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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Semi-Desert Grassland

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Madrean Encinal

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Interior Chaparral

Schussman, Heather. 2006. Historical Range of Variation and State and Transition Modeling of Historical and Current Landscape Conditions for Interior Chaparral of the Southwestern U.S. Prepared for the U.S.D.A. Forest Service, Southwestern Region by The Nature Conservancy, Tucson, AZ. 24 pp.

Madrean Pine-Oak

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

Mixed Conifer

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

Ponderosa Pine

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

Spruce-Fir

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

Aspen

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

Alpine Tundra

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

Montane Grassland

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

Pinyon-Juniper

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

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1.1 Introduction

Definition of HRV-

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

1.2 Methods Used in Determining HRV

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

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

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

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

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

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

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

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

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

US Forest Service Region 3	Albuquerque, NM	Sheila Poole	Some	alpine-tundra, aspen, interior chaparral, Madrean encinal, Madrean pine-oak, mixed conifer, montane grasslands, pinyon-juniper, riparian, semi-desert grasslands, spruce-fir	
US Forest Service unpublished report "*Wood plenty, grass good, water none*" by Harley Shaw	n/a	Harley Shaw	Yes	pinyon-juniper, semi-desert grasslands	Photographs taken from Harley's manuscript that will be published in the near future by the RMRS.

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

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

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

- We attempted to get as many historical photographs (vs. current day) as possible, although we were limited by availability

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

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

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

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

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

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

It is important to note some key caveats regarding the data sets. The percent of variation in the cool season precipitation record explained (R2 value) by Ni and others (2002) reconstruction varies for each climate division and should be considered when evaluating results (Table 1-4) (CLIMAS 2005 http://www.ispe.arizona.edu /climas/research / paleoclimate/product.html). Similarly, the Cook and others (1999) reconstructions are based on anywhere from 5 to 9 tree chronologies with less certainty in the reconstruction occurring with fewer chronologies (Table 1-5). Additionally, information used to build both reconstruction models comes from upper elevation pine species which should be considered when extrapolating these data to lower elevation warm season dominated vegetation types or areas. Even with the above mentioned constraints, these climate data give an unprecedented regional look at historic climate conditions throughout the Southwest.

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

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

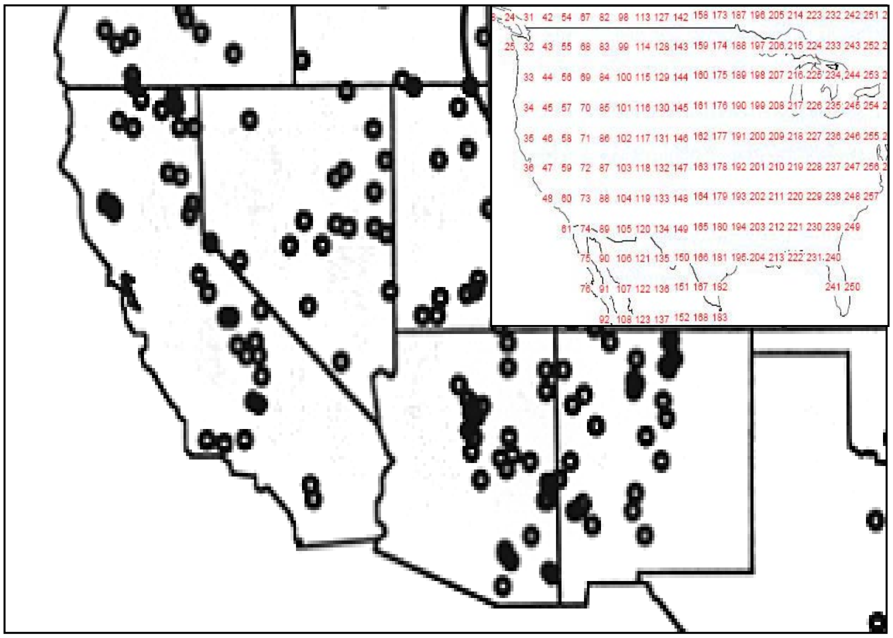
Table 1-5. Number of tree chronologies used in climate reconstructions for each PDSI grid point location for the Southwest.

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

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

Results - A comparison of the percent of dry and wet winter precipitation years, for the 15 climate divisions that span Arizona and New Mexico, showed a pattern of 19% of the years, between year 1000 and 1988, classified as severe drought or extremely wet years, 11% classified as drought years, 8% classified as wet years, and 43% classified as normal years (Figure 1-2 and Appendix 1- Table 1.1 and Figures 1.1 to 1.15). The long-term winter precipitation averages for each climate division range from 2.4 to 9.8 inches/yr. Comparisons of the 8 summer PDSI locations showed the pattern of 11% of the years classified as severe and extreme drought, 27 % classified as moderate and mild drought, 38% classified as near normal and incipient wet and dry spells, 20% classified as slightly or moderately wet, and 5% classified as very and extremely wet years (Table 1-, Figure 1-3, and Appendix 1 - Table 1.2 and Figures 1.16 to 1.23). Overall there is little regional variability in the percent of dry and wet years for either the winter precipitation or summer PDSI data sets. Of the regional variability that is present, the majority of the variation occurs within the winter precipitation data set between severe drought and drought years. For example, New Mexico climate divisions 2, 3, and 6 had fewer severe drought years than the average, but had higher drought years.

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



1a.

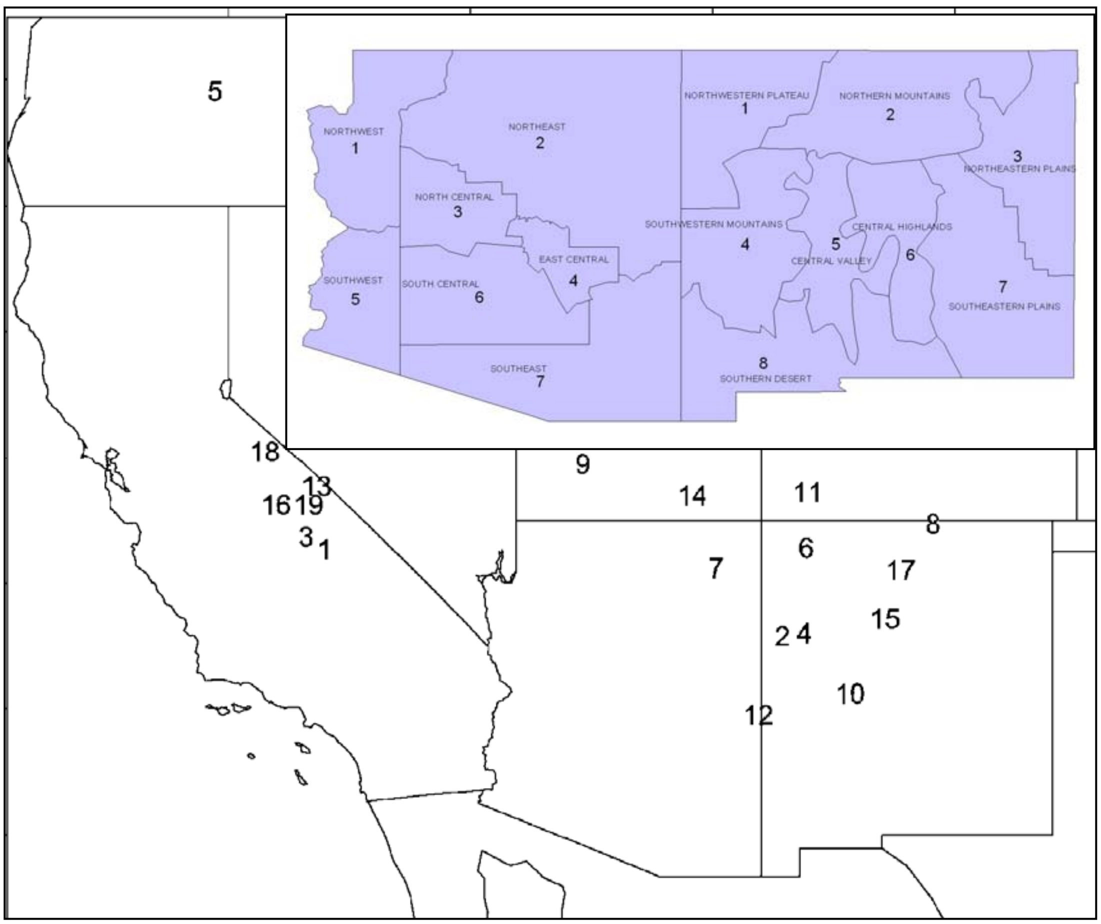


Figure 1-1. Identification of tree chronology locations for both the PDSI (1a taken from Cook and others 1999) and winter precipitation (1b taken from Ni and others 2002) data sets, as well as PDSI grid point locations and climate division boundaries.

