Historical Range of Variation for Potential Natural Vegetation Types of the Southwest

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

1.1 Introduction

Definition of HRV-

The Historical Range of Variation or Variability (HRV) is a description of the change over time and space in the ecological condition of potential natural vegetation types and the ecological processes that shape those types. In this document we base our 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 purposes of this document, HRV includes descriptions, when scientific data are available, of diverse characteristics of Potential Natural Vegetation Types (PNVTs – Table 1) of Arizona and New Mexico. Potential Natural Vegetation Types are coarse scale groupings of ecosystem types that share similar geography, vegetation, and historic ecosystem disturbance processes such as fire, flood, climate, native herbivory, and others.

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 and 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

The Historical Range of Variation description includes vegetation (or species) composition, age and size-class structure, and other attributes of patch or stand structure including patch size and spatial distribution, vegetation density, and canopy closure and/or cover. HRV also includes descriptions of the natural processes that have shaped PNVTs through time, including succession or patch dynamics, climatic fluctuations, wildfire regimes, and hydrological processes such as flooding and water table fluctuations. Other important processes that contribute to HRV include nutrient cycling,

predator-prey interactions, herbivory, insects and pathogens, wind-throw, avalanche, and erosion.

The rate of change of these characteristics is equally important to understanding HRV as the magnitude of change of these characteristics, thus it is important to articulate the time period for which the HRV is described, as well as the influence of humans on changes in ecosystem 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 inherent 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-Historical Range of Variation descriptions are important because they describe the evolutionary context that shaped the ecological systems we see today, and identify the processes and their variability that serve as major determinants of ecosystem function (Morgan and others 1994). The HRV can provide information about how an ecological system is capable of functioning, and identify the processes and patterns that constitute a system that has had little or no anthropogenic disruption of those processes and patterns. The HRV provides a range of *reference conditions* against which ecosystem change, whether anthropogenic or stochastic, can be measured (White and Walker 1997). Similarly, HRV helps with goal setting for future management plans by describing those reference conditions which managers can employ in descriptions of what they desire to achieve through management. Historical Range of Variation descriptions also allow land managers to

focus their attention on, and increase their understanding of areas of management that are likely to have the greatest effect on restoring patterns and processes identified in the HRV. The emphasis is not on recreating one static snapshot of conditions and processes on the landscape at one point in time, but rather on understanding the range of variation for those patterns and processes that have been studied. The value of an HRV is also in understanding the landscape-scale effects of succession and disturbance on vegetation characteristics over time (Landres 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 ecological systems. Complete understanding of complex ecological systems 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 justified, assumptions well 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:

- Appreciate 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 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	1,216,300	33,900	2,700	43,500	225,900	13,800	191,000	1,000	52,000	**1,780,100**

Potential Natural Vegetation Type	US Forest Service	Bureau of Land Management	Department of Defense	National Park Service	Private	State Trust	Tribal	US Fish and Wildlife Service	Other	Total
Forest										
Montane Grassland	17,200	0	0	0	16,900	0	2,300	0	0	**36,400**
Montane Willow Riparian Forest	17,300	14,400	800	600	42,800	11,500	12,100	100	4,100	**103,700**
Pinyon-Juniper Woodland	3,375,200	2,872,700	22,300	556,700	4,442,500	1,505,300	5,647,800	19,000	51,600	**18,493,100**
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**
Alpine/Tundra	1,600	0	0	0	6,100	0	0	0	0	**7,700**

*Urgency, Limitations, Assumptions, and Misuse of HRV*As time passes, fewer records of HRV are available to us to help fill in the gaps in our knowledge, as old trees, snags, stumps and logs burn or decay, older people who have witnessed change die or forget the details, and so these sources of information need to be assessed and recorded before they are gone. Also, it is important to identify data gaps to help prioritize efforts to fill those 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 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 3).

Table 1-3. Identification of the name, location, and contact information for each photographic archive searched as well as identification of the types of photographs obtained from each archive, as well as the PNVTs represented in the photograph.

Photograph Archives Researched:	PNVT photographs obtained from archive:
Jornada Experimental Range, Las Cruces, New Mexico (on-line photographs only; includes repeat photographs)	• Semi-desert grasslands
Santa Rita Experimental Range, southeastern Arizona (on-line photographs only; includes repeat photographs)	• Semi-desert grasslands
Ecological Restoration Institute, NAU (no repeat photographs) Contact: Dennis Lund *Note: This included 18 CDs of primarily historical photographs of ponderosa pine forests. These photographs were collected by Dennis Lund from a variety of sources including RMRS archives, Region 3 archives in Albuquerque, Kaibab NF archives, and the National archives in Washington D.C. There are about a dozen photographs that are of present day.*	• Ponderosa pine • Aspen • Mixed conifer • Pinyon-juniper

United States Geological Survey, Tucson, AZ at University of Arizona (includes repeat photographs) Contact: Diane Boyer or Ray Turner	• Semi-desert grasslands • Madrean encinal • Riparian
Rocky Mountain Research Station, Flagstaff, Arizona, US Forest Service (no repeat photographs) Contact: Susan Olberding *Note: includes mostly photographs from the Ft. Valley Research Station archive, but also from the RMRS on-line photographs*	• Ponderosa pine • Interior chaparral (on-line resource only) • Riparian
Coronado National Forest, Tucson, Arizona (no repeat photographs) Contacts: Bill Gillespie, Geoff Soroka *Note: Two sources of photographs were obtained. One source was from the archaeologist, Bill Gillespie, which included only historical photographs. The second source was from Geoff Soroka, where most of the photographs were taken in part to ground-truth the mid-scale vegetation mapping effort.*	• Aspen • Interior chaparral • Madrean encinal • Madrean pine-oak • Mixed conifer • Pinyon-juniper • Semi-desert grasslands
USFS Region 3 Office, Albuquerque, NM, US Forest Service (some repeat photographs) Contact: Sheila Poole	• Alpine-tundra • Aspen • Interior chaparral • Madrean encinal • Madrean pine-oak • Mixed conifer • Montane grasslands • Pinyon-juniper • Riparian • Semi-desert grasslands • Spruce-fir
Carson National Forest, Taos, New Mexico, US Forest Service (no repeat photographs) Contact: Bill Westbury or Dave Johnson (Forest archaeologists)	• Aspen • Mixed conifer • Montane grasslands • Riparian • Spruce-fir

Apache-Sitgreaves National Forest, Arizona (no repeat photographs) Contact: Bob Dyson *Note: The photographs came from the A-S historic archives, and were sent to me on a CD. The CD included about 500 photographs, although none of the photographs has information regarding dates taken or the specific locations of the photographs.*	• Aspen • Interior chaparral • Mixed conifer • Montane grasslands • Pinyon-juniper • Riparian • Spruce-fir
Santa Fe National Forest, Santa Fe, New Mexico (no repeat photographs) Contact: Mike Bremer	• Mixed conifer • Pinyon-juniper • Riparian • Spruce-fir
Sharlot Hall Museum, Prescott, Arizona (no repeat photographs) Contact: Ryan Flahive	• Aspen • Interior chaparral • Mixed conifer • Pine-oak • Pinyon-juniper • Riparian
Harley Shaw 'Wood plenty, grass good, water none', central Arizona (repeat photographs) *Note: Photographs taken from Harley's manuscript that will be published in the near future by the RMRS.*	• Semi-desert grasslands • Pinyon-juniper
E. Hollis Fuchs 'Historic increases in woody vegetation in Lincoln County, New Mexico', Lincoln county, New Mexico (repeat photographs) *Note: Photographs taken directly from Hollis' book.*	• Mixed conifer • Montane grasslands • Ponderosa pine • Pinyon-juniper • Riparian • Semi-desert grasslands
USGS, Los Alamos, New Mexico (repeat photographs) Contact: Craig Allen *Note: Photographs taken from an unpublished paper by Hogan and Allen (2000).*	• Pinyon-juniper • Mixed conifer • Ponderosa pine • Spruce-fir
Saguaro National Park, Tucson, Arizona (some repeat photographs) Contact: James Leckie	• Madrean encinal • Madrean pine-oak

Note: Photographs from several field season that investigated the effects of fire over several years	
Turner, Webb, Bowers, and Hastings. 'The Changing Mile Revisited.' southeastern Arizona (repeat photographs) Contact: Ray Turner or Diane Boyer *Note: These photographs were taken directly from this book.*	• Semi-desert grasslands • Madrean Encinal • Riparian
Gila National Forest, Silver City, New Mexico (no repeat photographs) Contact: Reese Lolly *Note: These photographs are from the TES mapping effort and also include some fire photographs*	• Mixed Conifer • Ponderosa pine • Interior chaparral

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. At this selection stage, photographs were retained if:

- They were taken before or around 1900
- They were a repeat series
- They documented conditions known to have occurred historically
- There were few photographs for this PNVT location (ie 10 or less), then all were retained

The final SWFAP database uses Extensis Portfolio 7 software and includes the selected photograph along with any information regarding the photograph (i.e. date taken, location, photographic descriptions, VDDT model state, photographer, title of photo, copyright, whether or not it is a repeat, and original archive source).

