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

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

1.1 Introduction

Definition of HRV-

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

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

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

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

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

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

HRV's Application in Land Management Decision-Making – Understanding the response of PNVTs to disturbance processes (or the absence of disturbance processes) and the characteristics of PNVTs over time enables land managers to better characterize components of ecosystem diversity. In the context of land management planning, HRV enables managers to identify desired future conditions and the need for change by comparing current conditions with the range of historical conditions. HRV also describes the evolutionary context for PNVTs present today by identifying the disturbance processes (and variability) that serve as major determinants of PNVT characteristics (Morgan and others 1994). Understanding the relationship among disturbance processes, the responses of organisms to these processes, and current conditions enables managers to evaluate the potential for proposed management actions to meet ecological sustainability goals. Moreover, since the form and function of PNVTs are shaped by these processes, HRV characterizations can assist land managers in evaluating how and where appropriate disturbance regimes may be integrated into management actions. HRVs characterize a range of *reference conditions* against which ecosystem change, anthropogenic or stochastic, can be measured (White and Walker 1997) and the landscape-scale effects of succession and disturbance on vegetation characteristics over time (Landres and others 1999). Identifying reference conditions and the range of variation is important for identifying land management goals and land-use allocations. Historical Range of Variation descriptions also enable land managers to better predict where management actions are likely to have the greatest effect on restoring some of the patterns and processes identified in the HRV. However, the current biophysical conditions under which land management is practiced are different from the evolutionary environment under which ecological systems developed. For example, climate continues to change, which affects vegetation mortality, reproduction, and disturbance processes. Anthropogenic effects of landscape fragmentation through road construction, exotic species introductions, and fire suppression also contribute to what has been called the "no

analogue" condition: the current evolutionary environment may be different from the historic evolutionary environment, and some historical conditions may be neither attainable nor desirable as management goals (Swetnam and others 1999).

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

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

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

Influence of Temporal and Spatial Scale on Reported Values - The effect of scale, both spatial and temporal is well recognized for its importance in HRV descriptions (Morgan and others 1994). Reported values of ecosystem characteristics and processes are dependent upon the scale at which they are measured, and the amount of variability of measured values also varies at different scales (Wiens 1985, Turner and Gardner 1991). For example, species richness (total number of species) increases in many ecosystem types with increasing plot size (Darlington 1957), a tenet that is basic to biogeography. Similarly, the reported values of ecological processes such as fire are dependent upon the temporal and spatial scales at which they are measured, due to differences in topography and aspect (spatial) and climatic changes (temporal). However, spatial variability of topography and aspect can be viewed at multiple scales, from microsite differences operating at the smallest scale of a few feet to the landscape scale of millions of acres.

Similarly, climatic differences can operate at multiple scales from short-term drought of a few years, to decadal to century scale trends of long-term drought. Also, size of the sampling area (spatial), and length of the sampling period (temporal) both affect the reported values for ecological processes, resulting in variation in the estimated parameter due to sampling. The selection of the appropriate scales of time and space for HRVs should be based upon the analytical objectives (Bourgeron and Jensen 1993). For this project, the focus of the analysis is in understanding vegetation dynamics for a variety of PNVTs in the Southwest Region of the United States. For this reason, we have chosen to report values for the full extent of each PNVT across the two-state Region III of the United States Forest Service. The spatial scale thus falls into the range of hundreds of thousands to millions of acres, depending on the PNVT, and with the exception of Alpine/Tundra, Gallery Coniferous Riparian Forest, Montane Grassland, and Wetland/Cienega (Table 1-2). Similarly, since the time period of inquiry for establishing HRV focuses on pre- and post-settlement times for these PNVTs, and time scale should encompass multiple generations of vegetation (Morgan and other 1994), the time scale of inquiry is over hundreds of years, from approximately 1000 until the present. Ultimately, we have allowed the availability of published empirical data to be our guide in determining and reporting relevant information regarding the magnitude and variability of ecosystem characteristics and processes for these HRVs.