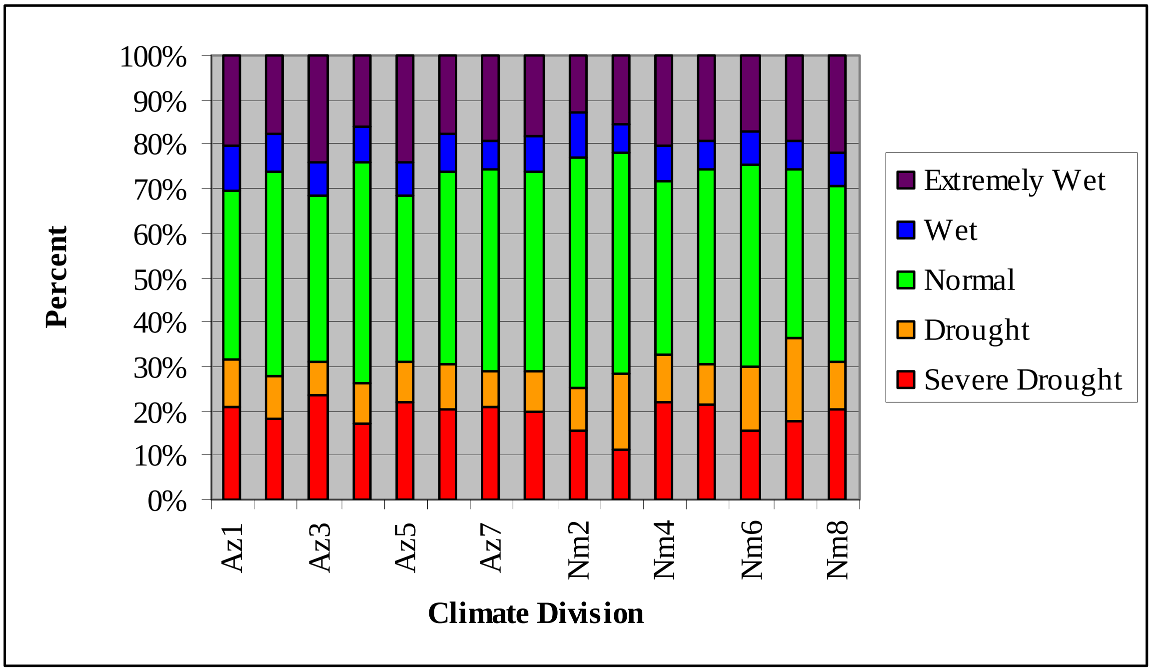


Figure 1-2. Comparison of the percent of years in all year types for all climate divisions in the Southwest.

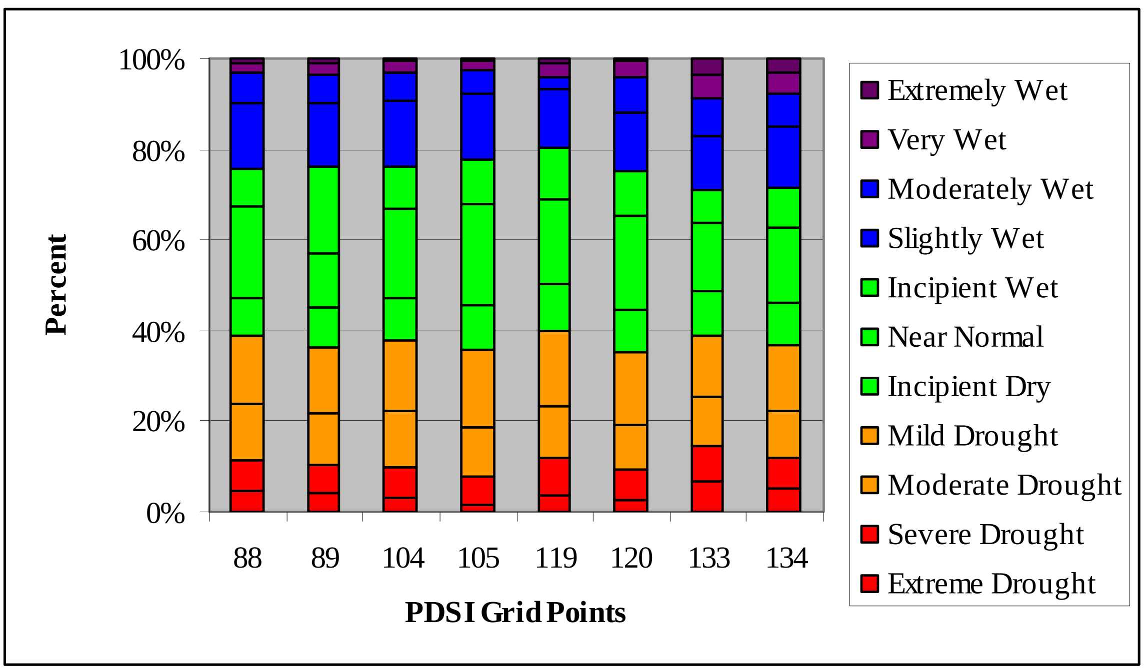


Figure 1-3. Comparison of the percent of years in all year types for all PDSI grid locations in the Southwest.

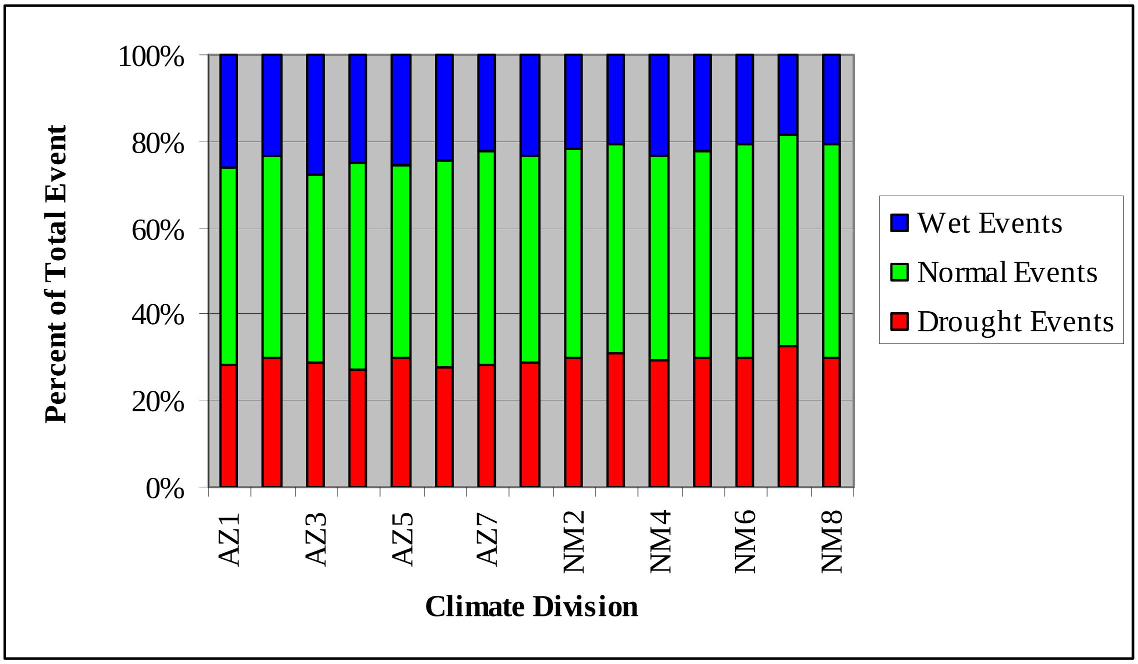


Figure 1-4. Comparison of the percent of events classified as drought, normal, and wet events for all climate divisions in the Southwest.

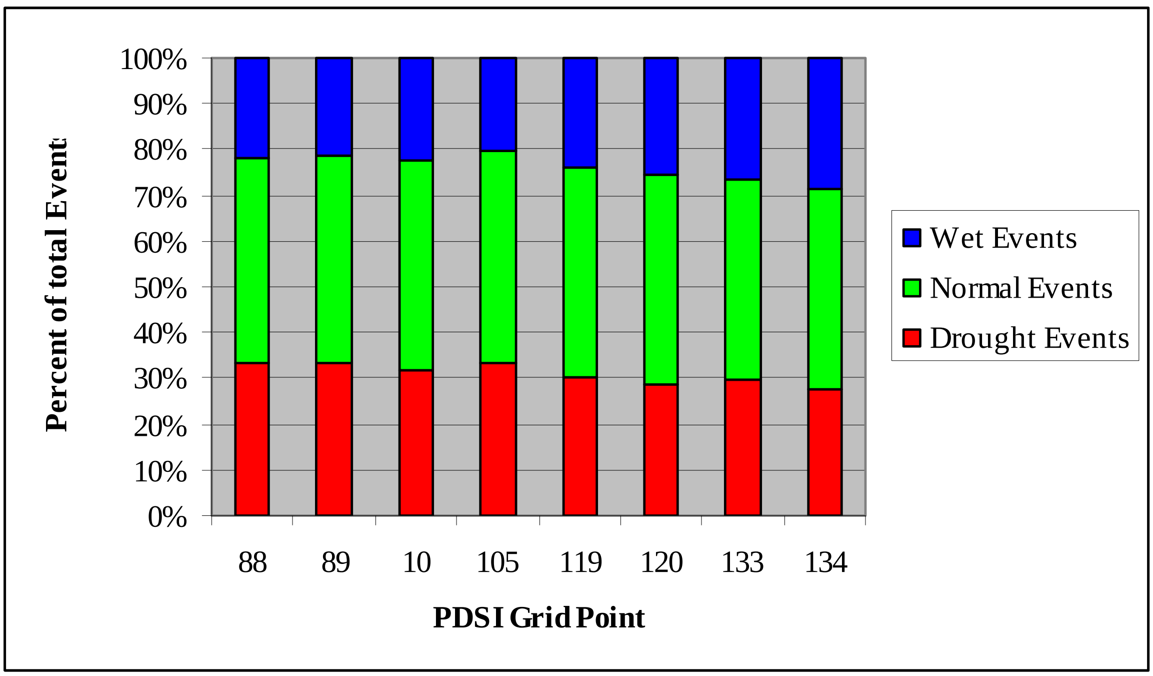


Figure 1-5. Comparison of the percent of events classified as drought, normal, and wet events for all PDSI grid locations in the Southwest.

