

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



**Southwest Forest Assessment Project
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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 Riparian Forest	17,300	14,400	800	600	42,800	11,500	12,100	100	4,100	103,700
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

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
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 PNV 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 includes mostly photographs from the Ft. Valley Research Station archive, but also from the RMRS on-line photographs
Rocky Mountain Research Station	Flagstaff, AZ	Susan Olberding	No	interior chaparral (on-line resource only), ponderosa pine, riparian	Photographs from several field season that investigated the effects of fire over several years
Saguaro National Park	Tucson, AZ	James Leckie	No	Madrean encinal, Madrean pine-oak	
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 PNVs that we thought were good examples of various model states within a PNV (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

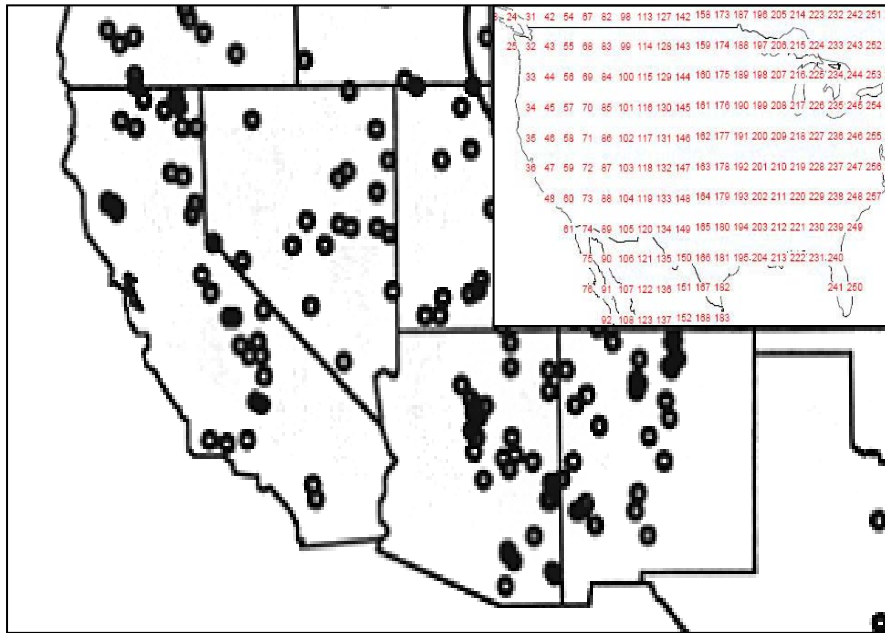
Methods- For a detailed discussion of the methodology used to identify 1) year to year variability, 2) range, frequency, and length of extreme and non-extreme events, and 3) spatial and temporal comparison, see Schussman 2006 (Assessing Low, Moderate, and High Severity Dry and Wet Events Across the Southwestern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

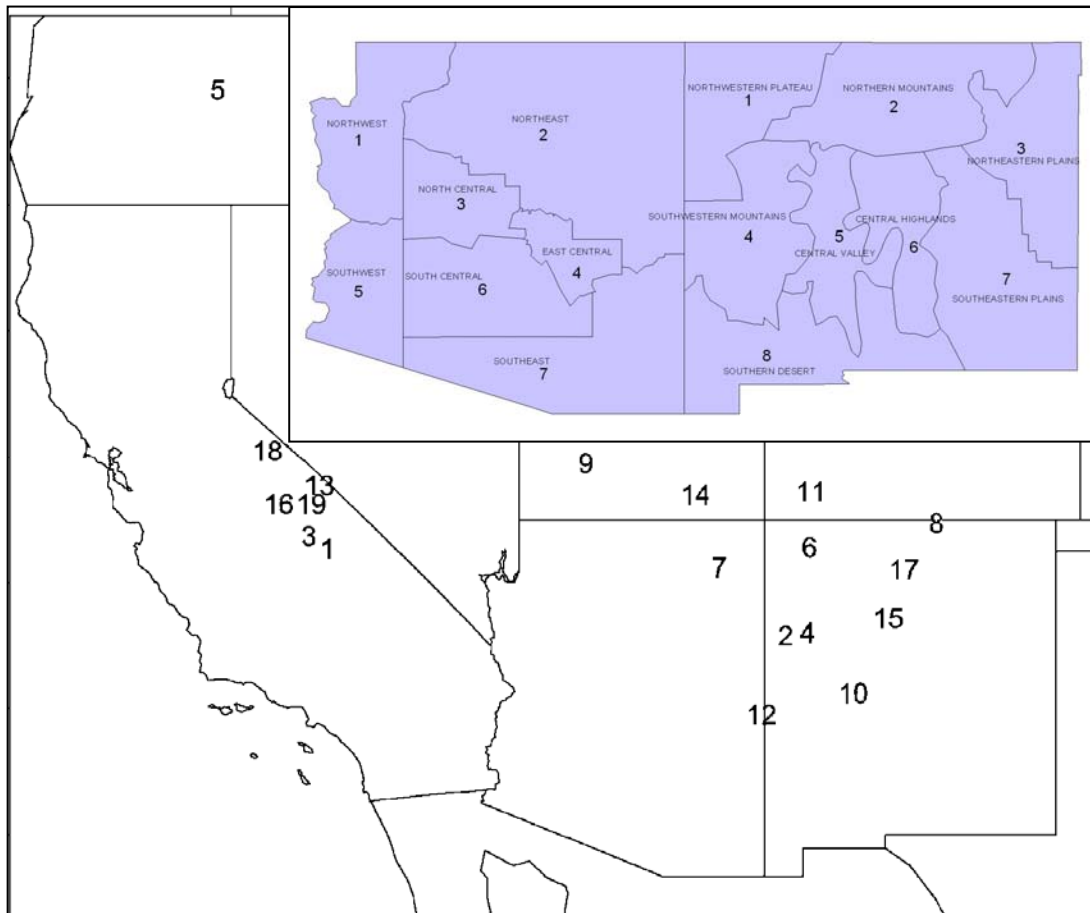
Table 1-5, , and Appendix 1 - Table 1.2 and Figures 1.16 to 1.23). Overall there is little regional variability in the percent of dry and wet years for either the winter precipitation or summer PDSI data sets. Of the regional variability that is present, the majority of the

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 (, , 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.

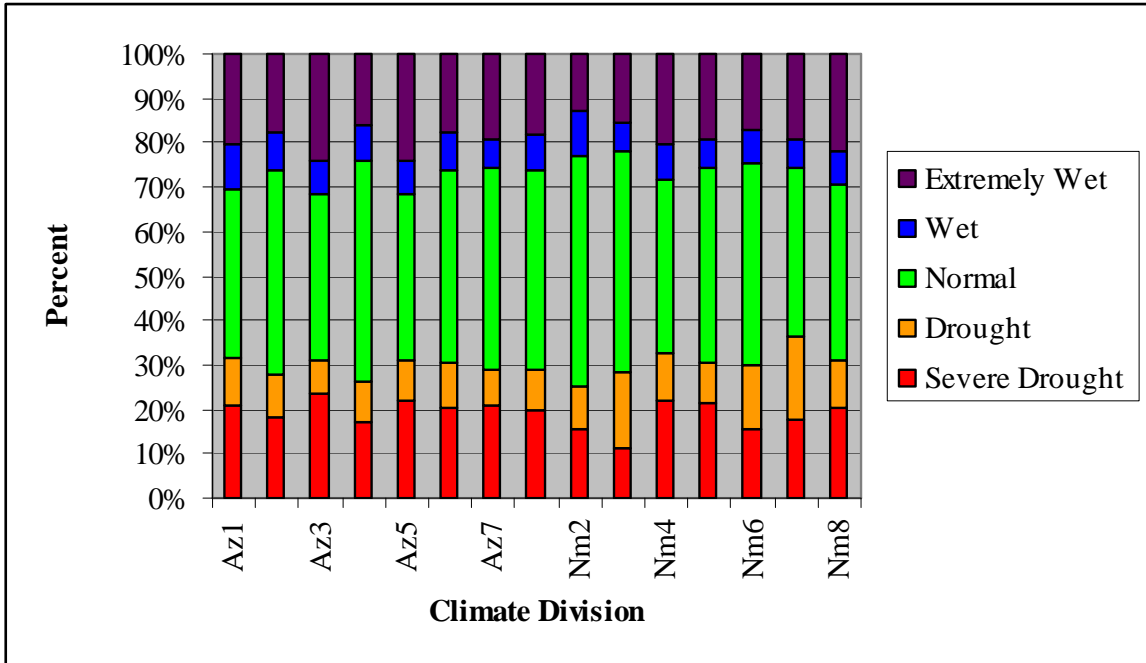


Figure 1-2. Comparison of the percent of years in all year types for all climate divisions in the Southwest.