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.

1.3 Bibliography

Allen, Craig D. 2002. Lots of lightning and plenty of people: an ecological history of fire in the upland Southwest *in Fire, Native Peoples, and the Natural Landscape*. Edited by Thomas R. Vale. (Washington, DC: Island Press, 2002. xv + 315 pp.

Allen, C.D. 1989. Changes in the landscape of the Jemez Mountains, NM. *Ph.D. Dissertation, UC Berkeley* 346 pp.

Arno, S.F., E.D. Reinhardt, and J.H. Scott. 1993. Forest structure and landscape aptterns in the subalpine lodgepole type: a procedure for quantifying past and present conditions. Gen. Tech. Rep. INT-294. USDA, USFS, Intermountain Research Station, Ogden, UT 17 pp.

Anderson, R.S. 1993. A 35,000 year vegetation and climate history from Potato Lake, Mogollon Rim, Arizona. Quaternary Research 40:351-359.

Bourgeron, P.S. and ME Jensen. 1994. An overview of ecological principles for ecosystem management, p. 45-57. *In* Jensen, M.E. and P.S. Bourgeron, tech. eds. Volume II: Ecosystem management: principles and applications. Gen. Tech. Rept. PNW-GTR-318. Portland, OR: US DA, USFS, PNWRS. 376 p.

Carlson, A.W. 1969. New Mexico's Sheep Industry, 1850-1900. *New Mexico Historical Review* **44**.

Cooper, C.F. 1960. Pattern in ponderosa pine. *Ecology* **42**, 493-99.

Covington, W.W. and Moore, M.M. (1994) Poststettlement changes in natural fire regimes and forest structure: Ecological restoration of old-growth ponderosa pine forests. *Journal of Sustainable Forestry* **2**, 153-181.

Darlington, P.J. Jr. 1957. Zoogeography: The Geographical Distribution of Animals. Wiley Press, NY.

DeBuys, W. 1985. Enchantment and exploitation; the life and hard times of a New Mexico mountain range. Albuquerque: University of New Mexico Press.

Fritts, H.C. and T.W. Swetnam. 1989. Dendroecology: a tool for evaluating variations in past and present forest environments. Advances in Ecological Research 19:111-188.

Landres, Peter B., P. Morgan, and F.J. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecol. Appl. 9(4). 1179-1188.

Morgan, P., G.H.Aplet, H.B. Haufler, H.C. Humphries, M.M. Moore and W.D.Wilson. 1994. Historical Range of Variability: a useful tool for evaluating ecosystem change. J. of Sust. For. 2:87-111.

Muldavin, E., Neville, T., McGuire, C., Pearthree, P. and Biggs, T. (2002) Final report submitted in partial completion of Cooperative Agreement No. 28-C4-856.

Romme, W. H. 1982. Fire history and landscape diversity in Yellowstone National Park. Ecological Monographs 52:199-221.

Touchan, R.A., C.D.Allen, Thomas W. Swetnam. 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. *C.D. Allen, ed., Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium. USDA Forest Service General Technical Report RM-GTR-286* 33-46.

Turner, M.G. and R.H. Gardner. 1991. Quantitative methods in landscape ecology: an introduction. In M. Turner and R. Gardner (eds.). Quantitative methods in landscape ecology. Springer-Verlag, NY. pp3-14.

Weng, C., and S. T. Jackson. 1999. Late glacial and Holocene vegetation and paleoclimate of the Kaibab Plateau, Arizona. Palaeogeography, Palaeoclimatology, Palaeoecology 153:179-201.

White, A.S. 1985. Presettlement regeneration patterns in a southwestern ponderosa pine stand. Ecology 66:589-594.

White, Peter S., and Joan L. Walker. 1997. Approximating nature's variation: selecting and using reference information in restoration ecology. Rest. Ecol. 5(4). 338-49.

Wiens, J.A. 1985. Spatial scaling in ecology. Functional Ecology 3:385-397.

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 - Within the semi-desert grasslands there are 3 main disturbance/soil regimes 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 Arizona and have been shown to trend from open grasslands with low shrub canopy cover (less than 10%) 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 (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 not a factor and water infiltration occurs at sufficient depths, lack of fire is key for maintaining these low shrub grasslands (McAuliffe 1995).

Valley bottom grasslands with deep argillic horizons are isolated within both states and have not shown shrub or tree encroachment and/or conversion in the absence of fire (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).

Black grama dominated grasslands are primarily located within New Mexico and 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). As with vegetation dynamics within the mixed native type, areas converted to shrublands are difficult to move back into grassland states as scattered nutrients and high erosion rates characteristic of the former reinforce a shrubland 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 – In Progress

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. 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 no studies that documented hydrological processes, such as flooding, as important ecological determinants for the semi-desert grassland vegetation type. Information on erosion associated with water is covered in the *Erosion* section.

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 (Finch 2004). 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 in 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 and recycled "50 % or more of all photosynthetically fixed carbon" back into the soil (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 main Introduction 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 2-1).

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 2-2).

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 2-3).

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 2-4).

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*)

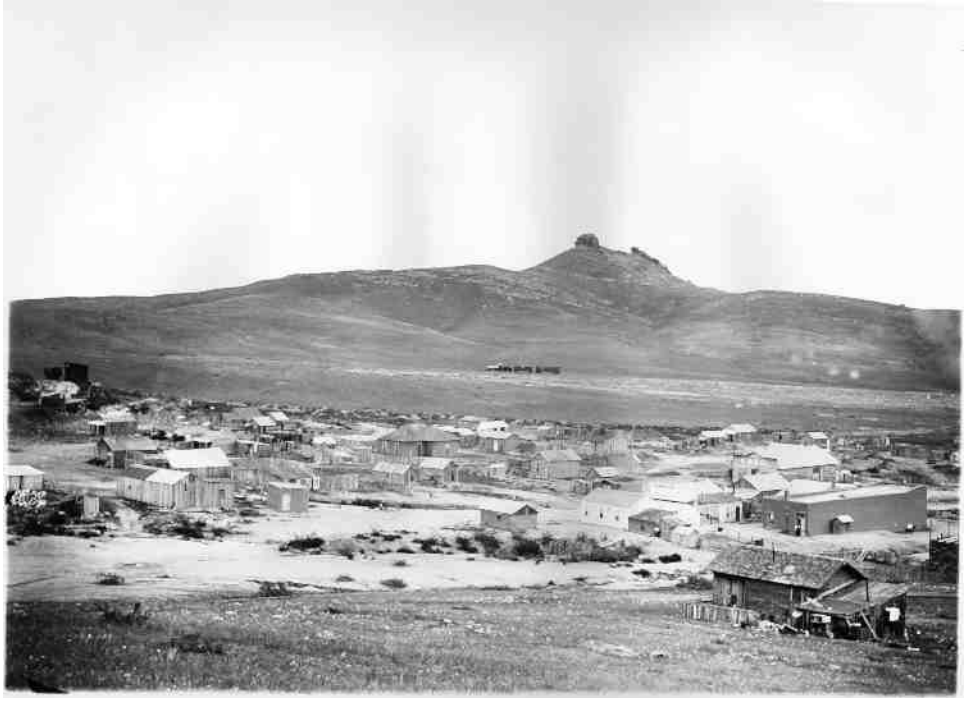

Figure 2-1. 1890's grassland photos taken near Lake Valley, New Mexico. Both photographs depict low shrub cover grasslands (Photographs courtesy Jornada Experimental Range).

