneration from seed	Unknown, not used in model.	Seed production and seedling recruitment is highly variable in both space and time, and there is insufficient information to assign a probability for this transition across the entire region.	Due to the lack of data on seedling recruitment, this transition is not included in the model, and hence the proportion of this seedling/sapling class of vegetation is presumed to be underestimated in the model
Stand Replacing Fire	Once every 200 to 400 years	Stand replacing fire was reported to occur in small patches throughout spruce-fir's range, but relatively infrequent (Dieterich 1983, Moir 1993, Fulé and others 2003, Vankat 2006).	Stand replacing fire occurred in small patches across all spruce-fir forests, returning the forest to the seedling/sapling stage.
Insect Outbreak	Once every 33 years for young	Spruce beetle outbreaks have been documented for the northern portion of the range (northern New Mexico, Kaibab	We assumed that insect outbreaks occurred at a lower frequency for young forest than for old/mature forest, returning young forest to regeneration state "A", and returning

	forests; once every 100 years for old/mature forests.	Plateau) and for the Pinalenos in southern Arizona(Baker and Veblen 1990, Koprowski and others 2005).	old/mature forest to equal proportions of regeneration and young forest.
Silvicultural Activities	Not used in model		Spruce-fir forests were too high in elevation, too inaccessible, or of too low value to have been significantly affected by timber harvest.

19.3 Results – Results of the historic spruce-fir models indicate a small amount of variability in the 900-year average for each state based on the fire interval range (Table 19-3 and 19-4). For both the high and low elevation forest types, as stand replacing fire decreases in frequency the proportion of the landscape in the youngest state decreases from 31.4% to 21.1% (low elevation), and from 31.1% to 20.7% for the high elevation model. Conversely, for both models, as stand replacing fire decreases in frequency from once every 200 years to once every 400 years, the proportion of the landscape in the oldest age class (C) increases from 32.1% to 44.5% (low elevation), and 32.5% to 45.0% (high elevation). Although the two different models differ in having a mixed fire regime (low elevation) and a stand replacing only fire regime (high elevation), the model outputs are very similar. Although there are small differences between the two models and among the different fire return intervals, all model outputs portray a landscape that has fairly even proportions of the different model states (total range = 18.8% to 47.2%).

Results for the current spruce-fir low and high elevation models (Table 19-5), which were run for 120 years following development of the Historic conditions, were very similar to results from the Historic models. For example, under a stand replacing fire return interval of 300 years, average landscape proportion of the historic low elevation spruce-fir was 39.8%, for high elevation spruce-fir it was 40.5%, and for the current model for both high and low elevation spruce-fir it was 45.4%.

Table 19-3. Results for the Historic spruce-fir low elevation VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (300 years), maximum (400 years), and minimum (200 years) of the estimated fire return interval range for stand replacing fire.

Stand Replacing Fire Return Interval (FRI) Modeled	Model Output	Grass/Seedling & Sapling A Open	Young Forest B Open to Closed	Old/Mature Forest C Multi-storied
Every 200 years	Average	31.4	36.6	32.1
	Minimum	29.1	34.1	29.8
	Maximum	33.7	39.1	34.3
	Standard Deviation	1.4	1.5	1.4
Every 300 years	Average	24.9	35.3	39.8
	Minimum	23.1	33.2	37.5
	Maximum	26.9	37.3	42.1
	Standard Deviation	1.2	1.3	1.4
Every 400 years	Average	21.1	34.4	44.5
	Minimum	19.3	32.1	42.2
	Maximum	23.1	36.8	47.1
	Standard Deviation	1.2	1.4	1.5

Table 19-4. Results of the Historic spruce-fir high elevation forest VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (300 years), maximum (400 years), and minimum (200 years) of the estimated fire return interval range for stand replacing fire.

Stand Replacing Fire Return Interval (FRI) Modeled	Model Output	Grass/Seedling & Sapling A Open	Young Forest B Open to Closed	Old/Mature Forest C Multi-storied
Every 200 years	Average	31.1	36.4	32.5
	Minimum	28.8	33.9	30.2
	Maximum	33.5	38.8	34.9
	Standard Deviation	1.4	1.5	1.4
Every 300 years	Average	24.5	35.0	40.5
	Minimum	22.5	32.8	38.1
	Maximum	26.5	37.3	42.7
	Standard Deviation	1.2	1.4	1.4
Every 400 years	Average	20.7	34.3	45.0
	Minimum	18.8	32.0	42.6
	Maximum	22.5	36.6	47.2
	Standard Deviation	1.1	1.4	1.4

Table 19-5. Results of the Current spruce-fir high and low elevation forest VDDT model, reported as the 120 year end value for average, minimum, maximum, and average standard deviation of the percent of the modeled landscape in each state.

Stand Replacing Fire Return Interval (FRI) Modeled	Model Output	Grass/Seedling & Sapling A Open	Young Forest B Open to Closed	Old/Mature Forest C Multi-storied
Every 200 years	Average	30.6	36.7	32.7
	Minimum	28.8	35.1	31.5
	Maximum	32.2	38.2	33.8
	Standard Deviation	1.0	0.9	0.7
Every 300 years	Average	21.1	32.5	45.4
	Minimum	19.4	30.9	43.8
	Maximum	22.8	34.6	47.5
	Standard Deviation	1.0	1.1	1.1
Every 400 years	Average	21.1	33.5	45.4
	Minimum	19.4	31.9	43.8
	Maximum	22.8	35.8	47.5
	Standard Deviation	1.0	1.1	1.1

19.4 Discussion – These modeled scenarios indicate that although there may be large differences in canopy and understory composition due to differences in elevation and moisture regimes, the resulting difference in disturbance regime (surface plus canopy versus strictly canopy fire) has a minor effect on the relative proportion of the landscape in the three model states (regeneration, young, and old forest). Also, elimination of a relatively long-rotation surface fire regime (31 years) has had minimal effect on the relative proportion of model states over 120 years. This is in concurrence with other authors’ contention that spruce-fir forest conditions in many areas of the Southwest may be within its historic range of variation, and that insufficient time has elapsed since the last crown fire for us to have experienced crown fire in contemporary time (Swetnam and others 2005, Koprowski and others 2005, Vankat 2006).

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