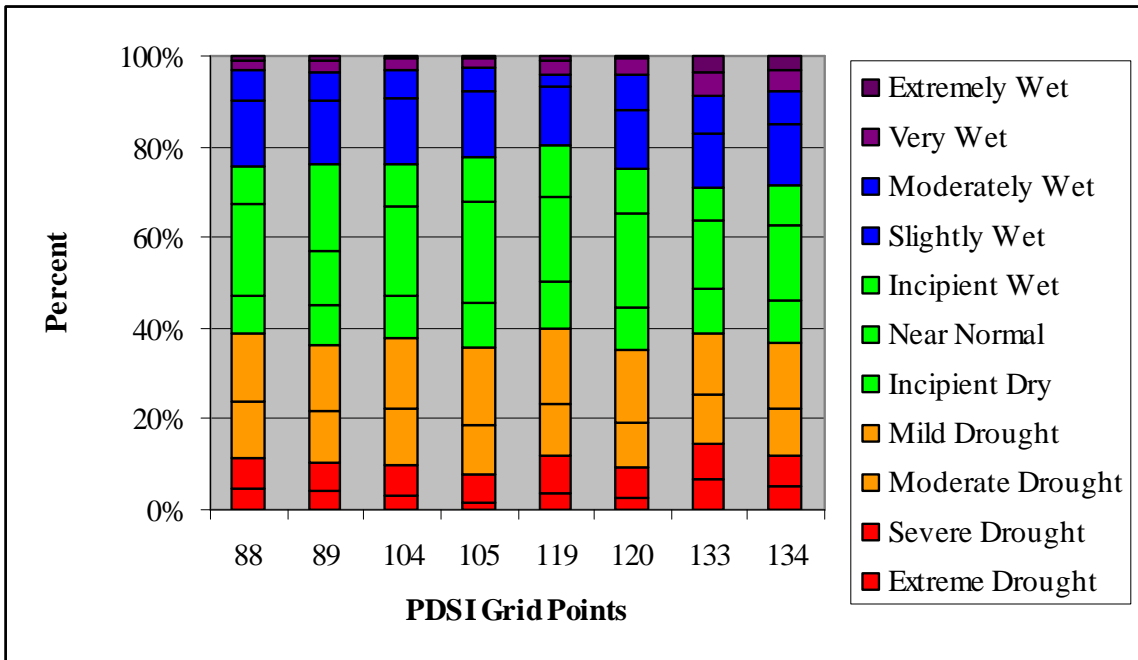


Figure 1-3. Comparison of the percent of years in all year types for all PDSI grid locations in the Southwest.

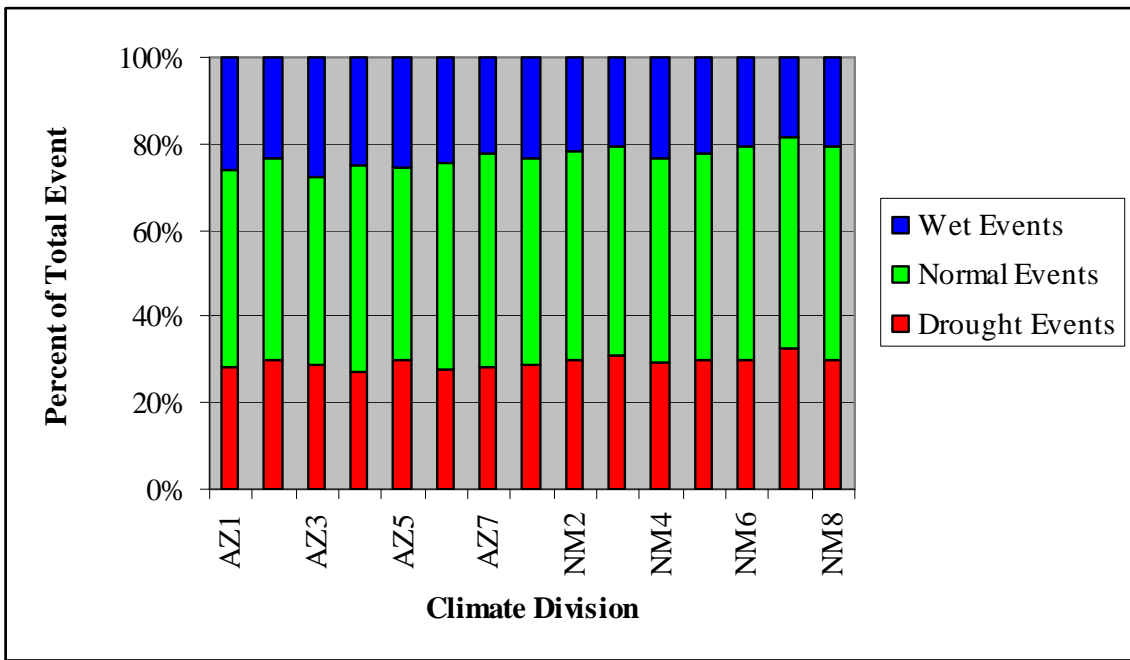


Figure1-4. Comparison of the percent of events classified as drought, normal, and wet events for all climate divisions in the Southwest.

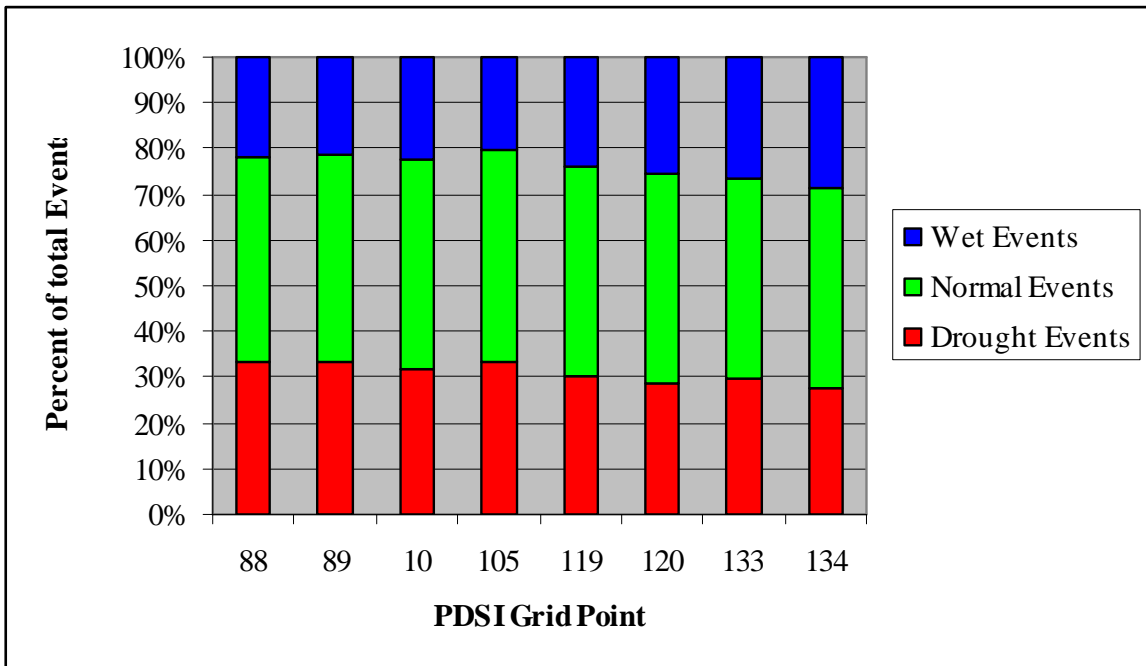


Figure1-5. Comparison of the percent of events classified as drought, normal, and wet events for all PDSI grid locations in the Southwest.

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 11 - Montane and Subalpine Grasslands

11.1 General Description – High-elevation grasslands generally occur in small (< 100 ac.) to medium-sized (100 to 1000 ac) patches, and rarely in patches >1000 ac throughout the extent of spruce-fir and mixed conifer forests in the southwestern region, with subalpine grasslands occurring at slightly higher elevations than montane grasslands (Allen 1984, Dick-Peddie 1993, White 2002). In this report, we refer to both types of grasslands as high elevation grasslands, unless differentiation is supported in the literature. These high elevation grasslands occur from about 8,500 ft. to 11,500 ft. in elevation, and have been divided by several authors into two categories, “upper slope” and “valley bottom” grasslands (Allen 1984, Dick-Peddie 1993, Muldavin and Tonne 2003). The valley bottom grasslands (or meadows or wet meadows *sensu* Brown 1994) are sometimes discontinuous and occur in smaller patches, but also occur as large contiguous grasslands known as *valles*, a Spanish word for valleys. The upper slope grasslands can form large, contiguous patches, often on the southern or western flanks of mountains (Allen 1984, Vankat 2006). Although sharing some of the same species, these grasslands are not the same as the bunchgrass communities found under many southwestern ponderosa pine forests (Chapter 7, this volume). Another community type found at higher elevation, alpine grasslands are covered in the alpine tundra chapter (Chapter 10) of this volume.