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

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

Discussion - For both Arizona and New Mexico, most areas have experienced drought and wet events of greater magnitude than the regional range of magnitudes experienced in the 1950's and 1980's. The magnitude and pattern of events in this analysis are in agreement with other climate assessments for the Southwest (Cook and others 1999. Ni and others 2002; Meko and others 1995; Salzer and Kipfmueller 2005; Stahl and others 2000). Specifically, high magnitude and/or persistent drought (1128 to 11160, 1584 to 1592, and 1776 to 1792) or wet conditions (1304 to 1360 and 1904 to 1920) identified in this analysis coincided with warm/dry or cool/wet periods documented for the southern Colorado Plateau, by Salzer and Kipfmueller's (2005). Additionally, the 16th century megadrought has been documented to have coincided with the abandonment of "a dozen" pueblos in New Mexico (Stahle and others 2000).

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

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

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

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

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

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Chapter 2 - Semi-Desert Grassland

2.1 General Description

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

2.2 Historic Range of Variation of Ecological Processes

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

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

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

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

Disturbance Processes and Regimes

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

Climate – See Chapter 1, climate analysis section.

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Windthrow - Not an applicable category for a grassland system

Avalanche - Not an applicable category for a grassland system

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

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

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

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

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

2.3 Historical Range of Variation of Vegetation Composition and Structure

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

Overstory –

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

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

Shrub species:
Not mentioned by name

Perennial grass species:
Not mentioned by name

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

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

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

Perennial grass species:
Not referred to by name

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

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

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

Perennial grass species:
Not referred to by name

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

4) Southeastern Arizona (Turner and others 2003)

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

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

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

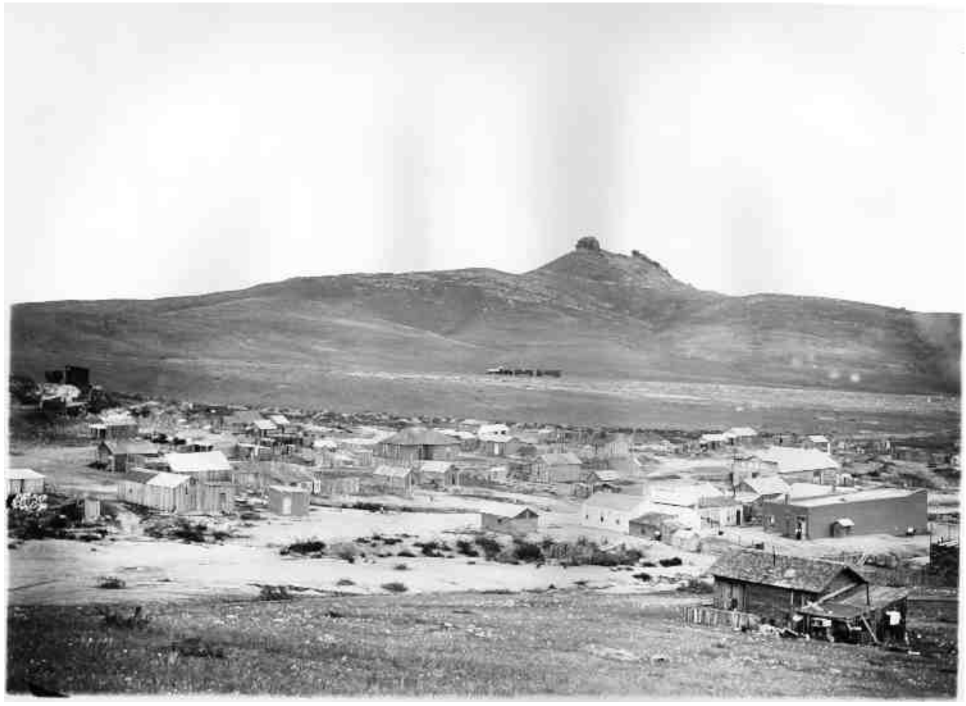


Figure 2-1. 1890's grassland photos taken near Lake Valley, New Mexico. Both photographs depict low shrub cover grasslands (Photographs courtesy Jornada Experimental Range).

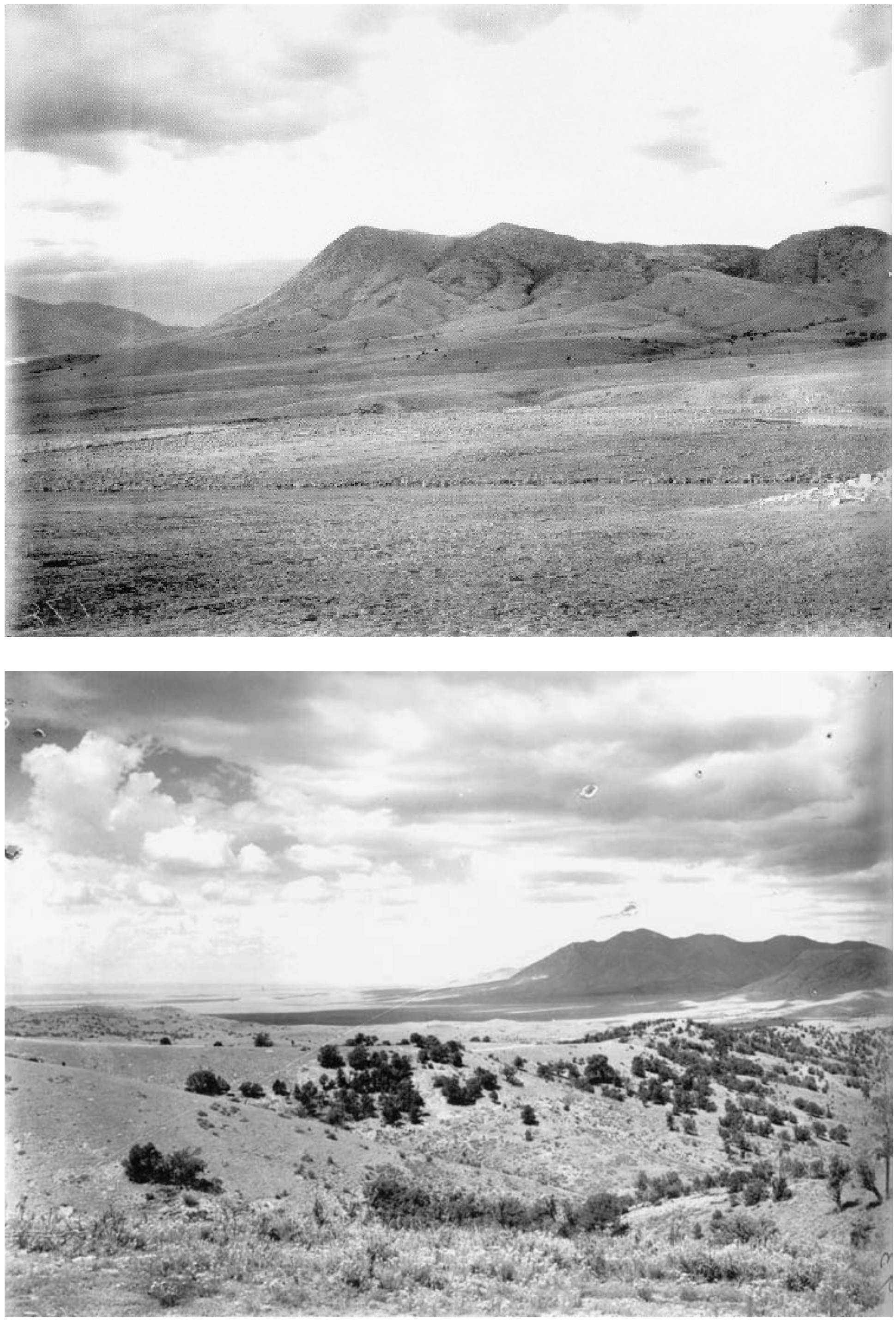


Figure 2-2. 1899 photographs of grasslands in Lincoln county showing open low shrub cover valleys with increasing shrubs and one-seed juniper on hillsides and drainages (Photographs courtesy of United States Geological Survey and Hollis Fuchs 2002)

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

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

Understory - Not an applicable category for a grassland system.