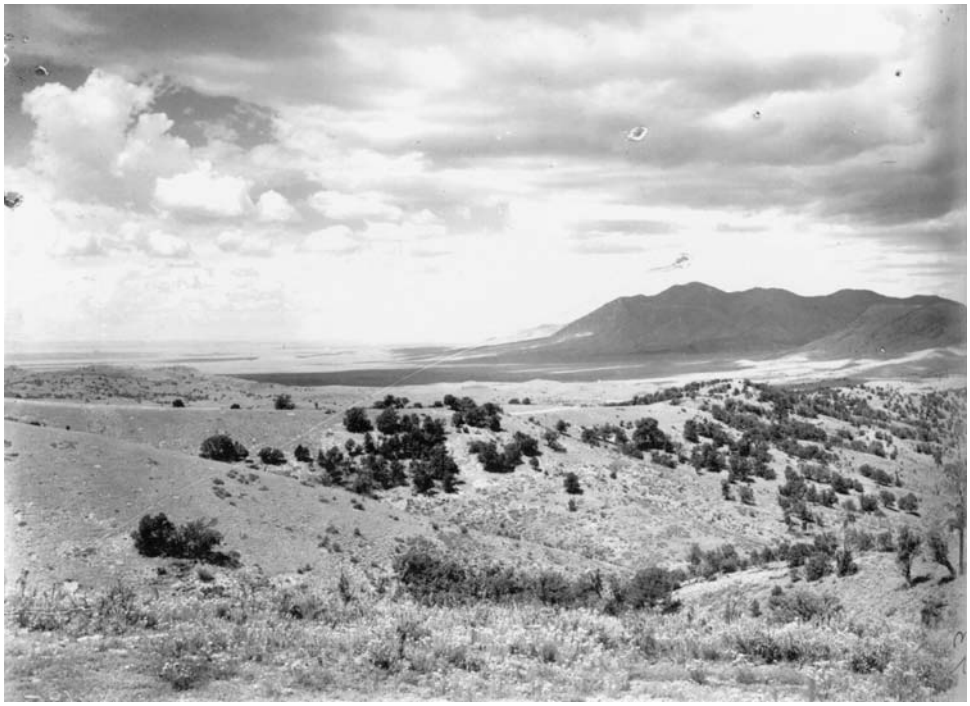

Figure 2-2. 1899 photographs of grasslands in Lincoln county showing open low shrub cover valleys with increasing shrubs and one-seed juniper on hillsides and drainages (Photographs courtesy of United States Geological Survey and Hollis Fuchs 2002)

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 within the semi-desert grassland vegetation type.

Patch or Stand Structure of Vegetation

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

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

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

Fire Management - Passive fire suppression, through livestock grazing beginning in the late 1800's as well as active suppression increasing over the last 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).

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). 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).

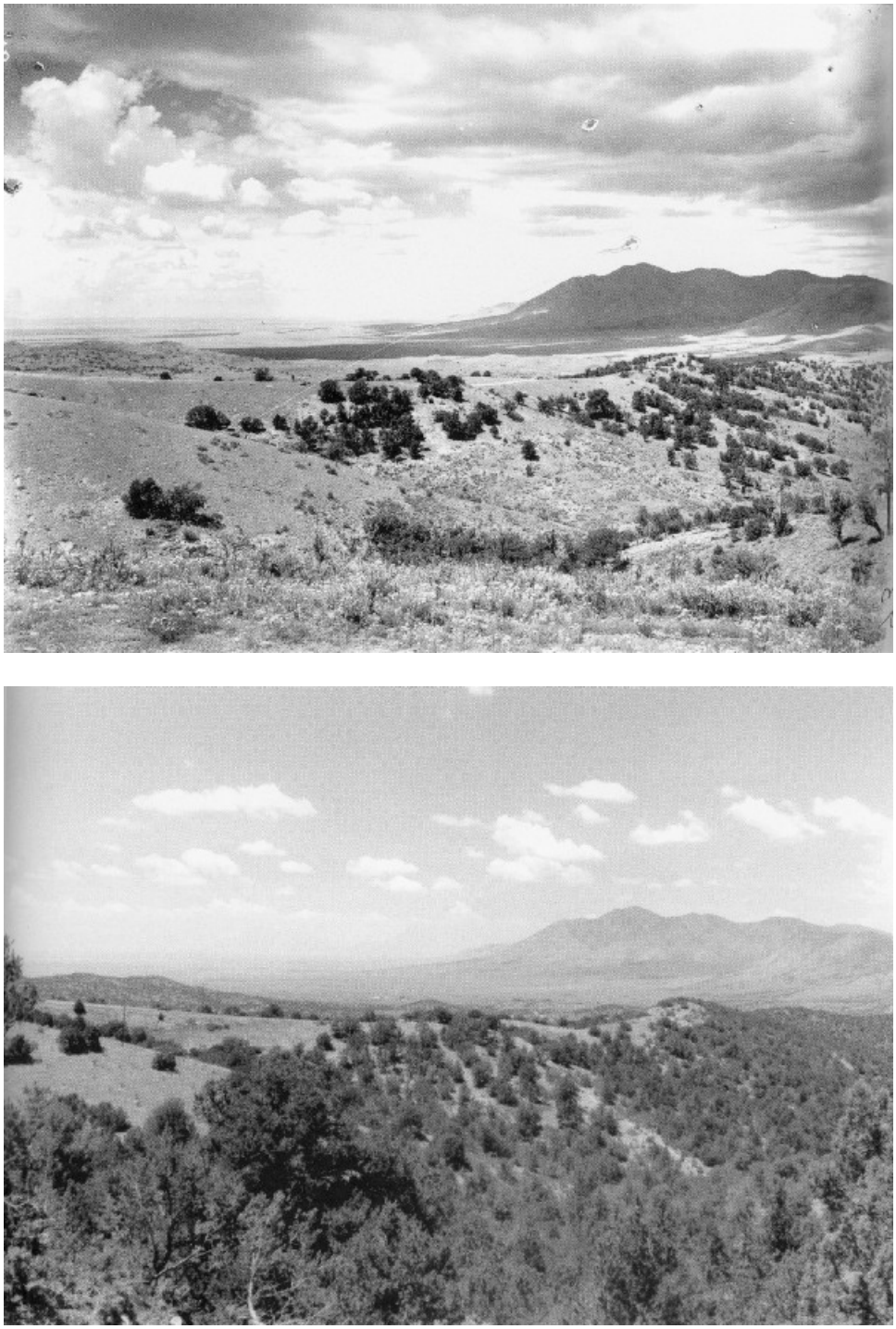

Figure 2-5. Repeat photography sequence taken in 1899 (top) and 1996 (bottom) at Fort Stanton, New Mexico. Photograph depicts expansion of juniper from the hillsides out into the open grassland valley bottom as well as increasing juniper cover on hillside (Photographs courtesy of Hollis Fuchs 2002).

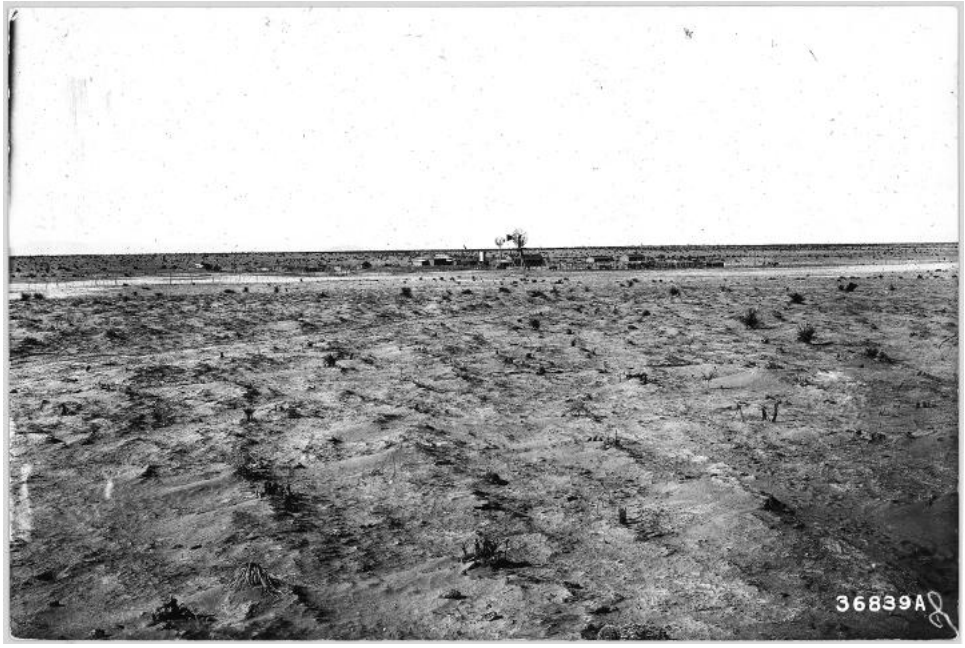

Figure 2-6. Repeat photography sequence taken in 1918 (top) and 1931 (bottom) at the Jornada Experimental Range in Las Cruces, New Mexico. Photograph depicts the transition from a open grassland to a dune shrubland. (Photographs courtesy of Jornada Experimental Range).