According to Southwest Regional Gap Analysis Program (SWReGAP) data, montane and subalpine grasslands cover approximately 641,500 acres, or 1.6% of the total land mass for Arizona and New Mexico (USGS 2004). The USFS manages about 328,800 acres of these grasslands, or about 51.3% of the extent of montane and subalpine grasslands in the SW (see Table 11-1). There are no published climate data for montane and subalpine grasslands, but the climate is assumed to be similar to the intervenient forests.

Apache-Sitgreaves	Carson	Cibola	Coconino	Coronado	Gila
56,800	130,500	1,600	31,900	0	10,200
Kaibab	Lincoln	Prescott	Santa Fe	Tonto	Total
26,800	14,700	0	56,300	0	328,800

Table 11-1. Approximate area (in acres) of montane and subalpine Grassland potential natural vegetation type (PNVT) across eleven Region 3 National Forests in Arizona and New Mexico (USGS 2004). 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.

The vegetation composition of montane and subalpine grasslands includes tall (<3 ft.) bunchgrasses (see Figure 11-1). Dominant genera include fescues (*Festuca*), muhly (*Muhlenbergia*), oatgrasses (*Danthonia*), tufted hairgrass (*Deschampsia*), pine dropseed (*Blepharoneuron*), junegrass (*Koeleria*), and some bluegrass (*Poa*) species, both native and exotic (introduced, non-native). When montane grassland is found on small valley bottoms, it is usually dominated by mixed sedge (*Carex*) species and tufted hairgrass (*D. caespitosa*), native bluegrasses (*P. interior* and *P. longiligula*), and the exotic Kentucky (*P. pratensis*) and Canada bluegrasses (*P. compressa*); or native species of danthonia (Dick-Peddie 1993). Rushes (*Juncus*) and flat sedges (*Cyperus*) are also common in these wet meadows (see Figure 11-2). Forbs share

dominance with grasses in high elevation grasslands, and they may contribute up to 35% of the live vegetation cover in these communities (Allen 1984). Composition of forbs in



Figure 11-1. Montane grasslands from the Apache-Sitgreaves National Forests. Note height of perennial bunchgrasses and absence of bare ground, and large extent of grasslands in background. Photograph courtesy of USDA USFS Regional Office archive, unknown date.



Figure 11-2. Wet meadow montane grassland near the San Francisco Peaks dominated by rich mixture of grasses and bracken fern, with scattered aspen and Bebb willow trees. Photograph by J.K. Hillers, US Geological Survey, photographer for J.W.Powell expedition, 1885.

these communities is due to the presence of several different species, rather than high densities of one or two species of forbs. Common forbs are mountain dandelion (*Agoseris aurantiaca*), fleabanes (*Erigeron spp.*), cinquefoils (*Pentaphyllum* or *Potentilla spp.*), Rocky Mountain iris (*Iris missouriensis*), and bracken fern (*Pteridium aquilinum*). More detailed plant lists from high elevation grasslands are available for New Mexico from Allen (1984), Muldavin and Tonne (2003), and Coop and Givnish (2007a) and for Arizona from White (2002), and Terrestrial Ecosystem Surveys for individual National Forests in AZ and NM (Laing and others 1989, Robbie 1980, Brewer and others 1984, Miller and others 1995, Robertson and others 2000, etc.).

Soils in montane and subalpine meadows tend to be deep Mollisols, or grassland soils, as opposed to Alfisols and Inceptisols, which are found beneath adjacent, old forested areas (Allen 1984, White 2002), although these other soil types are found within SW high elevation grasslands. By comparison, Mollisols have a deeper A horizon, that is darker and contains more organic matter than either forest soil (USDA/AID 1992). Also, the grassland soils tend to be finer-textured and more poorly drained (Dick-Peddie 1993).

11.2 Historical Range of Variation of Ecological Processes

Vegetation Dynamics – Very little is known about the historical (or pre-settlement) condition and processes of montane and subalpine grasslands. By the time ecologists began to describe the flora of these systems, natural fire regimes had already been disrupted, and introduced grazing

animals had been active on the landscape for many years (Leiberg and others 1904, Allen 1984). Furthermore, it has been suggested that no reference sites exist for southwestern montane and subalpine grassland ecosystems because no areas have had grazing excluded for a significant length of time (Chambers and Holthausen 2000) (see Figure 11-3). To this day, some physical evidence remains both within subalpine and montane grasslands and adjacent forested areas to allow some ecological inferences about the origins of high elevation grasslands, their composition, and the processes that maintained and changed them. From analysis of plant macrofossils and pollen found in lake sediments, higher percentages of grasses and composites did not occur on the landscape until about 10,400 years ago, indicating a distinct and widespread change of climate that favored grasses and composites over previously extant vegetation types (Anderson 1993).



Figure11-3. Montane grassland on west side of San Francisco Peaks near Flagstaff. Note low stature of grasses as evidence of early grazing. Photograph courtesy of UC Berkeley Bancroft Library. Photo by C.H. Merriam from 1889 expedition.

Allen (1984) described three lines of evidence that contribute to our understanding of the potential age of high elevation grasslands: the depth of current high elevation grassland mollic epipedons (upper soil horizon rich in organics) is on the order of 20 to 30 in or more, indicating thousands of years of development; the depth to organic carbon is sufficiently deep to indicate thousands of years of development; and early Holocene drought, and a concomitant increase in fires may have contributed to the decline of precedent alpine tundra vegetation. As the climate favored more fire tolerant grasses and forbs in high elevation areas, fire-intolerant forest tree species were less capable of colonizing these areas, and grasses and forbs were at a competitive advantage. Allen (1984) developed a generalized descriptive model of processes that maintain

montane grasslands. The major determinants of grassland maintenance included microclimate and other site characteristics, fire, grazing, competition, and forest tree establishment. These factors are described in the sections below. Little information exists on the specifics of historic high elevation grassland community dynamics, and most of the published information infers historic dynamics from what has been discerned from studies of changes in patterns and processes that have occurred as a result of post-settlement effects of modern disruption of natural disturbance processes.

Most authors studying high elevation grasslands have noted the presence of a few, scattered, pre-settlement coniferous trees, thus mechanisms have existed for their recruitment into grassland systems (Allen 1984, Moore and Huffman 2004, Coop and Givnish 2007a,b). Some high elevation grasslands have no pre-or post-settlement coniferous trees, especially in Thurber fescue dominated grasslands (Moir 1967). However, this site in the Sacramento Mountains of New Mexico should be revisited, since Moir's study was conducted in the mid-1960s. Moore and Huffman (2004) reported that the greatest tree establishment in Grand Canyon National Park occurred after the early 1970s. Coop and Givnish (2007a) recently studied the susceptibility of some non-invaded grasslands at Valles Caldera, New Mexico, to determine the relationships between forest/grassland gradients and several environmental variables, especially where an abrupt, long-term line exists with forest above, and grassland below, in what the authors called a "reversed tree-line." Slope inclination, soil texture, nighttime minimum temperatures, and soil nutrient levels are all significantly related to vegetation patterns, leading the authors to conclude that "lower nightly minimum temperatures and fewer consecutive frost-free days resulting from cold-air drainage may prevent tree seedling establishment in valley bottoms via photo-inhibition, tissue damage, or frost heaving. Fine textured soils may also impede tree seedling establishment in valley bottoms" (Coop and Givnish 2007a).

Clark's nutcracker (*Nucifraga columbiana*) is known to cache seeds of white pines (*P. flexilis*, *P. strobiformis*, *P. aristata*) in high elevation grasslands (Vander Wall and Balda 1977), and these seeds can remain viable following fire (Tomback and Schuster 1994). Individual birds can carry up to 150 seeds at a time up to 14 miles away from the source tree, and cache three to ten seeds at a time over a wide area to be retrieved later. Each bird is capable of caching 25,000 to 35,000 seeds per year, up to 100,000 seeds in years with good seed production. However, while their spatial memories are good, retrieval is not always 100%, and some seeds germinate, often forming clusters of trees, or multiple-stemmed, fused trees of different genetic origin (Tomback and others 2001).

Disturbance Processes and Regimes

Climate- See *Introduction* for climate information. Although some authors have claimed that climate has minimal effect on high elevation grassland composition and structure (Allen 1984), others have proposed climate as an important determinant of tree encroachment (Dyer and Moffett 1999).

Fire- Most grasslands are subject to invasion by woody vegetation, and subalpine and montane grasslands are found within a mosaic of reproductive coniferous forests. However, fire is acknowledged to be the most influential force in checking tree invasion or encroachment

(Daubenmire 1968, Allen 1984). Various studies of fire frequencies or fire rotations in spruce-fir and mixed conifer forests come from sites that are near or immediately adjacent to these high elevation grasslands (for mixed conifer: Swetnam and Baisan 1996, Touchan and others 1996, Fule and others 2003, Heinlein and others 2005, Swetnam and others 2005; for spruce-fir: Grissino-Mayer and others 1995, Swetnam and Baisan 1996, Fule and others 2003, Swetnam and others 2005, Vankat 2006). Adjacent grasslands, with drier microclimates and annual accumulation of fuels are assumed to have burned at least as frequently as surrounding forest types (Allen 1984). During these grassland fires, young tree seedlings would have been killed due to the intensity of fire behavior in grassland fuels, maintaining intact grasslands (Daubenmire 1968). For spruce fir forests, we reported an average fire rotation of 200 years (range = 100 to 300 years), and for mixed conifer forests, we reported an average fire rotation of 15 yrs (range = 5 to 33 yrs) (Schussman and Smith 2006).