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

Patch or Stand Structure of Vegetation

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

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

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

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

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

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

Reference Sites Used

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

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

2.4 Anthropogenic Disturbance (or Disturbance Exclusion)

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

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

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

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

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

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

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

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

Silviculture - Not an applicable category for a grassland system

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

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

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

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

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

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

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

2.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

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

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

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

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

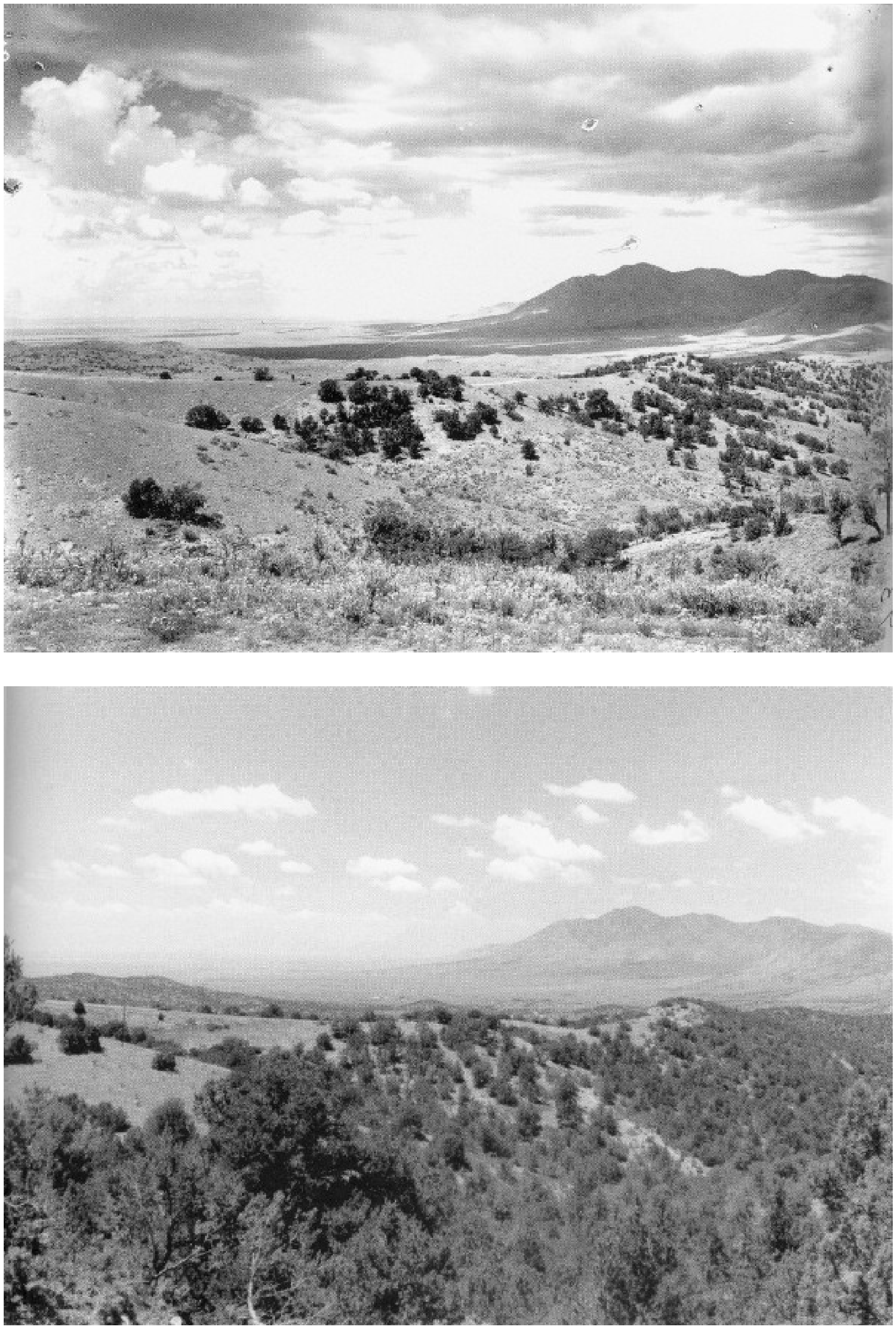
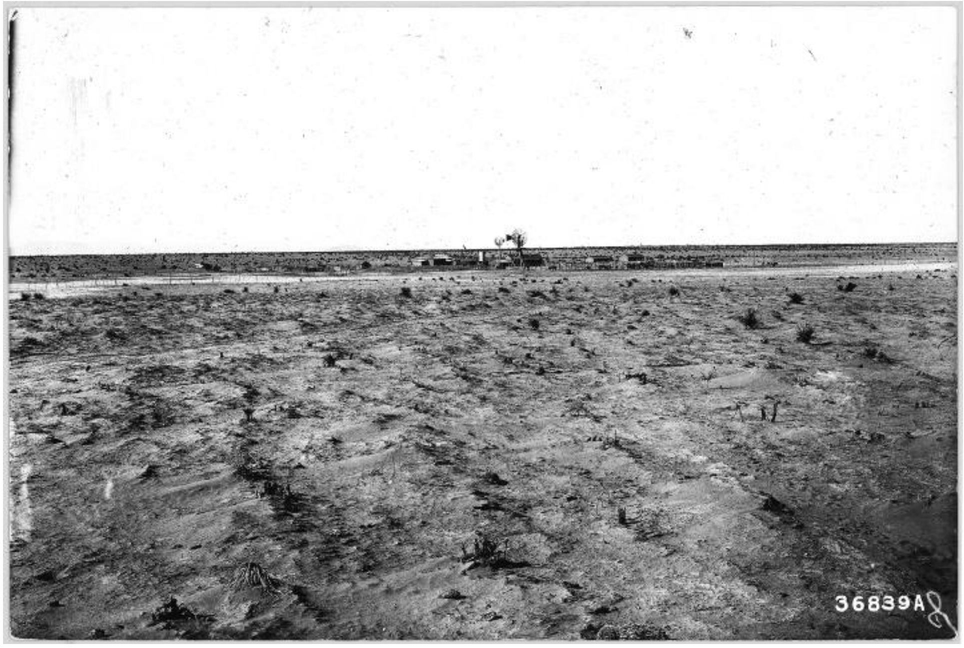


Figure 2-5. Repeat photography sequence taken in 1899 (top) and 1996 (bottom) at Fort Stanton, New Mexico. Photograph depicts expansion of juniper from the hillsides out into the open grassland valley bottom as well as increasing juniper cover on hillside (Photographs courtesy of Hollis Fuchs 2002).



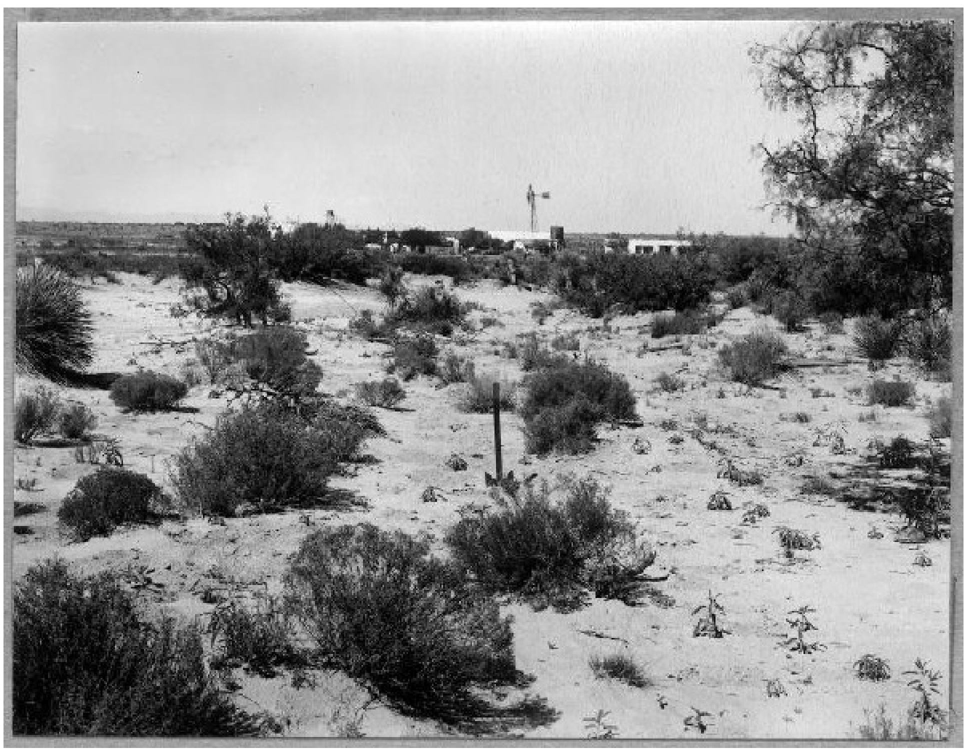


Figure 2-6. Repeat photography sequence taken in 1918 (top) and 1931 (bottom) at the Jornada Experimental Range in Las Cruces, New Mexico. Photograph depicts the transition from a open grassland to a dune shrubland. (Photographs courtesy of Jornada Experimental Range).