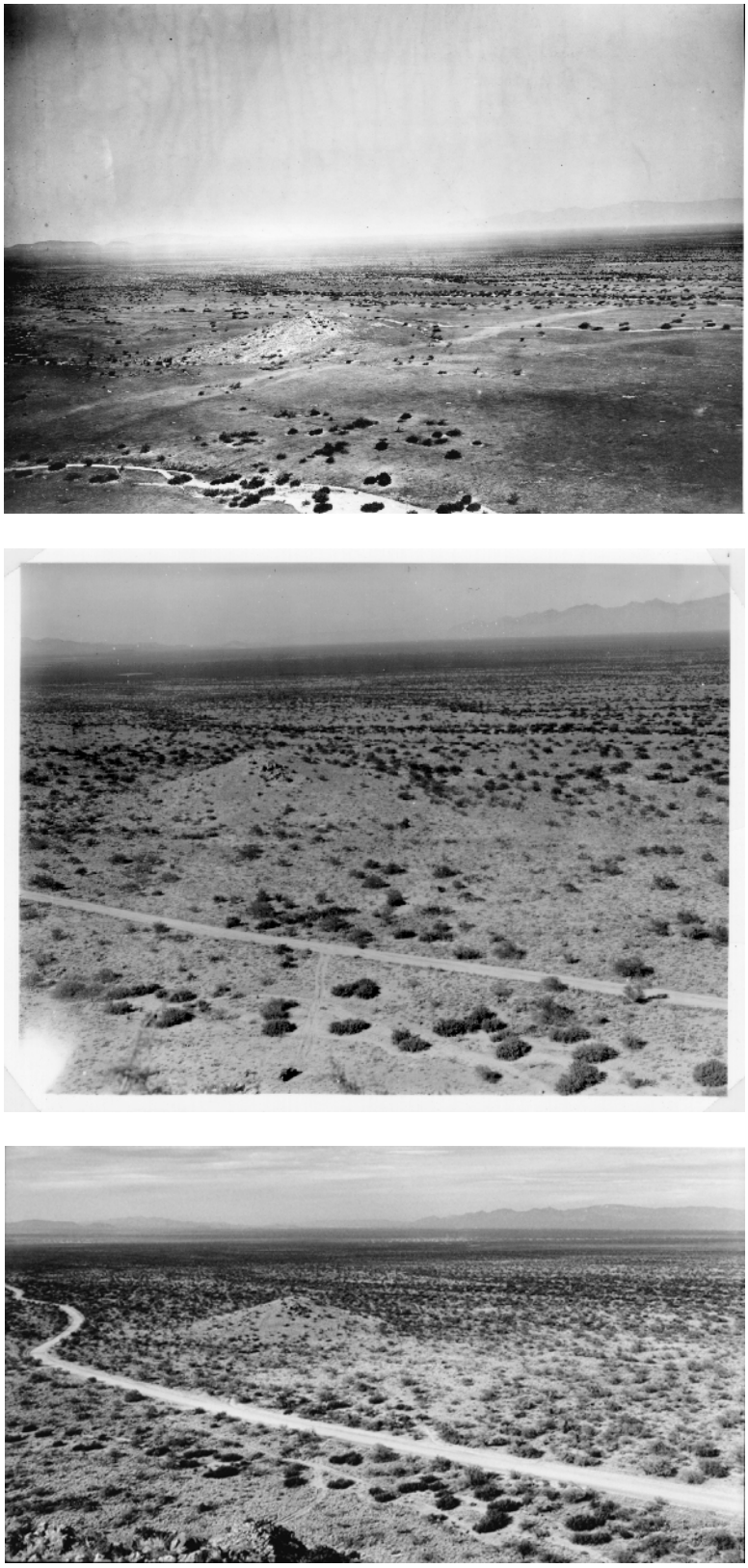

Figure 2-7. Repeat photography sequence taken in 1902 (top) and 1950 (middle) and 2000 (bottom) at the Santa Rita Experimental Range, southeastern Arizona. Photographs depict the change in cover and patch distribution of shrubs over the last 100 years on the SRER. Specifically, it is easy to see the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

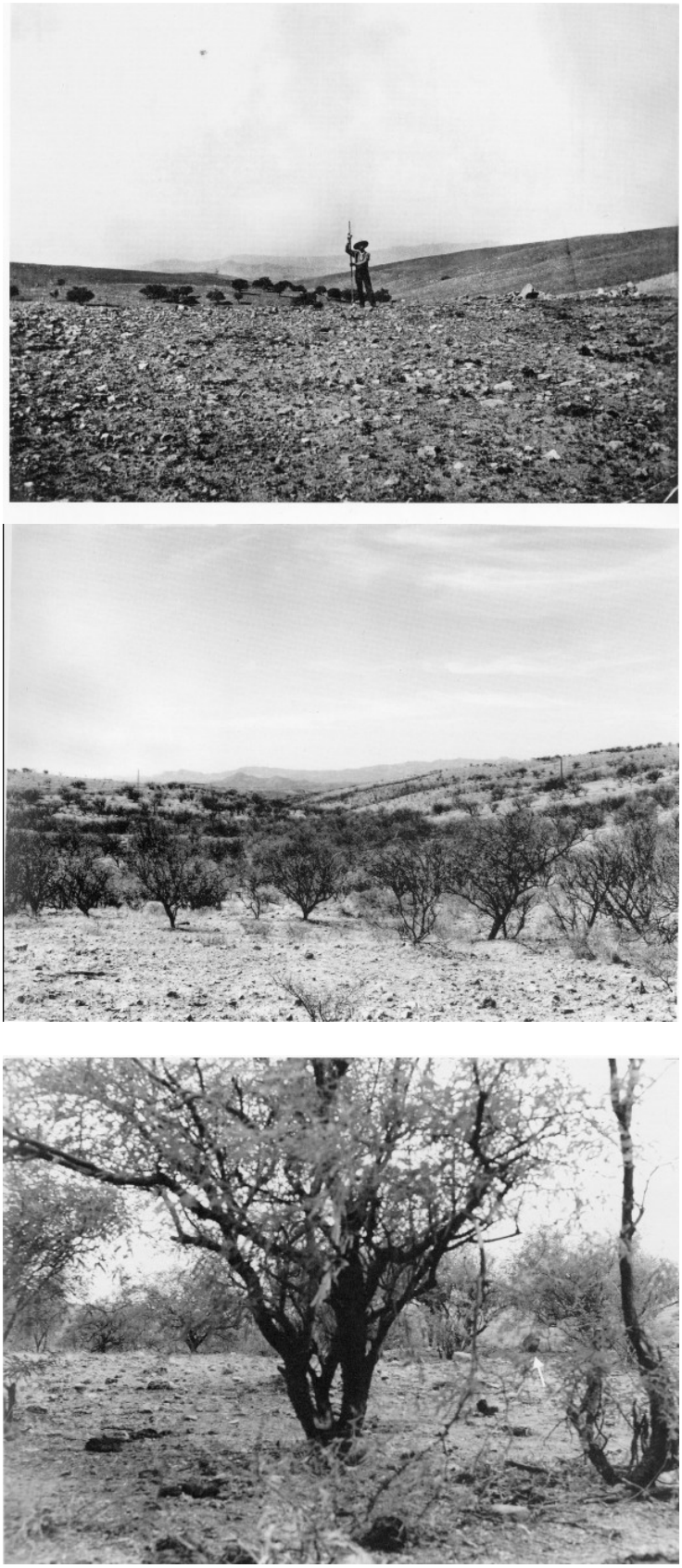

Figure 2-8. Repeat photography sequence taken in 1890 (top) and 1962 (middle) and 1996 (bottom) in Guevavi Canyon, Arizona. Photograph depicts the transition from an open grassland to a mesquite woodland. (Photographs courtesy of Unites States Geological Survey and Turner and others 2003).