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

Herbivory- We found no studies that documented herbivory as important historical ecological determinants for the high elevation grassland vegetation type. We assume that herbivores did use grasslands for forage, including mule deer, Merriam's elk, pronghorn antelope, and various rodents and insects (Turner and Paulsen 1976, Vankat 2006). Different grazers show preference for different species of grass and/or forbs, and reduce the more palatable species with high variability in space and time, but little is known about direct effects on high elevation grasslands.

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

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

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

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

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

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

Synthesis – High elevation grasslands, composed of a variety of perennial bunchgrasses and forbs, have been relatively stable vegetation types for thousands of years, based on descriptions of soil composition and depth. It is also apparent that tree recruitment occurred on the edges of these grasslands during the pre-settlement period, mediated by wind and Clark's nutcracker, but that tree recruitment was controlled by site characteristics and widespread and frequent fire,

based on fire histories of adjacent forests. Tree invasion of high elevation grasslands prior to the mid-1800s was very sporadic, and probably did not lead to type conversion of grassland to forest over a significant proportion of the landscape..

11.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation - We found no studies that documented historical patch composition of the high elevation grassland vegetation type.

Overstory – Not applicable.

Understory and Herbaceous Layer – We found nothing in the literature that documented anything beyond descriptions of the dominant grasses: Arizona fescue, Thurber fescue, oatgrass, Junegrass, mountain muhly, native bluegrasses, tufted hairgrass, rushes and sedges, and a variety of forbs, as described in the introduction, above. The earliest descriptions of high elevation grassland composition come from the White Mountains of eastern Arizona (unpublished range surveys reported by White 2002), the Kaibab Plateau (Mead 1930), southwestern Utah (Cottam and Stewart 1940), and the Coconino and Kaibab Plateaus (Strahler 1944), well after human settlement.

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

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

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

Life Form - We found no studies that documented the historical life form of the high elevation grassland vegetation type.

Density - We found no studies that documented the historical density of the high elevation grassland vegetation type.

Age Structure - We found no studies that documented the historical age structure of the high elevation grassland vegetation type.

Patch Dispersion – We found no studies that documented the historical patch dispersion of the high elevation grassland vegetation type.

Recruitment Dynamics - We found no studies that documented the historical recruitment dynamics of the high elevation grassland vegetation type.

Reference Sites Used – Almost all high elevation grasslands have been subjected to grazing for 150 to 400 years (Scurlock and Finch 1997), although sites exist that have had grazing excluded since the latter part of the 20th century or early part of the 21st century. There are high elevation

grasslands on the North Rim of Grand Canyon National Park that have been excluded from cattle grazing since 1920 to 1930 (Vankat 2006).

Synthesis – Because high elevation grasslands are primarily herbaceous vegetation consisting of forbs and graminoids, there remains little evidence of historical vegetation patterns. There are no pre-settlement studies that describe community composition and structure, thus we do not know if there are differences in community composition and structure for different seral stages, or differences in composition and structure between pre- and post-settlement grasslands.

11.4 Anthropogenic Disturbance (or Disturbance Exclusion)

Herbivory – Cattle and sheep were introduced to the Southwest in the 16th century, and their numbers have fluctuated widely, starting with thousands of animals brought in by early explorers and missionaries in 1598, with numbers peaking around 1900, when millions of sheep, and hundreds of thousands of cattle were allowed to graze throughout the two states (Scurlock and Finch 1997) (see Figure 11-4). The Forest Service reduced numbers of permitted livestock on public lands because of erosion problems and deterioration of range condition around the turn of the last century, and combined totals of permitted animals dropped from about 738,000 in 1919 to fewer than 536,000 ten years later (Scurlock and Finch 1997).



Figure 11-4 Band of sheep grazing on montane grasslands of the Apache-Sitgreaves National Forests. Photo courtesy of USDA USFS Region 3. Unknown date and location.

Grazing impacts from domestic cattle and sheep on high elevation grasslands have been described by Allen (1984). By ageing coniferous tree species within grasslands, Allen (1984) determined dates of tree invasion into high elevation grasslands in the Jemez Mountains of New

Mexico. While an occasional tree became established prior to 1800 in these grasslands, from 1800 to 1910 not a single tree became established, coinciding with a period of very high sheep numbers. Allen concluded that this was due either to high grazing pressure, or high incidence of fire started by shepherds. However, the abrupt and steady decline of sheep numbers in this area, starting around 1919, coincides with the beginning of the earliest invasions by conifers of Jemez Mountain grasslands (Allen 1984). High intensity, severity, and duration sheep grazing, by large numbers of sheep was replaced by relatively lower intensity and severity grazing by lower numbers of cattle. Allen (1984) mentions that the direct effects of sheep eating and trampling conifer seedlings also created areas of bare mineral soil where new conifer seedlings could become established, once the grazing pressure was reduced. Tree establishment is also facilitated by grazers' removal of competing biomass, which also lowers fire frequencies through removal of fine fuels (Allen 1984). In another study from the North Rim of Grand Canyon National Park in Arizona, Moore and Huffman (2004) found similar evidence of early 20th century tree invasion into montane grasslands following late 19th century cattle reductions (see Figures 11-5 through 11-7). While some tree species became established into grasslands in the early- and mid-1800s, 91% of all trees became established after the mid-1930s. This period coincides with the exclusion of cattle from the National Park (Vankat 2006).



Figure 11-5. Three photos in a series from Pleasant Valley on the North Rim of Kaibab National Forest. This initial photo was taken in 1929 showing young aspen and spruce trees invading a subalpine grassland. Photos courtesy of USFS North Kaibab RD.



Figure 11-6. Six years later in 1935.



Figure 11-7. 1942 photograph showing dense encroachment of subalpine grassland by spruce.

Brown (1994) claimed that overgrazing of montane grasslands commonly results in changes in plant composition from grass and forbs to perennial scrub mixtures of sage (*Artemisia*), rabbitbrush (*Chrysothamnus*), groundsel (*Senecio*), and others. He further claimed that if drying, trampling, and erosion of the deeper soils continue, the meadows themselves may eventually be replaced by forest (Brown 1994) (see Figures 11-5, 11-6, and 11-7).



Figure 11-8. Montane grasslands on west side of San Francisco Peaks, near Flagstaff, AZ. Circa 1880 photograph shows evidence of grazing in foreground, but wide open grasslands with scattered individual trees in background. Fern Mountain is marked with arrow. Photo courtesy of USDA-USFS.



Figure 11-9. Same area as above photograph, only 100 years later, circa 1980. Note heavy invasion and infill by conifers. Fern Mountain is marked with arrow. Photo courtesy of USDA-USFS.