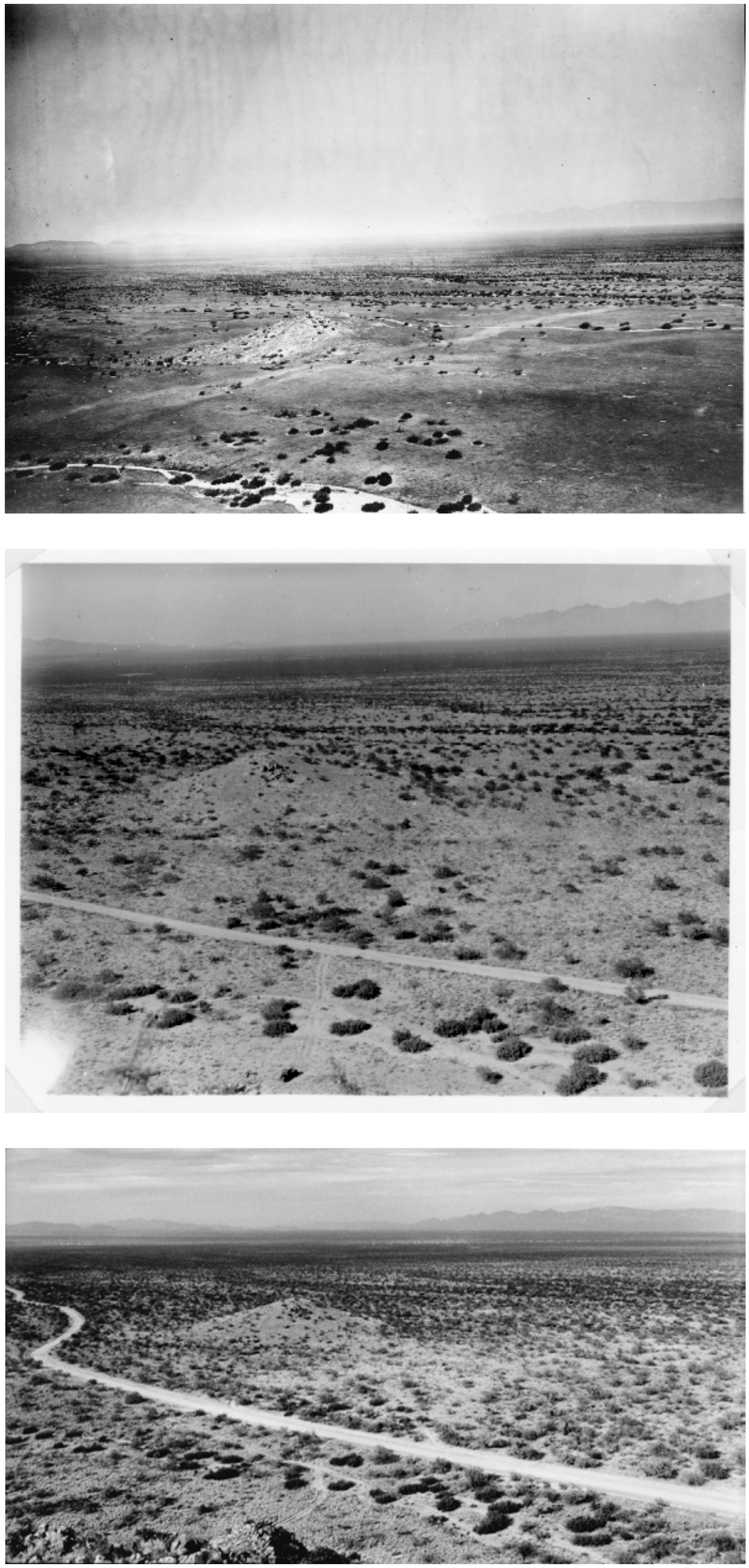


Figure 2-7. Repeat photography sequence taken in 1902 (top) and 1950 (middle) and 2000 (bottom) at the Santa Rita Experimental Range, southeastern Arizona. Photographs depict the change in cover and patch distribution of shrubs over the last 100 years on the SRER. Specifically, it is easy to see the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

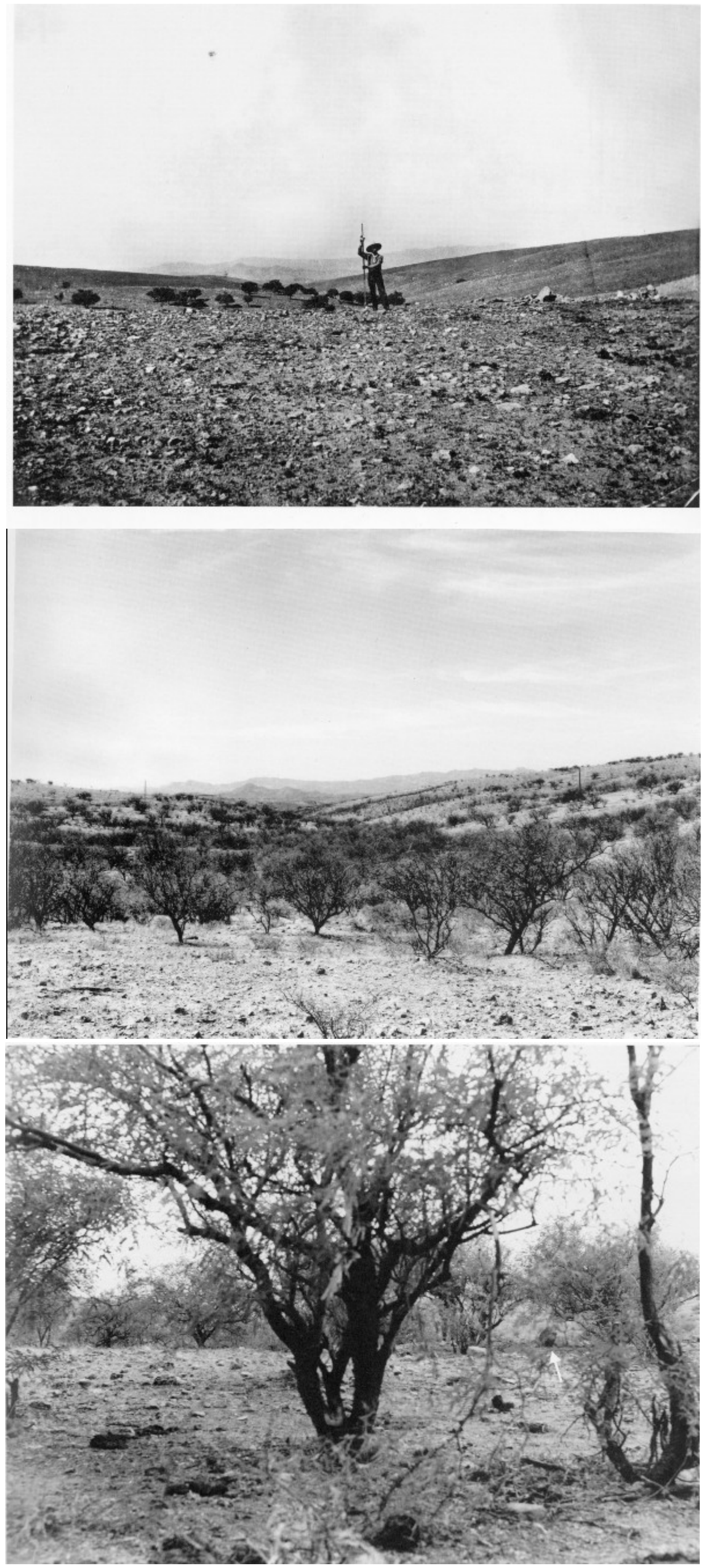


Figure 2-8. Repeat photography sequence taken in 1890 (top) and 1962 (middle) and 1996 (bottom) in Guevavi Canyon, Arizona. Photograph depicts the transition from an open grassland to a mesquite woodland. (Photographs courtesy of Unites States Geological Survey and Turner and others 2003).

Understory - Not an applicable category for a grassland system.

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

Patch or Stand Structure of Vegetation

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

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

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

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

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

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

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Chapter 13 - Vegetation Models for Southwest Vegetation

13.1 Introduction

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

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

13.2 Methodology

State and Transition Models - We defined all model states, transitions between states, and transition probabilities using information from published, peer-reviewed journal articles, as well as published conference proceedings, reports, theses and dissertations, and book chapters. We limited our search to relevant literature that came from studies of Southwest ecosystems, with a geographical emphasis on Arizona, New Mexico, and northern Mexico to ensure compatibility and relevance to Southwest ecosystems. This information is synthesized in narrative form for each PNVT in a companion document entitled "Historic Range of Variation for Potential Natural Vegetation Types of the Southwest" (Schussman and Smith 2006).