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 of semi-desert grasslands.

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure - See **Overstory** section

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

Life Form - See **Overstory** section

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

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

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 2-5).

2.6 Semi-desert Grassland References

Anable, M.E. (1990) Alien plant invasion in relation to site characteristics and disturbance: *Eragrostis lehmanniana* on the Santa Rita Experimental Range, Arizona 1937-1989. Univeristy of Arizona.

Anable, M.E., McClaran, M.P. and Ruyle, G.B. (1992) Spread of introduced Lehmann lovegrass *Eragrostis lehmanniana* Nees. in southern Arizona, USA. *Biological Conservation* **61**, 181-188.

Bahre, C.J. (1985) Wildfire in southeastern Arizona between 1859 and 1890. *Desert Plants* **7**, 190-194.

Barton, A.M., Swetnam, T.W. and Baisan, C.H. (2001) Arizona pine (*Pinus arizonica*) stand dynamics: local and regional factors in a fire-prone madrean gallery forest of southeast Arizona, USA. *Landscape Ecology* **16**, 351-369.

Berger, J. (2004) The last mile: How to sustain long distance migration in mammals. *Conservation Biology* **18**, 320-331.

Bestelmeyer, B.T. and Wiens, J.A. (2002) Scavenging ant foraging behavior and variation in the scale of nutrient redistribution among semi-arid grasslands. *Journal of Arid Environments* **53**, 373-386.

Bock, C.E., Bock, J.H., Jepson, K.L. and Ortega, J.C. (1986) Ecological effects of planting African lovegrasses in Arizona. *National Geographic Research* **2**, 456-463.

Bock, C.E. and Bock, J.H. (1991) Response of grasshoppers (Orthoptera acrididae) to wildfire in a southeastern Arizona grassland. *American Midland Naturalist* **125**, 162-167.

Bock, C.E. and Bock, J.H. (1997) Shrub densities in relation to fire, livestock grazing, and precipitation in an arizona desert grassland. *The Southwestern Naturalist* **42**, 188-193.

Bock, C.E., Bock, J.H. and Grant, M.C. (1992) Effects of bird predation on grasshopper densities in an Arizona grassland. *Ecology* **73**, 1706-1717.

Bock, J.H. and Bock, C.E. (1992) Short-term reduction in plant densities following prescribed fire in an ungrazed semidesert shrub-grassland. *The Southwestern Naturalist* **37**, 49-53.

Branscomb, B.L. (1956) Shrub invasion of a New Mexico desert grassland range. University of Arizona.

Brown, J.R. and Archer, S. (1989) Woody plant invasion of grasslands establishment of honey mesquite *Prosopis glandulosa var. glandulosa* on sites differeing in herbaceous biomass and grazing history. *Oecologia* **80**, 19-26.

Brown, J.R. and Archer, S. (1999) Shrub invasion of grassland: Recruitment is continuous and not regulated by herbaceous biomass or density. *Ecology* **80**, 2386-2396.

Buffington, L.C. and Herbel, C.H. (1965) Vegetational changes on a semidesert grassland range from 1858 to 1963. *Ecological Monographs* **35**, 139-164.

Cable, D.R. and Martin, S.C. (1975) Research Paper Rm-149, Fort Collins, Colorado: United States Department of Agriculture, Forest Service.

Cable, D.R. (1972) Fire effects in southwestern semidesert grass-shrub communities. edn. 109-127p. Tallahassee, FL: Tall Timbers Research Station. (1973)

Cable, D.R. (1971) Lehmann lovegrass on the Santa Rita Experimental Range, 1937-1968. *Journal of Range Management* **24**, 17-21.

Connin, S.L., Virginia, R.A. and Chamberlain, C.P. (1997) Carbon isotopes reveal soil organic matter dynaimcs following arid land shrub expansions. *Oecologia* **110**, 374-386.

Corkidi, L., Rowland, D.L., Johnson, N.C. and Allen, E.B. (2002) Nitrogen fertilization alters the funtioning of arbuscular mycorrhizas at two semiarid grasslands. *Plant and Soil* **240**, 299-310.

Cox, J.R. and Ruyle, G.B. (1986) Influence of climatic and edaphic factors on the distribution of *Eragrostis lehmanniana* Nees, in Arizona, USA. *Journal of the Grassland Society of South Africa* **3**, 25-29.

Crider, f.J. Circular No. 730, United States Department of Agriculture .

Davis, O.K., Minckleyz, T., Moutoux, T., Jullz, T. and Kalin, B. (2002) The transformation of Sonoran desert wetlands following the historic decrease of burning. *Journal of Arid Environments* **50**, 393-412.

Drewa, P.B. and Havstad, K.M. (2000) Effects of fire, grazing, and the presence of shrubs on Chihuahuan desert grasslands. *Journal of Arid Environments* **48**, 429-443.

Federal Government, USA. National Fire Plan. 2000. 2005.

Finch, D.M. (2004) Fort Collins, Colorado: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Fuchs, H. (2002) *Increase in Woody Vegetation in Lincoln County, New Mexico*, edn. Hollis Fuchs.

Gallardo, A. and Schlesinger, W.H. (1995) Factors determining soil microbial biomass and nutrient immobilization in desert soils. *Biogeochemistry* **28**, 55-68.

Gibbens, R.P., McNeely, R.P., Havstad K.M., Beck, R.F. and Nolen, B. (2005) Vegetation changes in the Jornada Basin from1858 to 1998. *Journal of Arid Environments* **61**, 651-668.

Gibbens, R.P. and Beck, R.F. (1988) Changes in grass basal area and forb densities over a 64 year period on grassland types of the Jornada Experimental Range. *Journal of Range Management* **41**, 186-192.

Gori, D.F. and Enquist, C.A.F. (2003) The Nature Conservancy, Arizona Chapter.

Gosz, R.J. and Gosz, J.R. (1996) Species interactions on the biome transition zone in New Mexico: response of blue grama (*Bouteloua gracilis)* and black grama (*Bouteloua eriopoda*) to fire and herbivory. *Journal of Arid Environments* **34**, 101-114.

Grissino-Mayer, H.D. and Swetnam, T.W. (1997) Multi-century history of wildfire in the ponderosa pine forests of EL Malpais National Monument. *New Mexico Bureau of Mines & Mineral Resources Bulletin* **156**, 163-171.

Herman, R.P., Provencio, K.R., Herrera-Matos, J. and Torrez, R.J. (1995) Resource islands predict the distribution of heterotrophic bacteria in Chihuahuan desert soils. *Applied and Environmental Microbiology* **61**, 1816-1821.

Humphrey, R.R. (1952) The desert grassland, past and present. Corvallis, Oregon:

Humphrey, R.R. (1949) Fire as a means of controlling velvet mesquite, burroweed, and cholla on southern Arizona ranges. *Journal of Range Management* **2**, 175-182.

Humphrey, R.R. and Everson, A.C. (1951) Effect of fire on a mixed grass-shrub range in southern Arizona. *Journal of Range Management* **4**, 264-266.

Jepson-Innes, K. and Bock, C.E. (1989) Response of grasshoppers (Orthoptera : Acrididae) to livestock grazing in southeastern Arizona: differences between seasons and subfamilies. *Oecologia* **78**, 430-431.

Kaib, M., Baisan, C., Grissino-Mayer, H.D. and Swetnam, T.W. Ffolliott, P.F., DeBano, L.F., Baker, M.B., Gottfried, G.J., Solis-Garza, G.E.C.B., Neary, D.G., Allen, L.S. and Hamre, R.H., (Eds.) (1996) Fire history of the Gallery pine-oak forests and adjacent grasslands of the Chiracahua Mountains of Arizona. RM-GTR-289, Fort Collins, Colorado: United States Department of Agriculture.

Kieft, T.L., White, C.S., Loftin, S.R., Aguilar, R., Craig, J.A. and Skaar, D.A. (1998) Temporal dynamics in soil carbon and nitrogen resources at a grassland-shrubland-ecotone. *Ecology* **79**, 671-683.