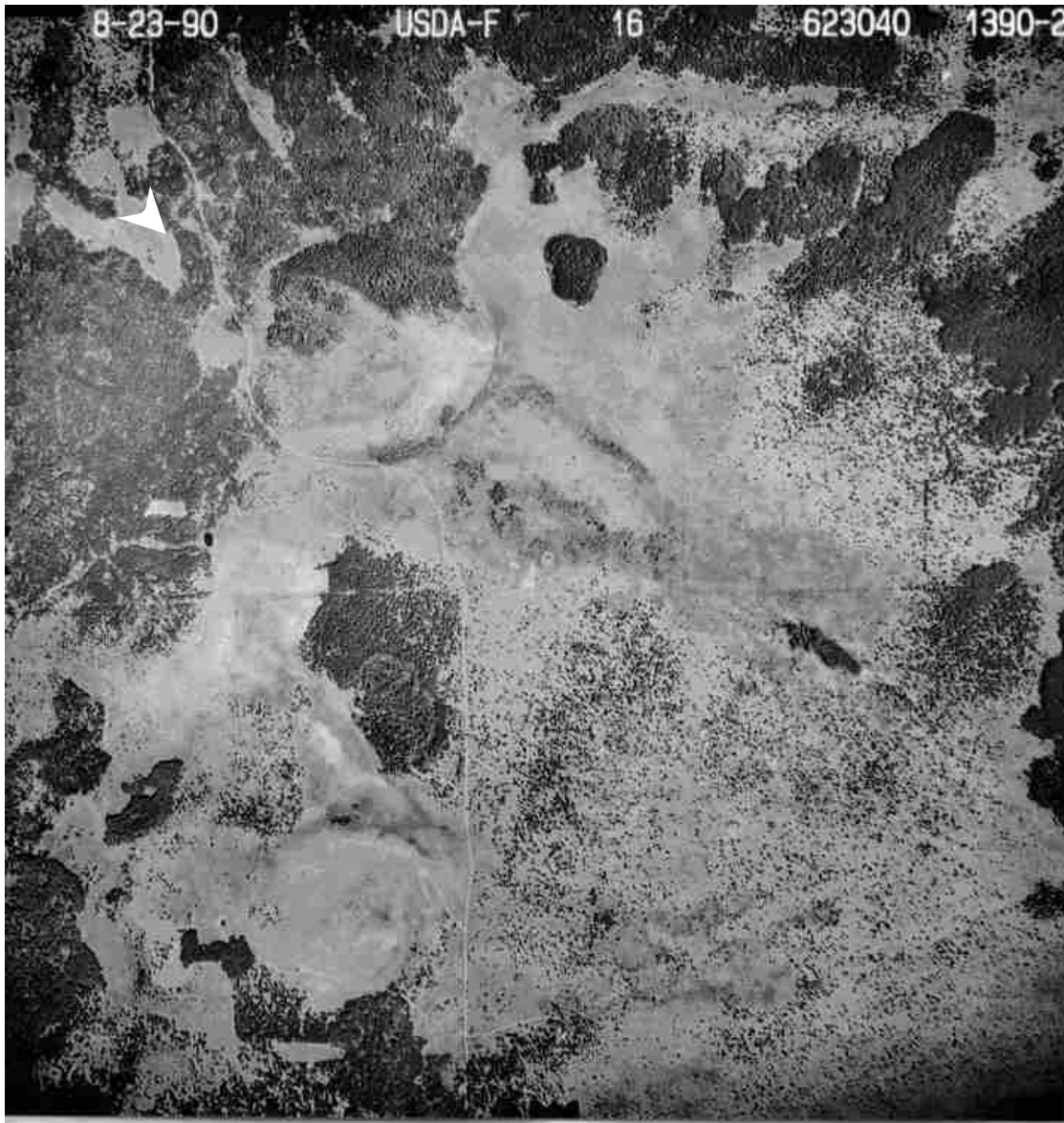


Figure 11-10. Aerial photograph from 1990 of area in previous two photos (11-8 and 11-9), with white arrow in upper left corner indicating approximate location of camera, looking SE. Note high density of invading and colonizing conifers within large expanse of montane grassland, locally referred to as Hart Prairie.

Other effects of cattle grazing mentioned by White (2002) include soil erosion, reduction in abundance of palatable plants, introduction and spread of exotic plants, and changes in fire regimes. Other herbivores, such as native ungulates, rodents, and invertebrates also affect high elevation grasslands, but their impacts are not well understood.

Silviculture –Not Applicable.

Fragmentation – We found no studies that documented fragmentation as an important ecological determinant for the high elevation grassland vegetation types in the Southwest.

Mining – We found no studies that documented mining as an important ecological determinant for the high elevation grassland vegetation types in the Southwest.

Fire Management – The disruption of historical fire regimes by introduced grazing animals has been well documented for southwestern ecosystems, and high elevation grasslands were utilized as summer range for large numbers of sheep and cattle (Allen 1984). Unintentional fire suppression, initiated in the early 19th century through grazing by sheep and cattle, transitioned in the early 1900s to active fire suppression through the construction of fire lines and roads in the mid-20th century. Concerted efforts with fire brigades, ground crews, and air tankers, functioned as the primary mechanisms for excluding fire from southwestern forests (Covington and Moore 1994, Swetnam and Baisan 1996). Fire exclusion was very successful initially, but subsequent accumulation of fuels, through litter-fall and logging debris accumulation, and development of fuel “ladders” of live and dead trees that are capable of conveying surface fires in to the crowns and canopies of forests (Covington and others 1994) made fire suppression more difficult. As the number and size of fires has increased over the last century (Dahms and Geils 1997), the emphasis on use of prescribed or “fire-use” fire has increased within land management agencies, with varying levels of success due to complex social, economic, and climatic factors (Zimmerman 2003). Fire suppression activities probably became effective in montane grasslands around 1930, judging by the success of tree invasions (Allen 1984, Binkley and others 2004, Moore and Huffman 2004), and fire suppression continues to the present (see Figures 11-11 and 11-12).

Exotic Introductions (Plant & Animal) – Kentucky bluegrass (*Poa praetensis*) was introduced into many high elevation grasslands by early settlers and their grazing animals (Allen 1984). Dick-Peddie (1993) reported that under heavy grazing, native bunchgrasses tend to be replaced by Kentucky bluegrass, but fescue meadows rested from livestock in the Pecos Wilderness reverted to Thurber Fescue and other native grasses, with Kentucky bluegrass and rhizomatous sedges between clumps. Reversion occurs within 2-4 years at elevations of 10,000 to 11,000 ft. In many areas, the Kentucky Bluegrass meadows seem relatively stable, possibly due to soil change; it is unlikely that these meadows will completely return to their pre-livestock composition. The introduced Rocky Mountain elk (*Cervus elaphus nelsonii*) may exert strong selection pressure against palatable grass species, especially in wet meadow grasslands (White 2002).

Synthesis – Climate is not considered to be a strong determinant of ecological change of high elevation grasslands (Allen 1984, White 2002) although some authors have noted trends in tree invasion correlated with inter- and intra-annual variation in temperature and precipitation (Dyer and Moffett 1999, Coop and Givnish 2007b). The major anthropogenic processes that cause changes in high elevation grasslands include grazing by sheep and cattle, and concomitant reduction in fire frequency, both of which have led to increased invasion by coniferous trees and aspen (*Populus tremuloides*). Early settlers and their grazing animals also have likely introduced one or more exotic grass species, of which Kentucky bluegrass has proven to be successful in out-competing and replacing native perennial bunchgrasses in wet meadows and other valley bottom grasslands. Long-term, chronically low to high levels of grazing by introduced herbivores may have resulted in shifts in species composition away from palatable species to dominance by less palatable grasses, forbs, and shrubs, although the specific mechanisms and community composition changes have not been well described.

11.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation - We found no studies that documented the effects of human disturbance on the patch composition of high elevation grassland vegetation.

Overstory – Both Allen (1984) and Moore and Huffman (2004) documented tree invasion of high elevation grasslands as a result of grazing and fire suppression since the 1800s (see Figures 11-11 and 11-12). In Moore and Huffman’s (2004) study of Grand Canyon National Park on the North Rim, of 3,481 live and dead trees sampled, 52% were quaking aspen (*Populus tremuloides*), 20% were spruces (either *Picea pungens* or *P. engelmannii*), 11% were subalpine fir (*Abies lasiocarpa*), 10% were white fir (*A. concolor*), and 7% were ponderosa pine (*Pinus ponderosa*). Tree density averaged 1,881 trees per acre, and while 62% of the trees sampled were less than 20 years old, 91% of the trees established after the mid-1930s (see Figures 11-5, 11-6, 11-7). Allen (1984) studied tree invasion of ten high elevation grasslands in the Jemez Mountains of New Mexico, and quantified tree invasion for one of the “moderately” invaded sites, Canada Bonito. Allen (1984) found 171 trees in a 2.7 acre block, and determined that modern tree invasion began in 1920 and that the majority of trees established between 1924 and 1949. Of those 107 trees, 70% were ponderosa pine, 18% were Douglas-fir (*Pseudotsuga menziesii*), 6% were limber pine (*Pinus flexilis*), 4% were aspen, and 1% was Engelmann spruce. Based on aerial photograph comparison between 1935 and 1981, Allen (1984) found a 55% decrease in grassland extent, which he attributed to a decrease in grazing pressure and fire suppression. In an adjacent area, Coop and Givnish (2007b) quantified tree invasion of higher and lower elevation montane grasslands. They found that around 63% of montane grasslands above 9200 ft. were now occupied by forest tree species, and that below 9200 ft, grasslands declined about 12%.

Understory and Herbaceous Layer – Many high elevation grasslands have been altered by either fire suppression, heavy grazing by introduced ungulates, including Rocky Mountain elk, or the introduction of exotic plants, although quantitative studies are lacking. White (2002) compared high elevation grassland composition between Early (1913 to 1915) and Modern (1997 to 1998), and found significant changes in soil surface cover, vegetation composition, and dominant species. He reported that mesic (moderately moist) montane grasslands had changed the most over the 85-year period, whereas xeric subalpine grasslands had changed the least over the same period (White 2002). Also, White (2002) reported that dominant species had shifted, and that soil surface cover provided by vegetation had declined, and that bare ground had increased.

White (2002) also reported *Importance Values* (IV) or dominance index for the top eight species that comprised the four different high elevation grasslands, again comparing species composition between the two time periods. For the **mesic subalpine grassland**, the top four Early dominant species were Arizona fescue (IV = 0.29), tufted hairgrass (0.28), mountain muhly (0.25), and pine dropseed (*Blepharoneuron tricholepis*) (0.20). The top four Modern dominant species were tufted hairgrass (0.49), Canada bluegrass (*Poa compressa*) (0.26), an exotic, Kentucky bluegrass (0.18), an exotic, and sheep fescue (*Festuca ovina*) (0.16). For the **xeric subalpine grassland**, the top four Early dominant species were Arizona fescue (IV = 0.54), mountain muhly (0.41), pine dropseed (0.35), and sheep fescue (0.12), while in the Modern period, dominance changed to muttongrass (0.32), Arizona fescue (0.31), pine dropseed (0.29), and mountain muhly (0.27).