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

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

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

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

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

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

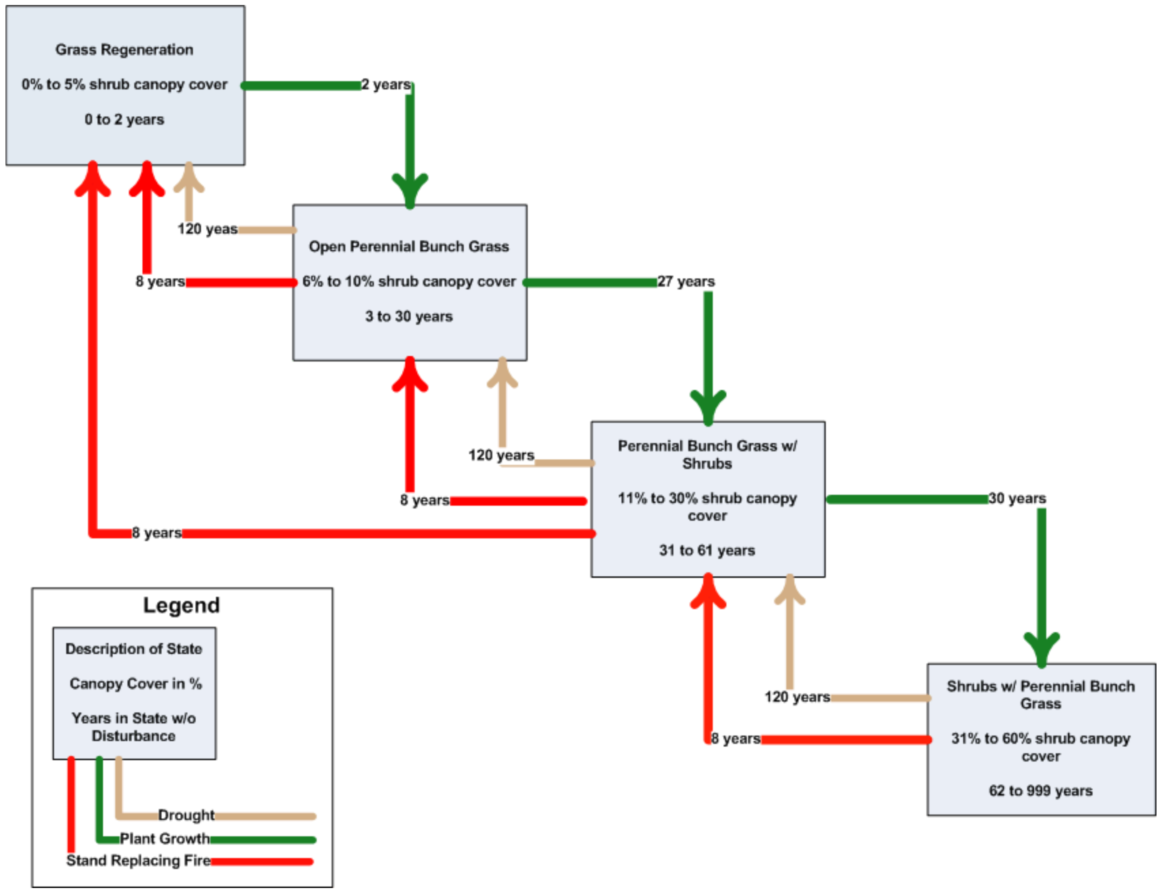


Figure 13-1. Simple grassland model used in sensitivity testing of VDDT software

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

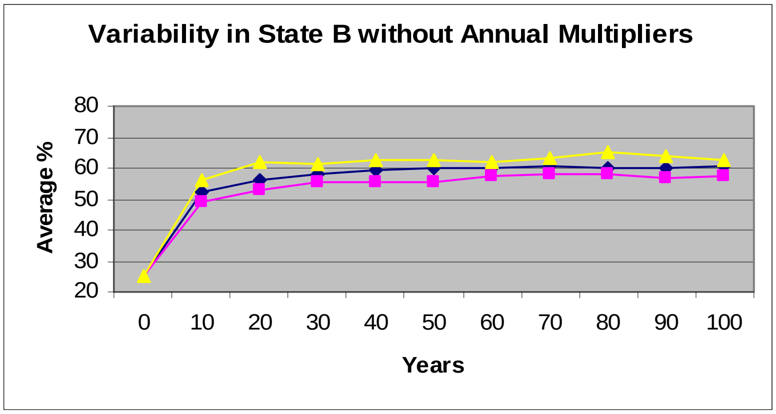
We ran the historic models for 1000 years, as this temporal span corresponds with the widest frame of reference offered by the scientific literature. Additionally, 1000 year long runs allowed for infrequent transitions, such as stand replacing fires in the spruce fir PNVT and extreme drought events in 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 (Figure 13-2). This was due to the consistency with which the probability of the transitions occurred (i.e., every year, each sample unit in which fire could occur had a probability of 0.12 of having that fire) as well as the large number of sampling units.

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



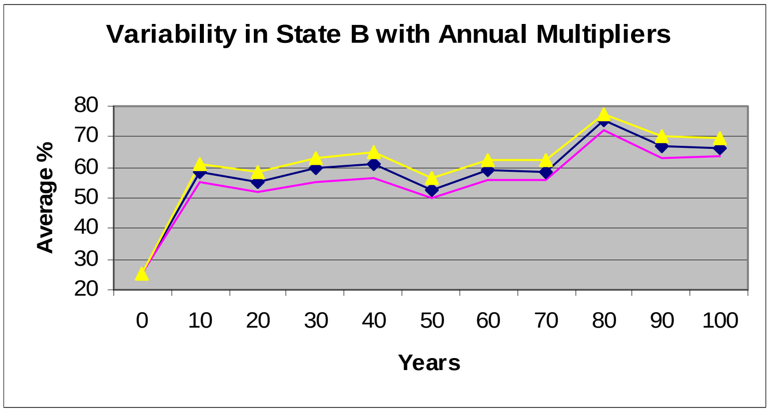


Figure 13-2. Comparison of year to year variability in state B of the simple grassland VDDT model with and without the use of annual multipliers. Maximum values in yellow, average values in blue, and minimum values in pink.

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

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

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

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

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

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

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

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

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

13.3 Introductory References:

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Chapter 14 - Semi-Desert Grassland Model

14.1 Mixed Native Vegetation Dynamics

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

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

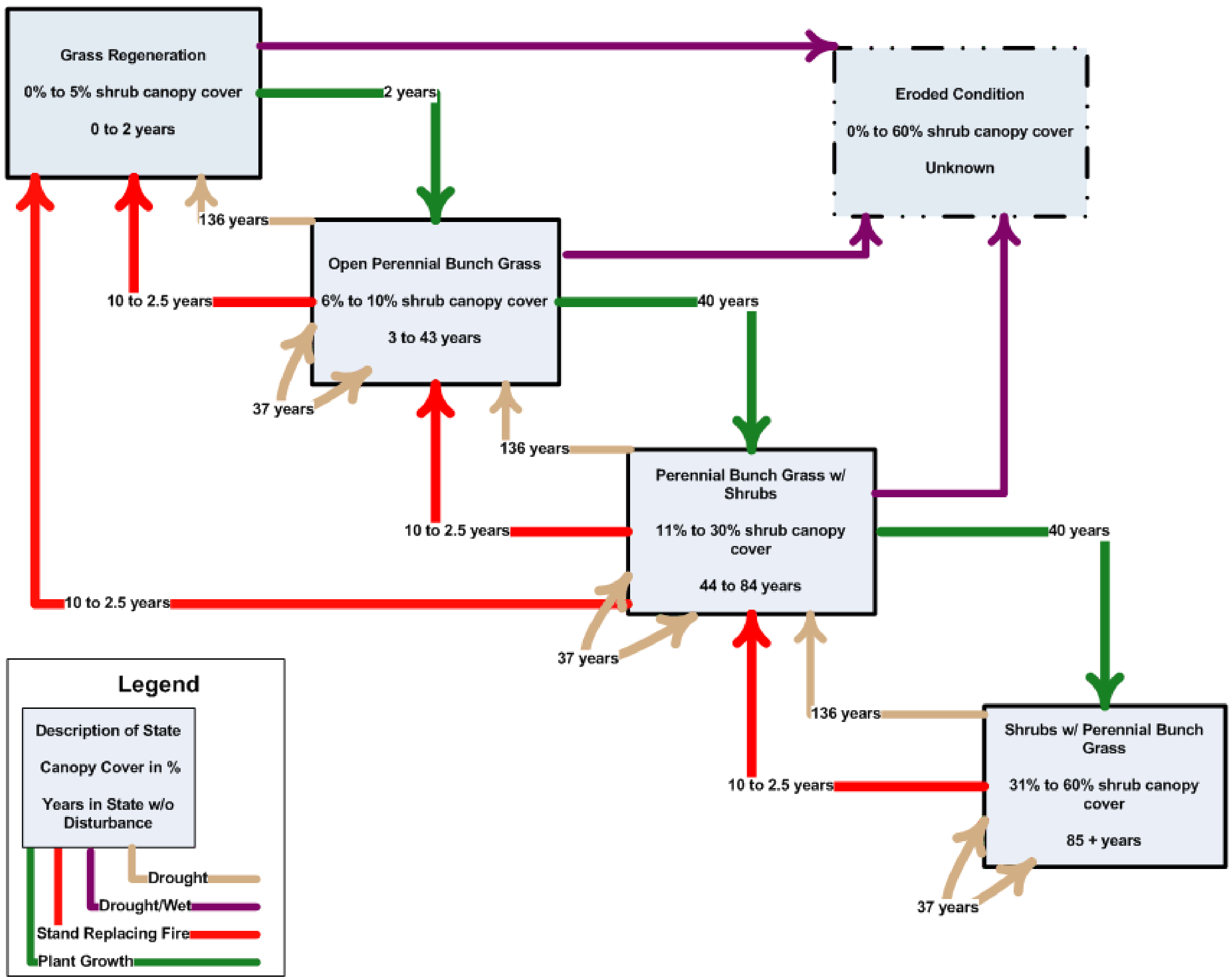

Figure 14-1. Conceptual historic state and transition model for the semi-desert grassland mixed native vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no information exists on the frequency of transitions the arrow is blank. Dashed outlines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