Kuvlesky, W.P., Fulbright, T.E. and Engel-Wilson, R. (2002) The impact of invasive exotic grasses on quail in the southwestern United States. In: DeMaso, S.J., Kuvlesky, W.P., Hernandez, F. and Berger, M.E., (Eds.) *Quail V: The fifth National Quail Symposium*, pp. 118-128. Austin: Texas Parks and Wildlife Department]

Martin, S.C. (1983) Responses of semidesert grasses and shrubs to fall burning. *Journal of Range Management* **36**, 604-610.

McAuliff, J.R. (1995) Landscape evolution, soil formation, and Arizona's desert

grasslands. In: McClaran, M.P. and Van Devender, T.R., (Eds.) *The desert grassland*, pp. 100-129. Tucson, Arizona: University of Arizona Press]

McClaran, M.P. (2003) A century of change on the Santa Rita Experimental Range. **RMRS-P-30**, edn. Ogden, Utah.

McPherson, G.R. (1997) *Ecology and management of North American savannas*, edn. Tucson, Arizona: The University of Arizona Press.

McPherson, G.R. (1995) The role of fire in the desert grasslands. In: McClaran, M.P. and Van Devender, T.R., (Eds.) *The Desert Grassland*, Tucson, AZ: University of Arizona Press

Medina, A.L. (1988) Diets of scaled quail in southern Arizona (USA). *Journal of Wildlife Management* **52**, 753-757.

Miller, B., Ceballos, G. and Reading, R. (1994) The prairie dog and biotic diversity. *Conservation Biology* **8**, 677-681.

Miller, R.F. and Rose, J.A. (1999) Fire history and western juniper encroachment in sagebrush steppe. *Journal of Range Management* **52**, 550-559.

Moroka, N., Beck, R.F. and Pieper, R.D. (1982) Impact of burrowing activity of the banner tail kangaroo rat on southern New Mexico desert rangelands. *Journal of Range Management* **35**, 707-710.

Muldavin, E., Neville, T., McGuire, C., Pearthree, P. and Biggs, T. (2002) Final report submitted in partial completion of Cooperative Agreement No. 28-C4-856,

Neff, D.J. (1986) Federal Aid in Wildlife Restoration Project W-78-R, Arizona: Arizona Game and Fish Department.

Ockenfels, R.A., Ticer, C.L., Alexander, A. and Wennerlund, J.A. (1996) Technical Report # 19, Phoenix, Arizona: Arizona Game and Fish Department Research Branch.

Reynolds, H.G. and Bohning, J.W. (1956) Effects of burning on a desert grass-shrub range in southern Arizona. *Ecology* **37**, 769-777.

Robinett, D. (1994) Fire effects on southeastern Arizona plains grasslands. *Rangelands* **16**, 143-148.

Ruyle, G. B. (2003) Rangeland livestock production: Developing the concept of sustainability on the Santa Rita Experimental Range. **RMRS-P-30**, edn. Ogden, Utah.

Snyder, S.R., O Christ, T. and Friese, C.F. Variability in soil chemistry and arbuscular mycorrhizal fungi in harvester ant nests: The influence of topography, grazing and region. *Geoderma* **110**, 109-130.

Swetnam, T.W. and Baisan, C.H. Allen, C.D., (Ed.) (1996) Historical fire regime patterns in the southwestern Unites States since AD 1700. RMRS-GTR-286,

Fort Collins, Colorado: U.S.Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Swetnam, T.W. and Betancourt, J.L. (1990) Fire - Southern oscillation relations in the southwestern United States. *Science* 1017-1020.

Swetnam, T.W. and Betancourt, J.L. (1998) Mesoscale disturbance and ecological response to decadal climatic variability in the American Southwest. *Journal of Climate* **11**, 3128-3147.

Turner, R.M., Webb, R.H., Bowers, J.E. and Hastings, J.R. (2003) *The changing mile revisited An ecological study of vegetation change with time in the lower mile of an arid and semiarid region*, edn. Tucson, Arizona: University of Arizona Press.

Uchytil, Ronald. Pleraphis mutica In: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (2003, July). Fire Effects Information System, [Online]. Available: http://www.fs.fed.us/database/feis/ . 88. 2003.

Valone, T.J., Brown, J.H. and Heske, E.J. (1994) Interactions between rodents and ants in the Chihuahuan desert: an update. *Ecology* **75**, 252-255.

Valone, T.J. and Kelt, D.A. (1999) Fire and grazing in a shrub-invaded arid grassland community: independent of interactive ecological effects? *Journal of Arid Environments* **42**, 15-28.

Van Poollen, H.W. and Lacey, J.R. (1979) Herbage response to grazing and stocking intensities. *Journal of Range Management* **32** , 250-253.

van Riper III, C. and Ockenfels, R. (1998). The influence of transportation corridors on the movement of pronghornantelope over a fragmented landscape in northern Arizona. In the International Conference on Wildlife Ecology and Transportation in Ft. Meyers, Florida, February 9-12, 1998.

Weltzin, J.F., Archer, S. and Heitschmidt, R.K. (1997) Small mammal regulation of vegetation structure in a temperate savanna. *Ecology* **78**, 751-763.

Whitford, W.G., Forbes, G.S., Kerley, G.I. (1995) Diversity, spatial variability, and functional roles of invertebrates in desert grassland ecosystems. In: McClaran, M.P. and Van Devender, T.R., (Eds.) *The Desert Grassland*, Tucson, AZ: University of Arizona Press

Whitford, W.G. (2002) *Ecology of Desert Systems*, edn. San Diego, CA: Academic Press.

Whitford, W.G., Rapport, D.J. and Soyza, A.G. (1999) Using resistance and resilience measurements for 'fitness' tests in ecosystem health. *Journa of Environmental Management* **57**, 21-29.

Williams, D.G. and Baruch, Z. (2000) African grass invasion in the Americas: ecosystem

consequences and the role of ecophysiology. *Biological Invasions* **2**, 123-140.

Wright, H.A. (1980) The role and use of fire in the semidesert grass-shrub type. INT-GTR-85, Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Wright, R.A. (1960) Increase of mesquite on a southern New Mexico desert grassland range. Mew Mexico State Univerity.

Chapter 3 - Mixed Conifer Forest

3.1 General Description

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

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

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

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

and alder (*Alnus spp).,* serviceberry (*Amelanchier alnifolia),* chokecherry *(Prunus virginiana),* and Rocky Mountain maple (*Acer glabrum*) in the shrubby midstory, and has an extremely rich and diverse herbaceous understory represented by *P. pratensis, Fragaria spp., E. superbus, G. richardsonii,* horsetail (*Equisetum spp).,* northern bog violet (*Viola nephrophylla), Schizanche pupurescens,* and cow parsnip (*Heracleum lanatum).* The *Picea pungens/Carex foena* Habitat Type occurs in the White Mountains, North Kaibab Plateau, and in the Mogollon Mountains of New Mexico.

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

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

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

3. The *Abies concolor-Psedotsuga menziesii* HT with sparse understory occurs in two phases, either having *Berberis repens* or *Juniperus communis* or *Pachistima myrsinites* as the evergreen shrub layer, or with deciduous shrubs such as *R. neomexicana, S. oreophilus, S. scouleriana, and Q. gambelii.* This HT is widespread, with the *B. repens* phase occurring on the North Kaibab Plateau, White Mountains, San Juan and Sangre de Cristo mountains, while the *R. neomexicana* phase is in the Sacramento, Pinaleno, Chiricahua, Mogollon, and White Mountains.
4. The *Abies concolor/Acer grandidentatum* HT has a minor component of *P. menziesii*, and occurs both on top of, and along drainages of the Mogollon Plateau, and in the Pinaleno and Santa Catalina Mountains (Whittaker and Niering 1965).
5. The final HT in this series is the *Abies concolor/Festuca arizonica* HT, which includes *Pseudotsuga menziesii* and *Pinus ponderosa*, but for which shrubs are a minor component. There are conspicuous patch openings that are inhabited by a rich understory composed of the grasses *F. arizonica, M. montana, M. virescens, P. fendleriana, K. cristata, E. elymoides,* and *S. pringlei.* There are also forbs associated with the grasses, including *L. multiflorum, Antennaria spp., L. arizonicus, Thalictrum fendleri, A. lanulosa,* and *Erigeron spp.* This HT occurs on the San Francisco Peaks, Mogollon Plateau, White Mountains, and San Juan Mountains.