For the **mesic montane grassland**, White (2002) reported dominance for the Early community from annual muhly (*M. minutissima*) (0.52), Rocky Mountain Iris (0.28), little hogweed (*Portulaca oleracea*) (0.22), and blue grama (*Bouteloua gracilis*) (0.21). The Modern community was dominated by Kentucky bluegrass (0.37), spike muhly (*M. wrightii*) (0.20), trailing fleabane (*Erigeron flagellaris*) (0.16), and Canadian bluegrass (0.14). And the **xeric montane grassland** was dominated in the Early period by blue grama (0.59), pine dropseed (0.31), matted grama (*B. simplex*) (0.27), an annual, and creeping bentgrass (*Agrostis stolonifera*) (0.25), whereas the Modern community was dominated by pine dropseed (0.30), blue grama (0.16), trailing fleabane, and prairie junegrass (*Koeleria macrantha*) (0.13). The trends in these data indicate that Early communities were dominated by perennial bunchgrasses with some dicots, and that Modern communities are also dominated by perennial bunchgrasses, but that they have a higher proportion of exotic grasses, native annuals, and dicots.

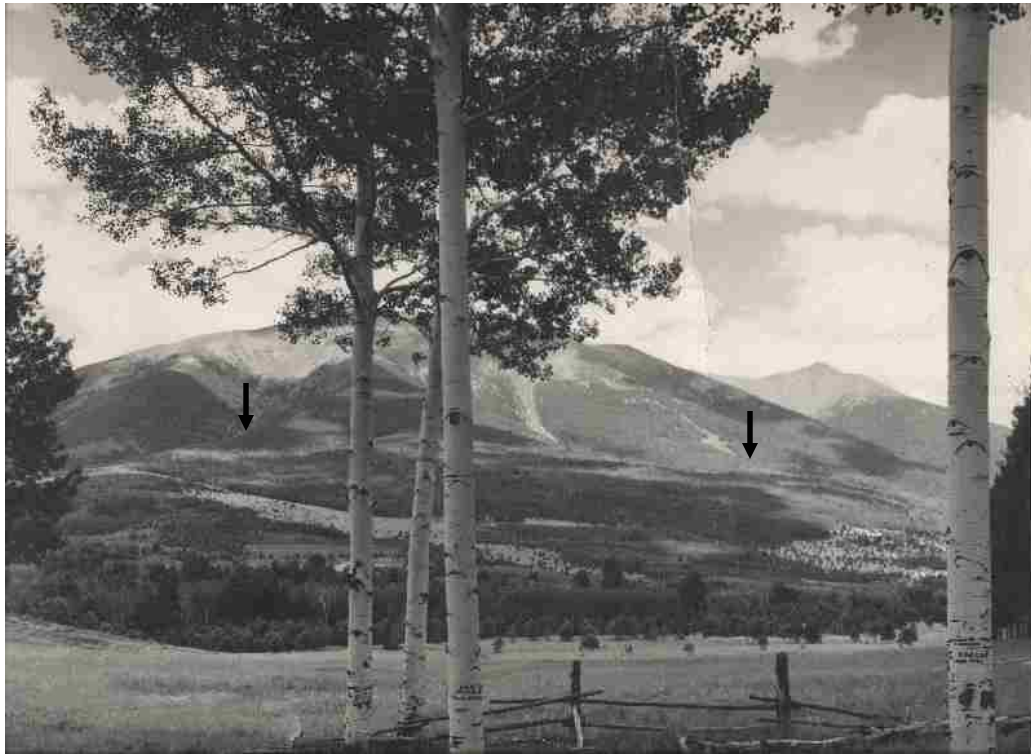


Figure 11-11. West side of San Francisco Peaks, near Flagstaff, AZ. Photo from circa 1970 shows filling in of montane meadows by bristlecone and southwestern white pine in background, and scattered ponderosa pine and aspens in the middle- and foreground. Photo by R.C. Smith.



Figure 11-12. Thirty-six years later, montane grasslands continue to close. Note almost complete closure of openings in right and left background (arrows). July 2007 photo by the author (E.B. Smith).

Patch or Stand Structure of Vegetation - We found no studies that documented the effects of human disturbance on the patch or stand structure of high elevation grassland vegetation. However, Coop and Givnish (2007b) noted that distance to forest line had some effect on rate of tree invasion into openings, thus smaller openings experienced a greater degree of tree invasion and closure.

Canopy Cover Class (%) or Canopy Closure - We found no studies that documented the effects of human disturbance on the canopy cover of high elevation grassland vegetation. However, Allen (1984) reported average cover values of 81% for slope grasslands in the Jemez Mountains (43% grasses, 31% forbs, and 7% sedges), and Warren and others (1972) found that valley bottom grasslands of Grand Canyon National Park had 35 to 100% cover with highly variable species composition.

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

Life Form – We found no studies that documented the effects of human disturbance on the life form of high elevation grassland vegetation.

Density - We found no studies that documented the effects of human disturbance on the density of high elevation grassland vegetation.

Age Structure - We found no studies that documented the effects of human disturbance on the age structure of high elevation grassland vegetation.

Patch Dispersion - We found no studies that quantified the effects of human disturbance on the patch dispersion of high elevation grassland vegetation.

Recruitment Dynamics - We found no studies that documented the effects of human disturbance on the recruitment dynamics of high elevation grassland vegetation, although it has been noted that increased grazing pressure has created openings in the vegetation canopy and soil surface (soil litter loss). These bare patches are more likely to be colonized by tree seedlings, exotics, or native weedy species, and contribute to soil loss through erosion (Allen 1984, White 2002).

Synthesis –The interaction of late 18th century and early 19th century grazing and fire suppression has led to tree invasion of many, but not all high elevation grasslands. In some areas, tree invasion appears to be correlated with changes in climate that occurred around the same time as changes in grazing practices (Dyer and Moffett 1999). Many of the valley bottom grasslands have been impacted by the change in dominance from native perennial bunchgrasses to exotic Canada and Kentucky bluegrasses. Reversion from exotic bluegrass dominated grasslands to native grasslands is possible with rest from grazing, but exotic bluegrasses and other species are likely to persist.

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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.

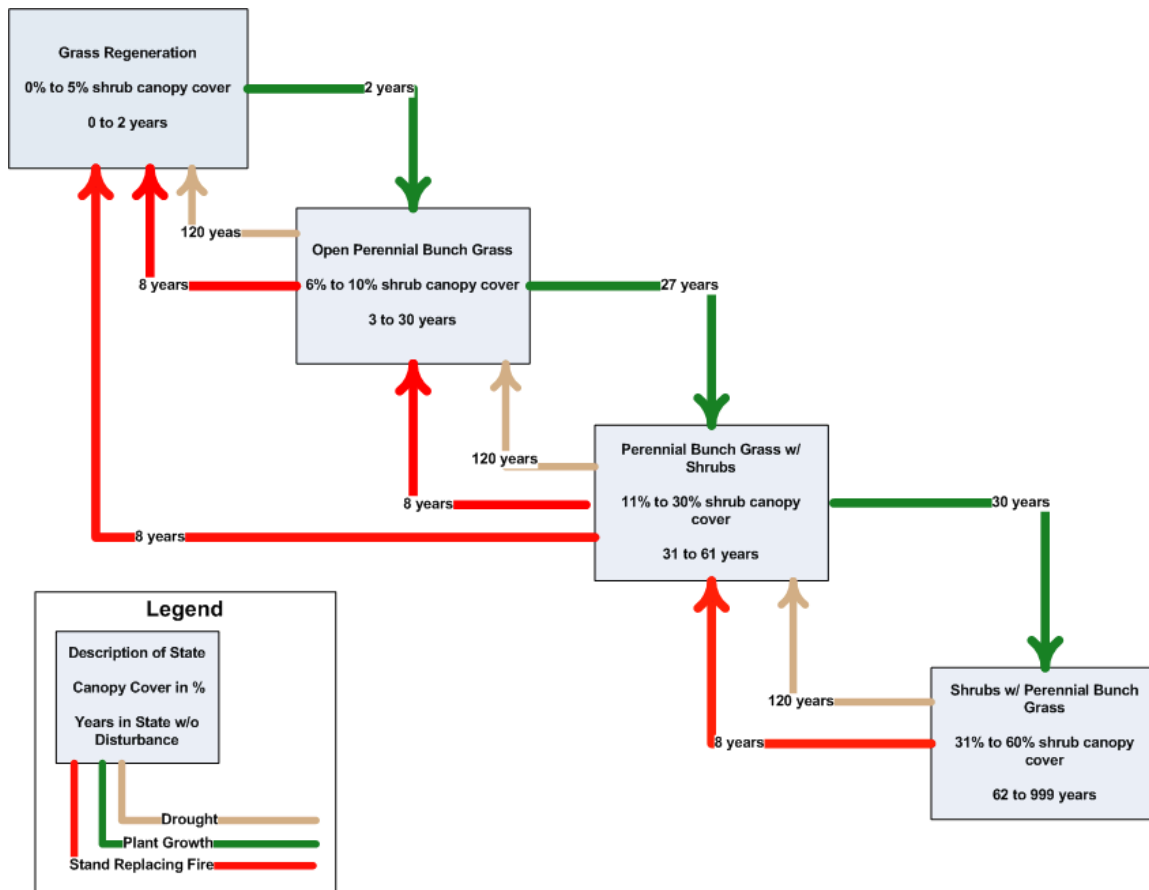


Figure 13-1. Simple grassland model used in sensitivity testing of VDDT software

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.