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

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

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

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

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

1.2 Methods Used in Determining HRV

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

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

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

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

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

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

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

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

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

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

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

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

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

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

- We retained any photographs of PNVTs that we thought were good examples of various model states within a PNVT (i.e., open canopy, closed canopy, early seral, late seral)
- We attempted to get as many historical photographs (vs. current day) as possible, although we were limited by availability

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

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

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

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

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

Data - There are two publicly available climate reconstruction data sets that cover the Southwest region for the last 1000 years; a summer (June to August) Palmer Drought

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

It is important to note some key caveats regarding the data sets. The percent of variation in the cool season precipitation record explained (R2 value) by Ni and others (2002) reconstruction varies for each climate division and should be considered when evaluating results (Table 1-4) (CLIMAS 2005 http://www.ispe.arizona.edu /climas/research / paleoclimate/product.html). Similarly, the Cook and others (1999) reconstructions are based on anywhere from 5 to 9 tree chronologies with less certainty in the reconstruction occurring with fewer chronologies (

Table 1-5). Additionally, information used to build both reconstruction models comes from upper elevation pine species which should be considered when extrapolating these data to lower elevation warm season dominated vegetation types or areas. Even with the above mentioned constraints, these climate data give an unprecedented regional look at historic climate conditions throughout the Southwest.

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

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

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

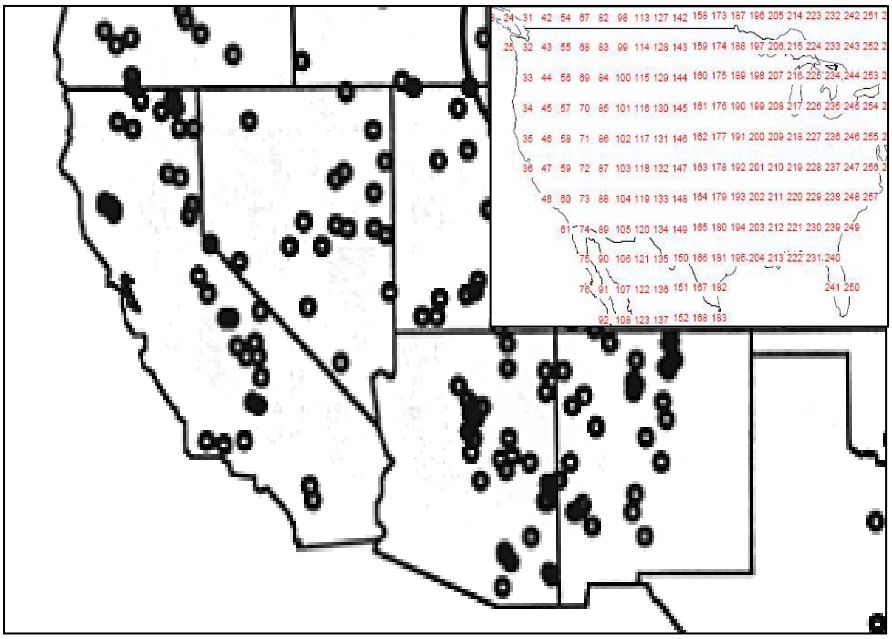
	88	**89**	**104**	**105**	**119**	**120**	**133**	**134**
# of Tree Chronologies	8-9	5-9	8-9	5-9	9	6-9	8-9	5-9