The last series is the *Pseudotsuga menziesii* Series, and it is represented by the *Pseudotsuga menziesii/Pinus strobiformis/Muhlenbergia virescens HT.* This HT contains *P. ponderosa*, shrubs are minor, and the understory is dominated by *M. virescens.* This HT is found in the Chiricahua, Mogollon, Pinaleno, and Santa Catalina Mountains (Whittaker and Niering 1965), and is the hottest and driest of MC Forests (See Figure 3-1).

Figure 3-1 Early photograph of mixed conifer forest (Douglas-fir and possibly southwestern white pine) in the Sierra Ancha Range of Tonto National Forest, 1917. Note mixed age of stand, with large trees in foreground (with fire scars) and younger trees in background. Photograph courtesy of the USFS Regional Office collection #164752.

3.2 Historical Range of Variation of Ecological Processes

Vegetation Dynamics - Outside of fire histories and climate studies using tree rings, little research has been focused on MC forest stand dynamics. This is despite the fact that MC forests have been used heavily for grazing, timber harvest, recreation, and hunting since around 1700. One study in the Sacramento Mountains (Hanks 1966) articulated the approximate seral stages of succession following fire. Hanks (1966) determined that stand replacing fires occurred in 1886, 1939, 1945, 1950, and 1963. Following the fire, stands were dominated by herbaceous species for 1 to 3 years, followed by increased growth of *Q. gambeli* and *R. neomexicana* until these attained tree size, at which point conifers gradually began to dominate. Another study by Hanks and Dick-Peddie (1974) found that after a stand-replacing fire, a forb (herbaceous) stage lasted for 1 to 2 years before resprouting oaks assumed dominance. This oak stage dominated until conifers

began to colonize and overtop the oaks. Conifers that could colonize within the oak thickets included *P. ponderosa* and *P. strobiformis,* as well as *A. concolor* and *P. menziesii.* Low-intensity surface fires killed seedling conifers and even sapling and juvenile *Abies* and *Pseudotsuga*, creating open, park-like savannas with scattered groves of oak (Cooper 1961, Weaver 1968, Hanks and Dick-Peddie 1974). Hanks (1966) suggested that the oakbrush stage was never a climax in this sere, and would eventually be replaced by conifers, albeit slowly. In many other MC forests, *Populus tremuloides* is the primary initial colonizer, although several coniferous species quickly establish themselves as well (Moir and Ludwig 1979).

Disturbance Processes and Regimes-

Climate- In progress.

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

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

Grissino-Meyer and others (1995) studied fire scars of mixed conifer forest trees from two sites in the Pinaleño Mountains, and found a WMPI of 4 to 6 years for low intensity

surface fires prior to 1880. They also determined from the age structure of the residual spruce-fir forest (based on tree ring data reconstructions) that it established in 1685 after one of the most widespread and intense stand replacing fires. They also determined that pre-1880 fires were initiated in the early part of the season (May to June).

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

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

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

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

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

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

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

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

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

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

3.3 Historical Range of Variation of Vegetation Composition and Structure

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

Overstory - We found no studies that documented the historical overstory composition of mixed conifer forests.

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

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

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

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

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

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

Density - We found no studies that documented historical density of mixed conifer forests. Several historic photographs suggest (e.g., see Figure 3-2), and some authors have postulated that historic forests were lower density than contemporary forests, due to

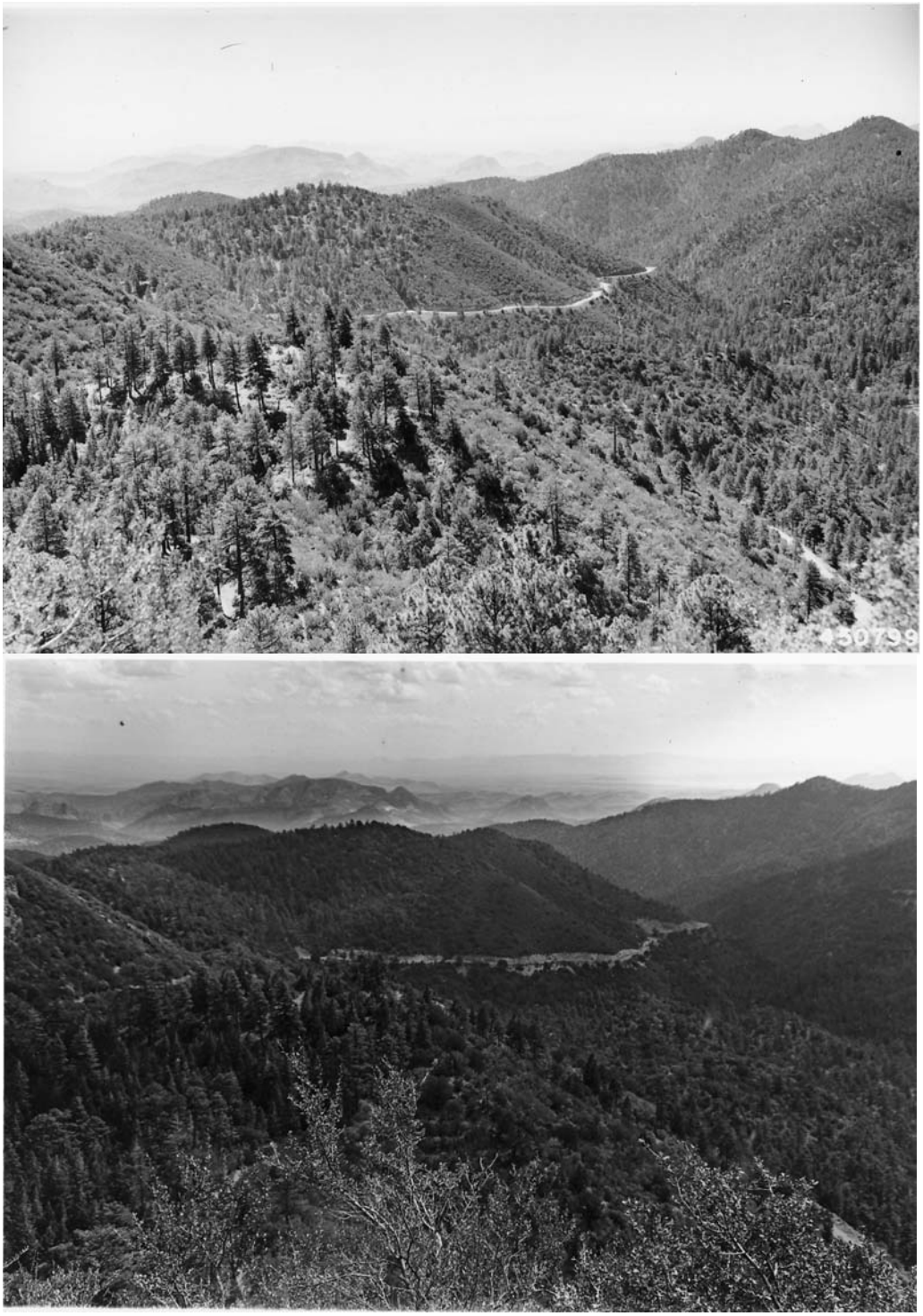

Figure 3-2. Photographic comparison from the Gila National Forest, top photo from 1948, and bottom photo from 1984. These images depict an increase in tree density, especially in the understory over 36 years. Photos courtesy USDA Forest Service Region 3 Office.[Top image15a, #450799. Looking east from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

fire suppression and in-filling by white fir and Douglas-fir (Swetnam and Baisan 1996a, Muldavin and Tonne 2003).

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

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

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

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

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

3.4 Anthropogenic Disturbance (or Disturbance Exclusion)

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

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

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

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

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

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

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

3.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory - In some mixed conifer forests on the North Kaibab Plateau and other areas, lack of fire has led to dominance of former ponderosa pine ecosystems by less fire-tolerant mixed conifer species (e.g. *P. menziesii, A. concolor*). This has led to the mistaken classification of former ponderosa pine forests as mixed conifer forests, when in fact they are ponderosa pine forests that have become dominated by mixed conifer

species (Fule *pers. comm.*, Allen and others 2002). Fire suppression has probably led to an increase in density of young white fir and Douglas-fir trees, although we could find no studies that quantified this change (Muldavin and Tonne 2003).