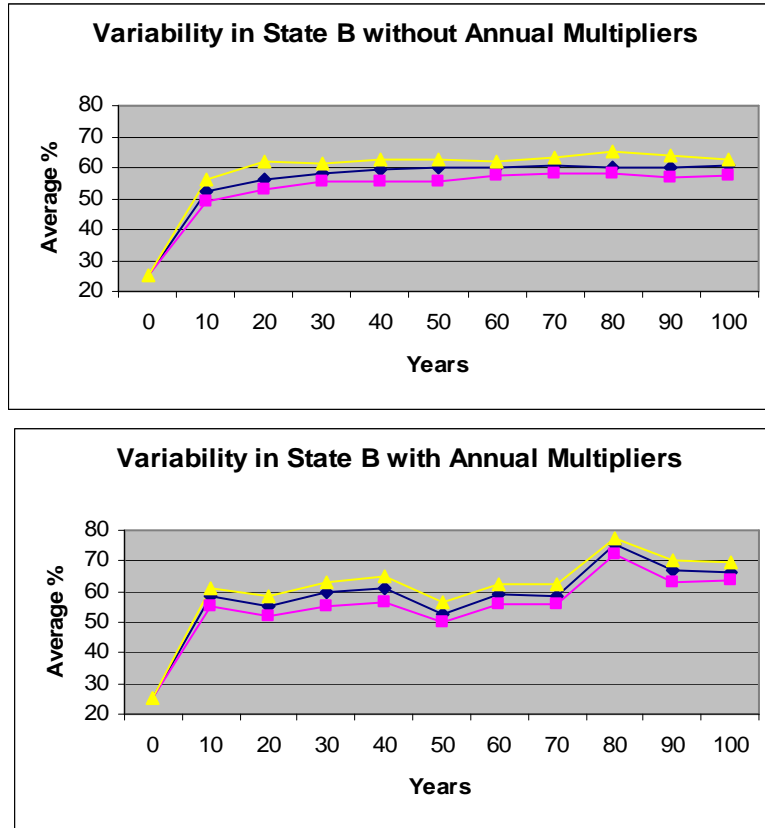


Figure 13-2. Comparison of year to year variability in state B of the simple grassland VDDT model with and without the use of annual multipliers. Maximum values in yellow, average values in blue, and minimum values in pink.

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 PNVNT 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 PNVNT 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 20 - Montane and Subalpine Grasslands Model

20.1 Montane and Subalpine Grasslands Vegetation Dynamics – High-elevation grasslands generally occur in small (< 100 ac.) to medium-sized (100 to 1000 ac) patches, and rarely in patches >1000 ac throughout the extent of spruce-fir and mixed conifer forests in the southwestern region, with subalpine grasslands occurring at slightly higher elevations than montane grasslands (Allen 1984, Dick-Peddie 1993, White 2002). In this report, we refer to both types of grasslands as high elevation grasslands, unless differentiation is supported in the literature. These high elevation grasslands occur from about 8,500 ft. to 11,500 ft. in elevation, and have been divided by several authors into two categories, “upper slope” and “valley bottom” grasslands (Allen 1984, Dick-Peddie 1993, Muldavin and Tonne 2003). The valley bottom grasslands (or meadows or wet meadows *sensu* Brown 1994) are sometimes discontinuous and occur in smaller patches, but also occur as large contiguous grasslands known as *valles*, a Spanish word for valleys. The upper slope grasslands can form large, contiguous patches, often on the southern or western flanks of mountains (Allen 1984, Vankat 2006)

Soil depth and composition indicate that grassland vegetation has dominated many high elevation sites adjacent to and embedded within forests for thousands of years (Allen 1984). Seed dispersal of coniferous tree species, mediated by wind and birds (Vander Wall and Balda 1977, Allen 1984), and asexual aspen reproduction has led to ongoing recruitment of trees in grasslands, however, frequent surface fire kept tree densities extremely low during the pre-settlement or historic period (Allen 1984, Moore and Huffman 2004, Coop and Givnish 2007a, 2007b). Livestock grazing followed by cessation of frequent fire regimes, followed by grazing release increased tree seedling recruitment during the post-settlement period, allowing many grasslands to become invaded by forest species (Allen 1984, Moore and Huffman 2004, Coop and Givnish 2007a, 2007b). However, tree encroachment of high elevation grasslands is highly variable in both space and time, with some areas exhibiting no encroachment (Moir 1967, Coop and Givnish 2007a), and other areas exhibiting 3 to 65% increase in conifer coverage between 1935 and 1996 (Coop and Givnish 2007b). Many factors interact to control the level of encroachment, including soil type, minimum summer temperature, precipitation and soil moisture level, grazing level, fire regime, elevation, aspect, slope inclination, bare soil patch size, and proximity to trees as seed source (Allen 1984, Moore and Huffman 2004, Coop and Givnish 2007b, Potter and Tierny 1985).

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 vegetation type (Figures 20-1 and 20-2). Additionally, we used information from the state and transition models to develop quantitative Vegetation Dynamics Development Tool (VDDT) models. The VDDT software allows the user to model succession as a series of vegetation states that differ in structure, composition, and cover and to specify the amount of time it takes to move from one vegetation state to another in the absence of disturbance. Various disturbance agents affecting the movement of vegetation between states can then be incorporated (e.g., surface fires, stand-replacing fires, grazing, insect outbreaks). By varying the types and rates of disturbance across the landscape, the effects of different management treatments, such as wildland fire use, fire suppression, prescribed burning, grazing practices, and mechanical fuel treatments, on future vegetation can be investigated. While VDDT models can be used to “game

play” with different management scenarios, the models we ran in this analysis only include states and transitions for which there is published information to support their inclusion within the model. We discuss model parameters, output, and analysis below (Tables 20-1 and 20-2).

Historic Southwest High Elevation Grassland

State and Transition Model

July 2007

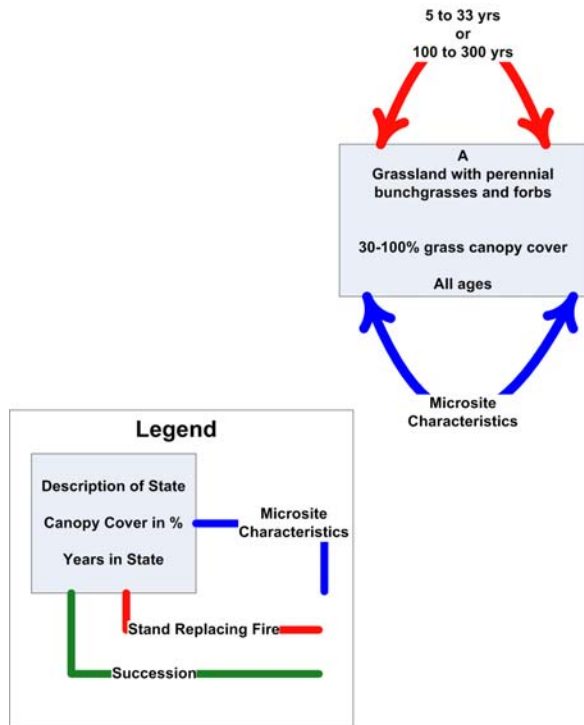


Figure 20-1. Conceptual Historic state and transition model for the montane and subalpine grassland vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Current Southwest High Elevation Grassland