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

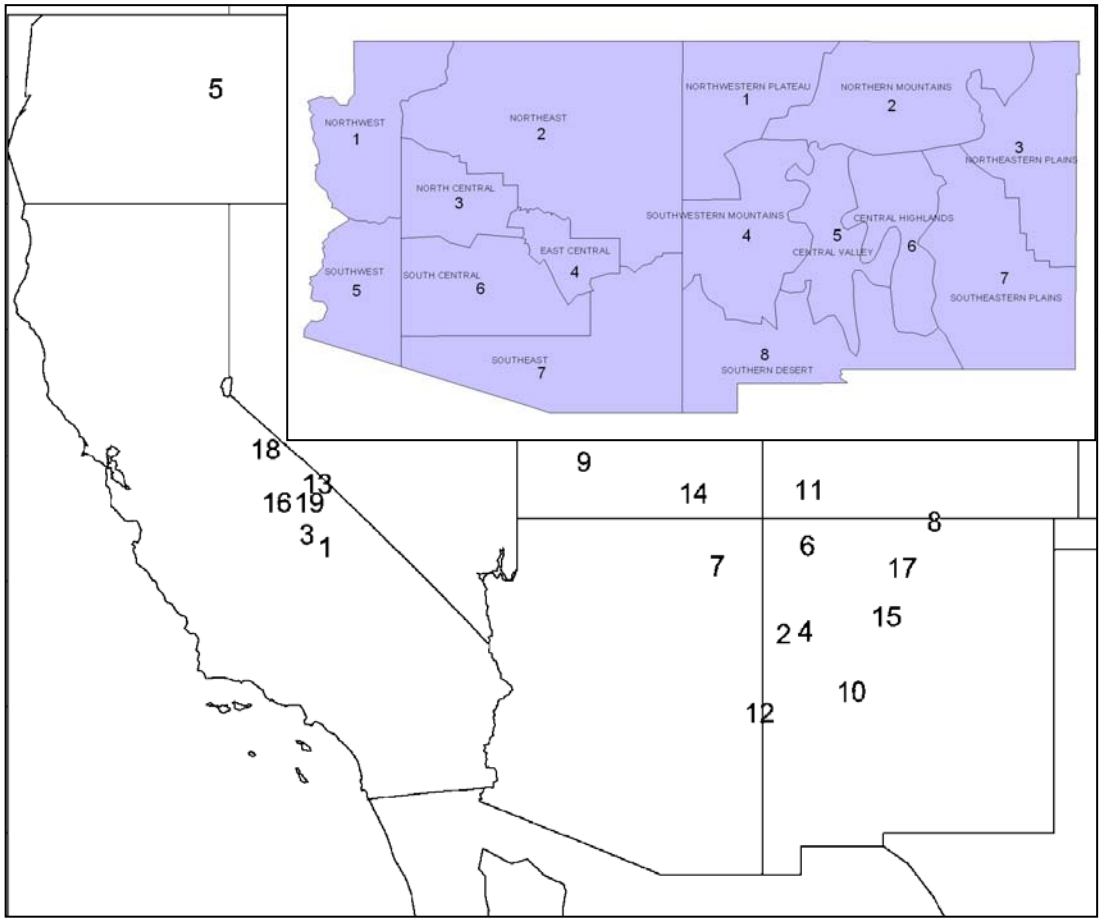
Results - A comparison of the percent of dry and wet winter precipitation years, for the 15 climate divisions that span Arizona and New Mexico, showed a pattern of 19% of the years, between year 1000 and 1988, classified as severe drought or extremely wet years, 11% classified as drought years, 8% classified as wet years, and 43% classified as normal years (Figure 1-2 and Appendix 1- Table 1.1 and Figures 1.1 to 1.15). The long-term winter precipitation averages for each climate division range from 2.4 to 9.8 inches/yr. Comparisons of the 8 summer PDSI locations showed the pattern of 11% of the years classified as severe and extreme drought, 27 % classified as moderate and mild drought, 38% classified as near normal and incipient wet and dry spells, 20% classified as slightly or moderately wet, and 5% classified as very and extremely wet years (

Table 1-5, Figure 1-3, and Appendix 1 - Table 1.2 and Figures 1.16 to 1.23). Overall there is little regional variability in the percent of dry and wet years for either the winter precipitation or summer PDSI data sets. Of the regional variability that is present, the majority of the variation occurs within the winter precipitation data set between severe drought and drought years. For example, New Mexico climate divisions 2, 3, and 6 had fewer severe drought years than the average, but had higher drought years.

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



1a.



1b.