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

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

Patch or Stand Structure of Vegetation - Many mixed conifer forests have experienced logging activity for economical purposes, although the extent of this disturbance and its impacts on stand dynamics or other ecological effects has not been well documented (Bahre 1991). Muldavin and Tonne (2003) noted that fire suppression has led to greater homogeneity of forest stands, and a decrease in homogeneity of structure among stands through time.

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

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

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

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

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

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

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

Synthesis - Unfortunately, little has been quantified on the effects of anthropogenic disturbance on mixed conifer forests. However, a combination of anthropogenic factors, namely grazing management, fire management, and silvicultural practices have had several profound effects on general trends in mixed conifer forest form and function. The

historical mixed severity fire regime of small patches of infrequent, high severity fire within a broader matrix of low severity, frequent fires has been replaced with a more uniform and large scale, low frequency and high severity fire regime. This change in disturbance, combined with selective logging techniques that preferentially removed the larger, older Douglas-fir trees has resulted in more homogenous mixed conifer forests, with greater numbers of smaller, fire susceptible trees.

3.6 Bibliography

Allen, C.D. 1989. Changes in the landscape of the Jemez Mountains, NM. *Ph.D. Dissertation, UC Berkeley* 346.

Allen, C.D., Savage, M., Falk, D.A., Suckling, K.F., Swetnam, T.W., Schulke, T., Stacey, P.B., Morgan, P., Hoffman, M. and Klingel, J.T. 2002. Ecological restoration of southwestern ponderosa pine ecosystems: A broad perspective. *Ecological Applications* **12**, 1418-1433.

Bahre, C.J. 1991. *A legacy of change: historic human impact on vegetation of the Arizona borderlands*, edn. Tucson, AZ: The University of Arizona Press.

Bailey, L.R. 1980. If you take my sheep. 300 p.

Baisan and Swetnam 1997. Interactions of fire regime and land use history in the central Rio Grande valley. Ft Collins, CO: US Forest Service Research Paper, RM-RP-330.

Carlson, A.W. 1969. New Mexico's Sheep Industry, 1850-1900. *New Mexico Historical Review* **44**.

Cooper, C.F. 1961. Pattern in ponderosa pine. *Ecology* **42**, 493-99.

Covington, W.W. and Moore, M.M. 1994. Poststettlement changes in natural fire regimes and forest structure: Ecological restoration of old-growth ponderosa pine forests. *Journal of Sustainable Forestry* **2**, 153-181.

Covington, W.W. and Sackett, S.S. 1986. Effect of periodic burning on soil nitrogen concentrations in ponderosa pine. Soil Science Society of America Journal 50: 452-57.

Dieterich, J.H. 1983. Fire history of southwestern mixed conifer: a case study. Forest Ecology and Management 6:13-31.

Fule, P.Z., Covington, W.W., and Moore, M.M. 1997. Determining reference conditions for ecosystem management of southwestern ponderosa pine forests. Ecological Applications 7(3):895-908.

Grissino-Mayer, H.D. and Swetnam, T.W. 2000. Century scale changes in fire regimes and climate in the Southwest. *The Holocene* **10**, 207-214.

Grissino-Mayer, H.D., Baisan, C.H. and Swetnam, T.W. 1995. Fire History in the Pinaleno Mountains of southeastern Arizona: effects of human-related disturbances. *L. Debano et al. tech. coords., Biodiversity and Management of the*

Madrean Archipelago: The Sky Islands of Southwestern United States and Northwestern New Mexico, September 19-23, 1994, Tucson, Arizona, USDA Forest Service General Technical Report RM-GTR-264 399-407.

Hanks, J.P. 1966. Vegetation of the mixed conifer zone; White Mountains, NM. *M.S. Thesis, NM SU, Las Cruces* 39.

Hanks, J.P. and Dick-Peddie, W.A. 1974. Vegetation patterns of the White Mountains, N. Mex. *Southwest Nat.* **18**, 371-382.

Hermann K.R., Lavender, D.P. 1990. Pseudotsuga menziesii (Mirb.) Franco. *Silvics of North America.*

Kay, C.E. 1997. Is aspen doomed? Journal of Forestry 95 (5):4-11.

Lynch, A.M. and Swetnam, T.W. 1992. Old-growth mixed conifer and western spruce budworm in the southern Rocky Mountains. In M.R. Kaufmann, WH Moir, RL Bassett (Tech Coord), Old-growth forests in the Southwest and Rocky Mountain regions: proceedings of a workshop, Portal, AZ March 9-13, 1992; RM-GTR-213, USDAFS, June 1992: 66-80.

Moir, W.H., and Ludwig, J.A. 1979. A classification system of spruce-fir and mixed conifer habitat types of Arizona and New Mexico. Res. Pap. RM-207. Ft. Collins, CO: USDA, FS, 47 pp.

Muldavin, Esteban H.; De Velice, Robert L.; Ronco, Frank, Jr. 1996. A classification of forest habitat types: Southern Arizona and portions of the Colorado Plateau. RM-GTR-287. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 130.

Muldavin, E. and Tonne, P. 2003. A vegetation survey and preliminary ecological assessment of Valles Caldera National Preserve, New Mexico, Final report submitted in partial completion of Cooperative Agreement No. 01CRAG0014 between the University of New Mexico and the USGS, Biological Resources Division.

Pearson, G.A. 1931. Washington : U.S. Dept. of Agriculture.

Ripple, W.J., Larsen, E.J., Rankin, R.A., and Smith, D.W. 2001. Biological Conservation. Vol. 102, no. 3, pp. 227-234.

Romme, W.H., Turner, M.G., Wallace, L.L., and Walker, J.S. 1995. Aspen, elk, and fire on the northern range of Yellowstone National Park. Ecology 76, 2097-2106.

Shepperd, W.D. and Fairweather, M.L. 1994. Impact of large ungulates in restoration of aspen communities in a southwestern ponderosa pine ecosystem. In *Sustainable*

ecological approach to land management, eds. W.S. Covington and L.F. DeBano, 344-47. General Technical Report RM-247, Fort Collins, DO: USDA Forest Service.

Swetnam, T.W., Allen, C.D., and Betancourt, J.L.. 1999. Applied historical ecology: Using the past to manage for the future. *Ecological Applications* **9**, 1189-1206.

Swetnam, T.W. and Baisan, C.H. 1996a. Historical fire regime patterns in the southwestern United States since AD 1700. Pages 11-23 in Fire effects in Southwestern forests: proceedings of the second La Mesa fire symposium, C.D. Allen, (Ed.) RMRS-GTR-286, Fort Collins, Colorado: U.S.Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Swetnam, T.W. and Baisan, C.H. 1996b. Histories of montane forests in the Madrean borderlands. Pages 15-36 in Effects of fire on Madrean province ecosystems; tech. coord. P.F. Ffolliott and others. GTR RM-289. Ft. Collins, CO: USDA Forest Service.

Swetnam, T.W. and Lynch, A.M. 1989. A tree ring reconstruction of western spruce budworm history in the southern Rocky Mountains. Forest Science 35(4): 962-986.

Touchan, R.A., Allen, C.D., and Swetnam, T.W. 1996. Fire history and climatic patterns in ponderosa pine and mixed-conifer forests of the Jemez Mountains, northern New Mexico. *C.D. Allen, ed., Fire Effects in Southwestern Forests: Proceedings of the Second La Mesa Fire Symposium. USDA Forest Service General Technical Report RM-GTR-286* 33-49.

USFS 1997. Plant associations of Arizona and New Mexico, ed. 3, Volume 1:Forests. USDA Forest Service, Southwestern Region, July 1997.

Weaver, H. 1968. Fire and its relationship to ponderosa pine. *Tall Timbers Fire Ecol. Conf.Cal;if. Proc.* **7**, 258.

White, C.S. 1994. Monoterpenes: their effect on ecosystem nutrient cycling. J. Chem. Ecol. 20:1381-1406.

White, C.S. 1996. The effects of fire on nitrogen cycling processes within Bandelier National Monument, NM. In: Allen, C.D. (Ed.) Fire effects in Southwestern forests:proceedings of the second La Mesa fire symposium; March 1994, Los Alamos, NM. RM-GTR-286: 123-139.

Whittaker, R.H. and Niering, W.A. 1965. Vegetation of the Santa Catalina Mountains, AZ: a gradient analysis of the south slope. *Ecology* **46**, 429-452.