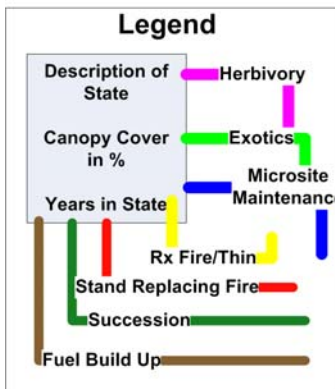
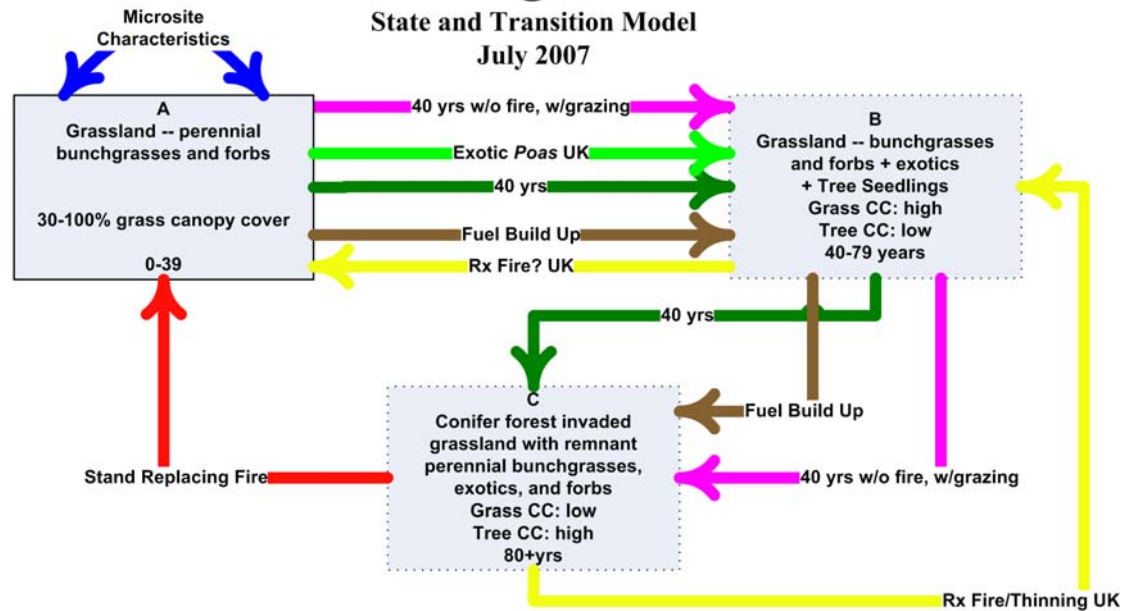


Figure 20-2 Conceptual Current state and transition model for the montane and subalpine grassland vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation. Dashed outlines represent states which may have been uncharacteristic for the historic period.

20.2 Model Parameters

In Tables 20-1 and 20-2 below, we describe the parameters included or not included within the Historic and Current VDDT models, as well as the sources of information and any assumptions used to create model parameters.

Table 20-1 Identification of historic transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Microsite Characteristics	Not used in model	Allen 1984, Moore and Huffman 2004, Coop and Givnish 2007b	There are large differences in the ‘invasibility’ of grasslands, largely controlled by climate, but also controlled by several site gradients.
Stand Replacing Fire	5 to 33 yrs for lower elevation 100 to 300 yrs for higher elevation	Swetnam and Baisan 1996, Touchan and others 1996, Fule and others 2003, Heinlein and others 2005, Swetnam and others 2005 Grissino-Mayer and others 1995, Swetnam and Baisan 1996, Fule and others 2003, Swetnam and others 2005, Vankat 2006	We assumed that montane grasslands embedded or adjacent to mixed conifer forests had the same fire return interval or fire rotation. However, surface fire in forested systems functions as stand replacing fire in grassland systems, because all or most of the standing biomass was assumed to be removed, including most of the tree seedlings and saplings. At higher elevation, fire rotations in adjacent spruce-fir forests were longer, although grasslands may have supported more frequent fires if they were contiguous with lower elevation forests.
Plant Growth	One ecological state for model (no transitions)	Allen 1984, Moore and Huffman 2004, Coop and Givnish 2007a, 2007b)	Most authors studying high elevation grasslands have noted the presence of a few, scattered, pre-settlement coniferous trees, thus mechanisms have existed for their recruitment into grassland systems. We assumed that tree recruitment occurred at the less frequent end of fire rotation values, i.e., when stand replacing fire was infrequent enough to allow tree seedlings to grow to maturity, or within very small, protected micro-sites. We assumed that the spatial scale of these individual trees was

			not sufficiently large to constitute an independent patch that would be maintained through time, and that individual trees were part of some grasslands. Their persistence indicates that individual trees were able to escape mortality at very low frequency through fire resistance mechanisms.
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Table 20-2. Identification of Current transitions, frequency of transitions, sources of information used, and assumptions used to develop the frequency of transitions and its effect on vegetation included in the VDDT models.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Microsite Characteristics	Not used in model	Allen 1984, Moore and Huffman 2004, Coop and Givnish 2007b	There are large differences in the ‘invasibility’ of grasslands, largely controlled by climate, but also controlled by several site gradients.
Plant Growth	40 years	Ahlstrand and others 1980, Allen 1984, Wyant and others 1986	During the post-settlement period, tree encroachment began 30-40 years after cattle grazing and fire cessation, leading to the formation of a new ecological state that included more tree species. Several encroaching tree species become resistant to cambium scorching after attaining an age of 15 to 40 years.
Fuel Build Up	Once every year.	Covington and Moore 1994, Swetnam and Baisan 1996, Allen and others 2002, Fule and others 2003	Several authors have documented the cessation of surface fires around 1880, which has led to the accumulation of fuels.
Stand Replacing Fire	Once every 100 years	Cessation of surface fires and accumulation of fuels and development of fuel ladders has led to an increase in the frequency of stand replacing fires (Covington and Moore 1994, Swetnam and Baisan 1996, Covington and others 1997, Allen and others 2002).	We based our estimate of fire on fire scar data. Specifically, regional fire scar data shows drastic declines in fires from 1900 to present. Given these data, we estimated a fire occurrence of 1 in the last 100 years during the current period. This frequency may have regional variation based on site characteristics, sources of ignition, and suppression success.
Non-native Grazing	P=1 for all areas	Allen 1984	Grazing by sheep and cattle, and concomitant reduction in fire frequency have led to increased invasion of high elevation grasslands by coniferous trees and aspen. Invasion followed an approximately 40-year lag period, during which time grazing practices changed in large part from sheep to cattle grazing.
Exotic Plant introduction	Not used in model	Allen 1984, Dick-Peddie 1993, White 2002	Canada and Kentucky bluegrass and other plant species were introduced into many high elevation grasslands by early settlers and their grazing animals. This factor

			primarily affects understory composition and habitat quality. However, no transition probability could be discerned from the literature.
Prescribed fire and mechanical tree thinning.	Not used in model		Mechanical removal of trees and prescribed burning may be effective in restoring grasslands, but this work has not been well documented.

20.3 Results – Results of the Historic high elevation grassland model indicate that 100% of the landscape existed in the pure grassland state over the 900-year modeling period (Table 20-3).

The Current high elevation grassland model, which was run for 120 years following the Historic conditions, had very different results from the Historic model (Table 20-4). Native grassland (State A) only occupied 23 to 28% of the landscape after 120 years of fire suppression, grazing, and exotic introductions. Grassland with low density of encroaching trees (State B) occupied 18 to 20% of the landscape, and conifer forest (State C) had invaded 53 to 58% of the landscape. From these model runs, 71 to 78% of the high elevation grassland landscape is anomalous to the historic conditions.

Table 20-3 Results for the Historic High Elevation Grassland VDDT model, reported as the 900 year average for the percent of the modeled landscape in each state. As a one box model, all of the high elevation grassland is assumed to have been in the single state, A regardless of fire frequency.

Fire Return Interval (FRI) Modeled	Model Output	Grassland with perennial bunchgrasses and forbs A Open
ALL	Average	100.0

Table 20-4 Results of the Current High Elevation Grassland VDDT model, reported as the 120 year end value for average, minimum, maximum, and average standard deviation of the percent of the modeled landscape in each state.

Fire Return Interval Modeled	Model Output	Grassland with forbs A Open	Grassland with exotics and scattered trees B Open	Conifer Forest with remnant perennial bunchgrasses and forbs C Closed
ALL	Average	25.8	19.0	55.2
	Minimum	23.3	17.7	53.3
	Maximum	27.8	20.2	57.8
	Standard Deviation	1.5	0.6	1.5

20.4 Discussion – These modeled scenarios underscore the importance of frequent surface fire in maintaining open, high elevation montane grassland ecosystems. The increase in proportion of the landscape that is closed and susceptible to uncharacteristic, stand replacing fires is very high at more than 50%. This proportion is toward the high end of reported values of grassland encroachment for the Valles Caldera grasslands studied by Coop and Givnish (2007b), but close to the value of 55% grassland loss reported by Allen (1984). There are several variables that control grassland invasibility, including precipitation and temperature, and several site characteristics such as topographic position, aspect, slope inclination, soil type and depth, and disturbance

history (Allen 1984, Moore and Huffman 2004, Coop and Givnish 2007b). Although it could not be quantified for this model, several areas have been invaded by nonnative plant species, including species of *Poa*, *Bromus inermis* (smooth brome), *Taraxacum officinale* (common dandelion), and other plant species that change habitat suitability for wildlife species (Allen 1984, Dick-Peddie 1993, White 2002). The simplistic model presented here incorporates the factors that have been quantified in the literature to date, and as more data become available, the model can incorporate that information to reflect the level of sophistication of our knowledge.

20.5 Montane and Subalpine Grasslands References

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