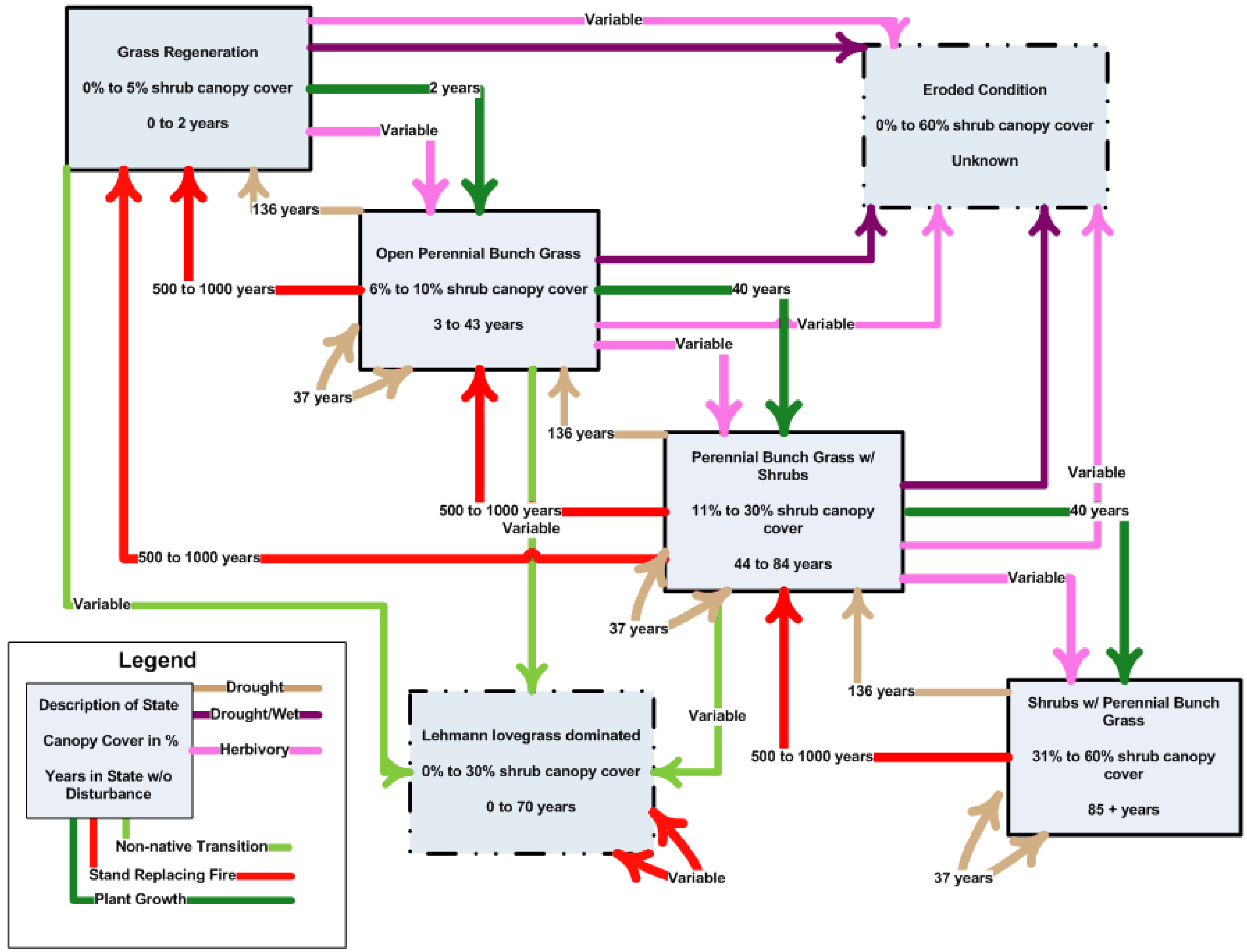

Figure 14-2. Conceptual current state and transition model for the semi-desert grassland mixed native vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, a blank arrow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

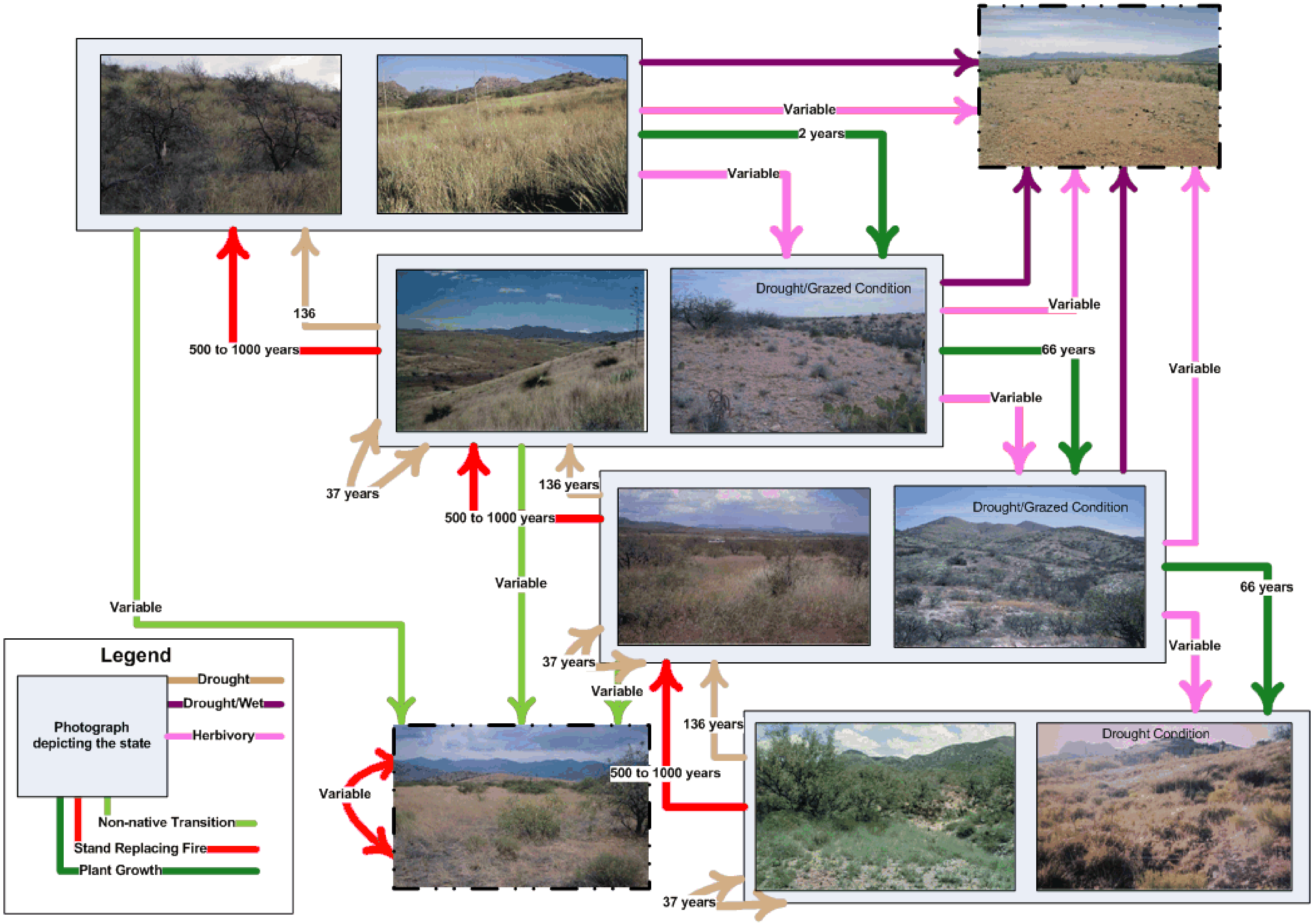

Figure 14-3. Photographic depiction of current conceptual state and transition model for the semi-desert grassland mixed native vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions,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

Model Parameters

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

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

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

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

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

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

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

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

Results

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

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

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

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

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

Discussion

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

14.2 Black Grama Vegetation Dynamics

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

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

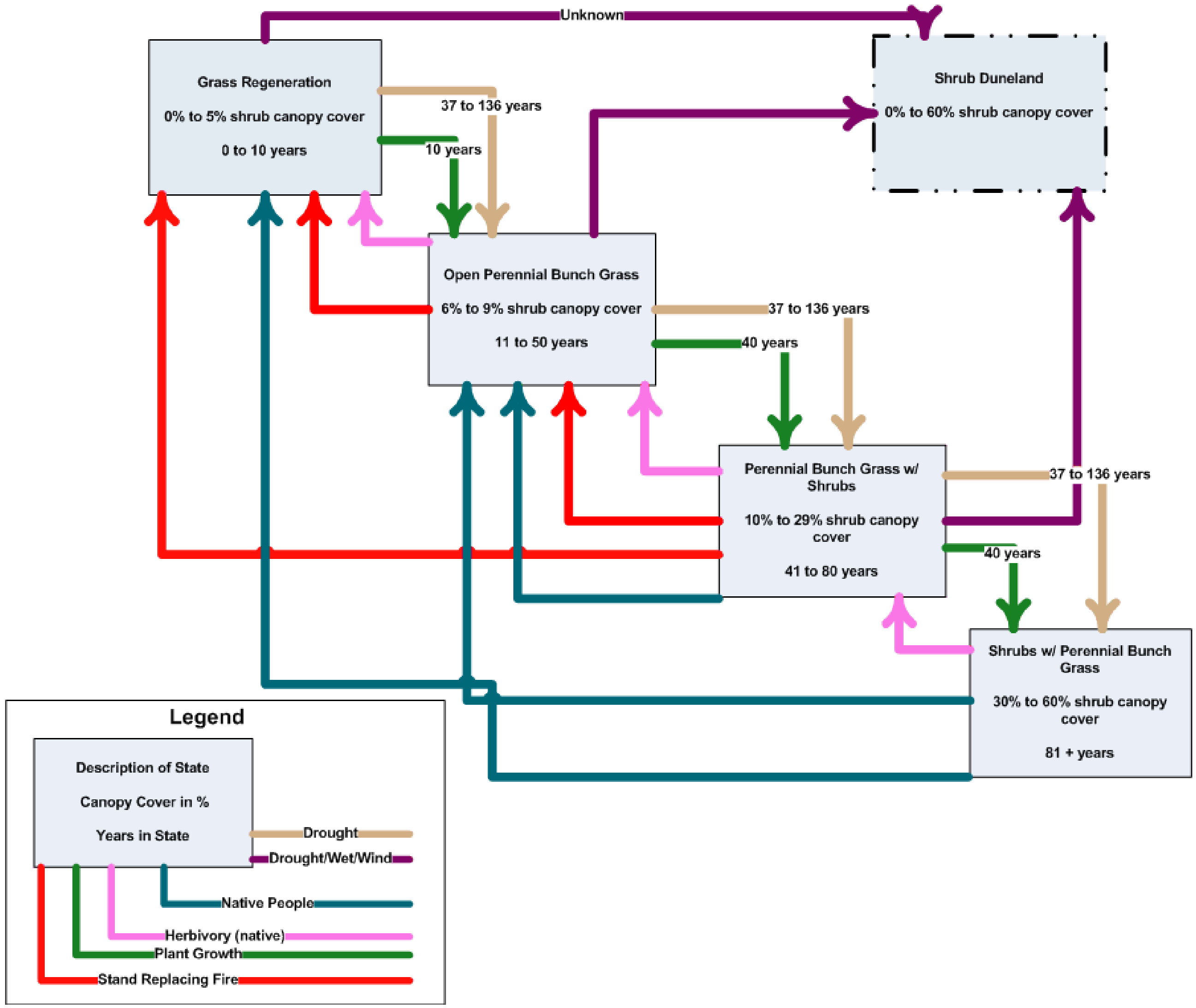

Figure 14-4. Conceptual historic state and transition model for the semi-desert grassland, black grama type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, a blank arrow or variable,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

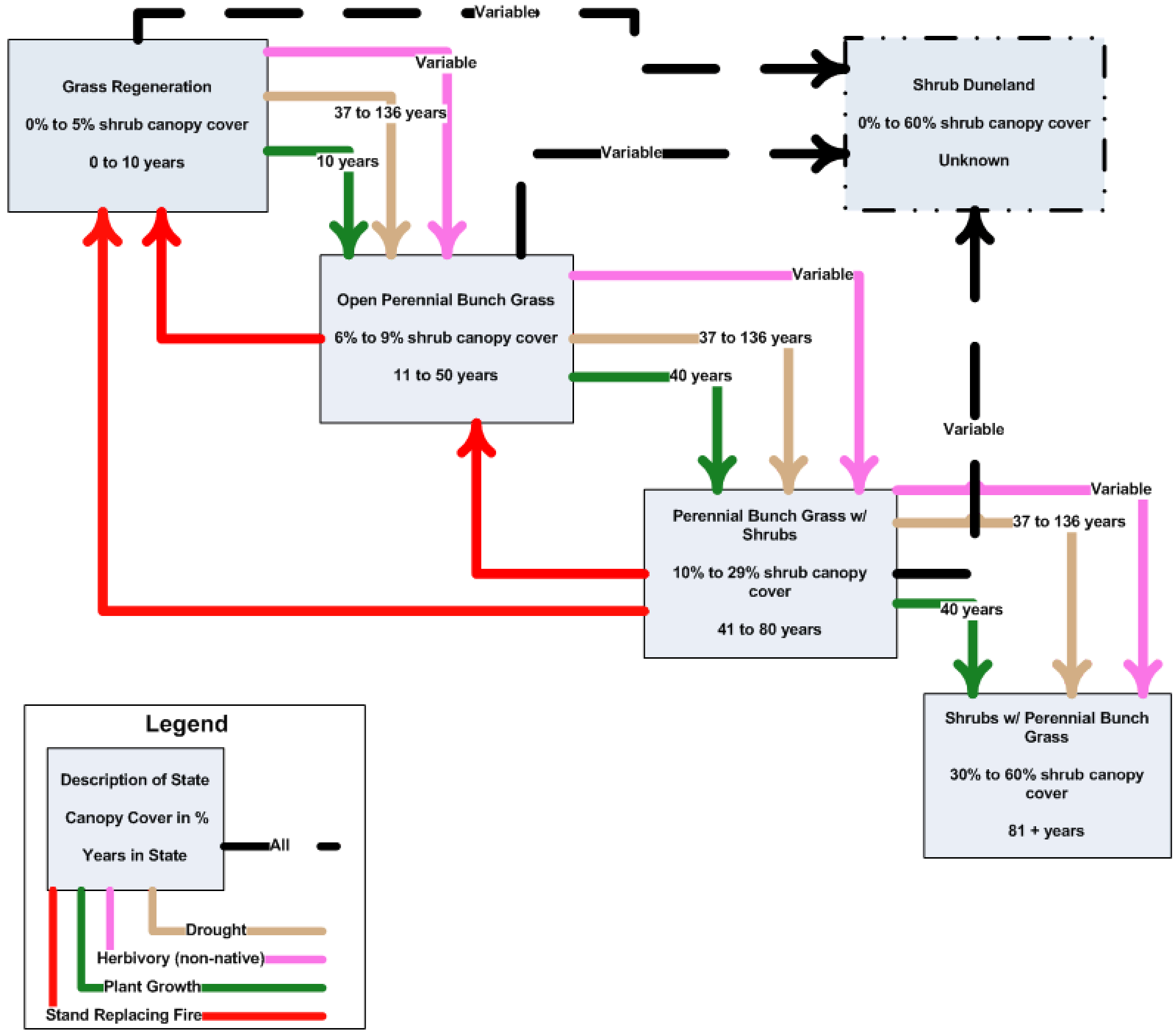

Figure 14-5. Conceptual current state and transition model for the semi-desert grassland, black grama vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, a blank arrow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

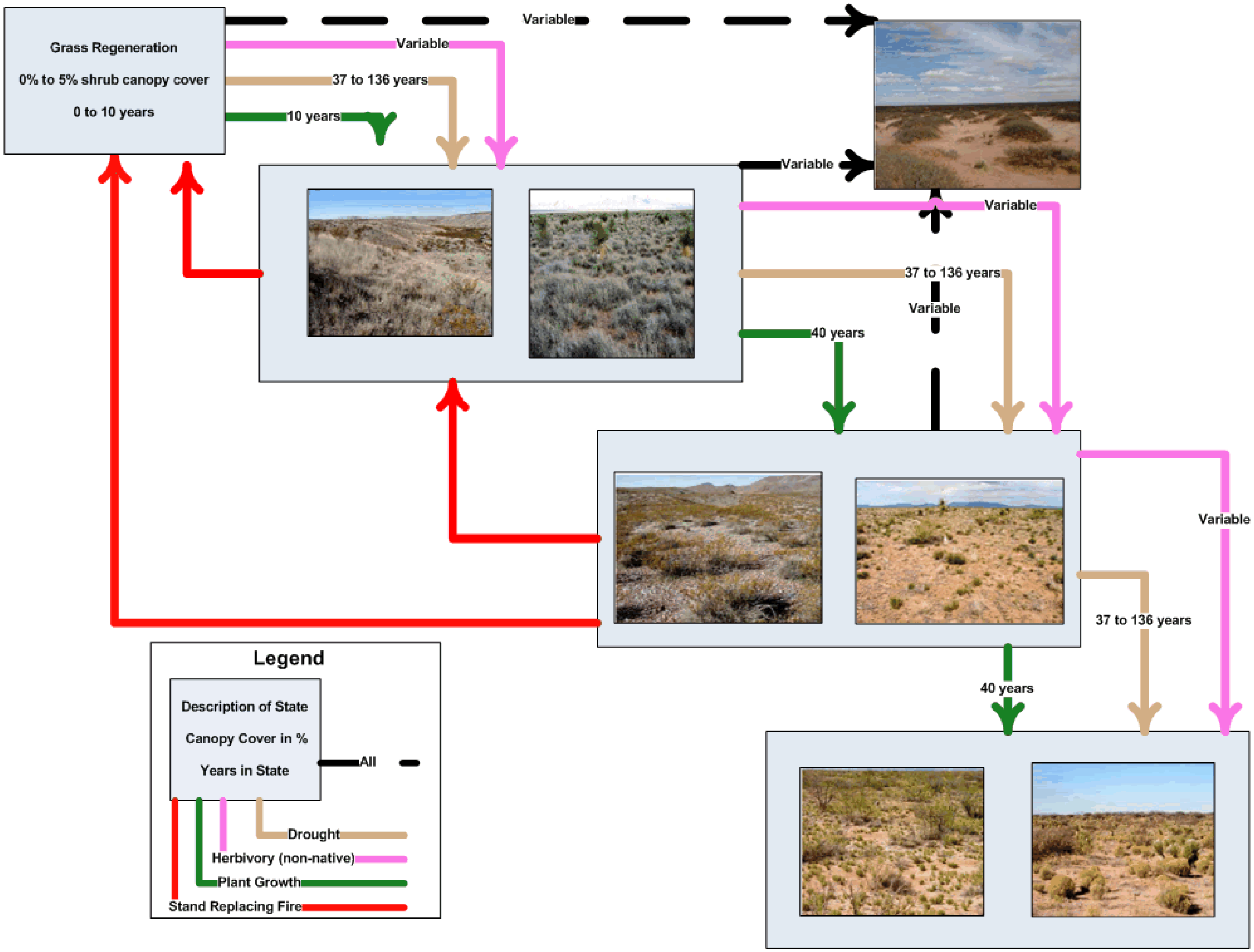

Figure 14-6. Photographic depiction of current conceptual state and transition model for the semi-desert grassland black grama vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions,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

Model Parameters

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

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

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

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

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

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

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

Valley Bottom Grassland Vegetation Dynamics

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

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