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

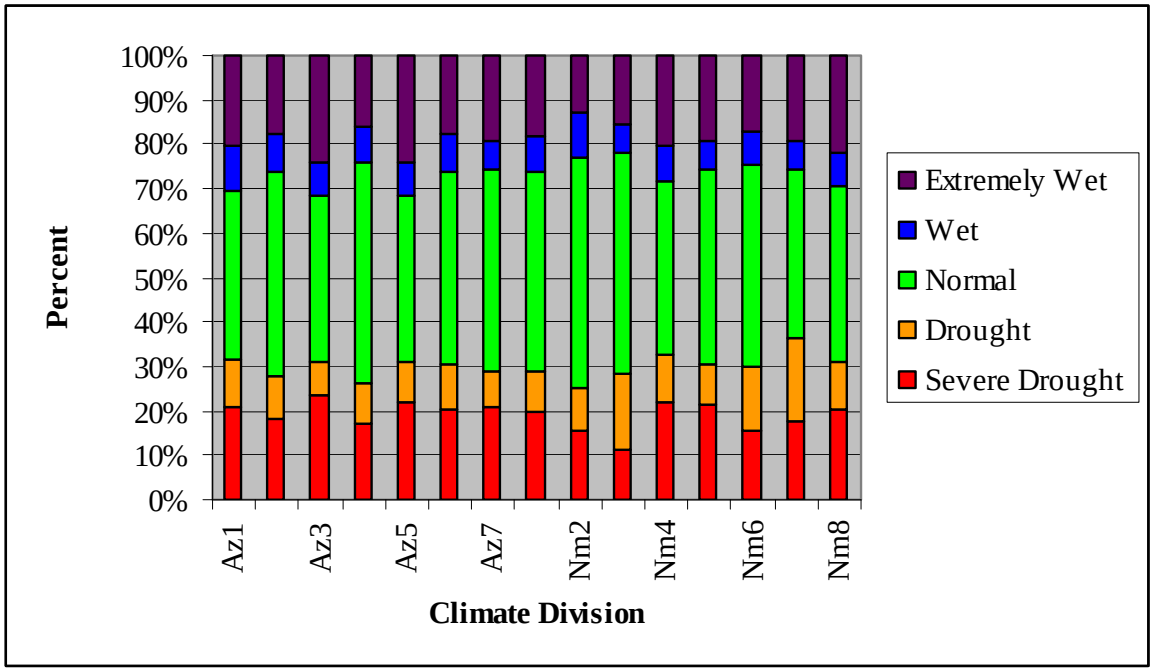


Figure 1-2. Comparison of the percent of years in all year types for all climate divisions in the Southwest.

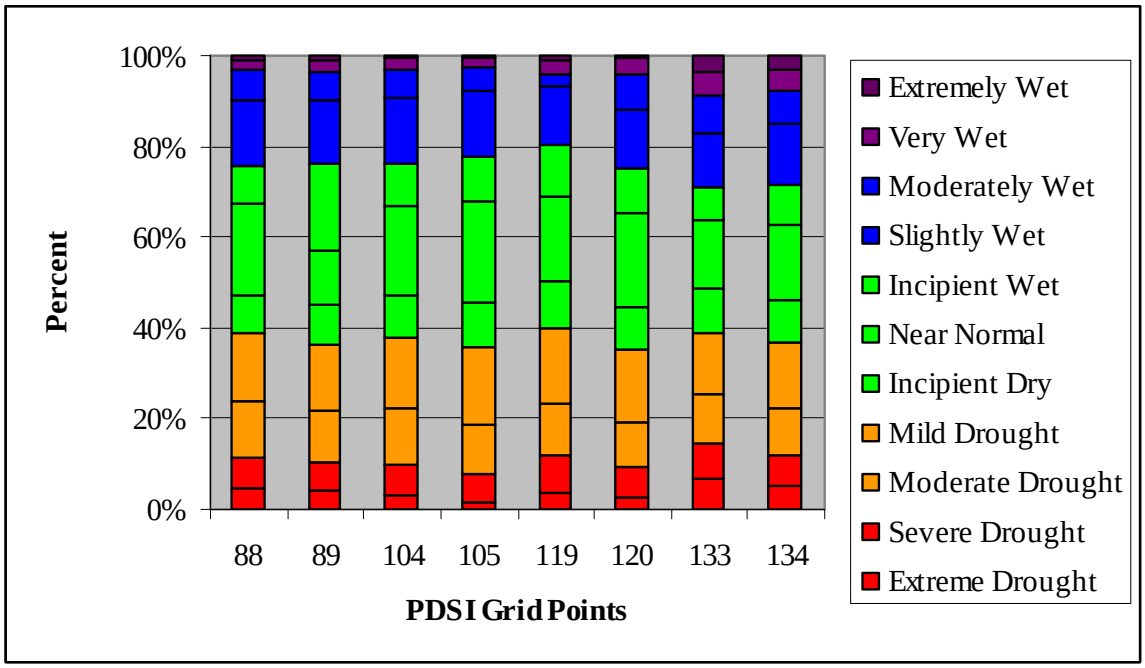


Figure 1-3. Comparison of the percent of years in all year types for all PDSI grid locations in the Southwest.

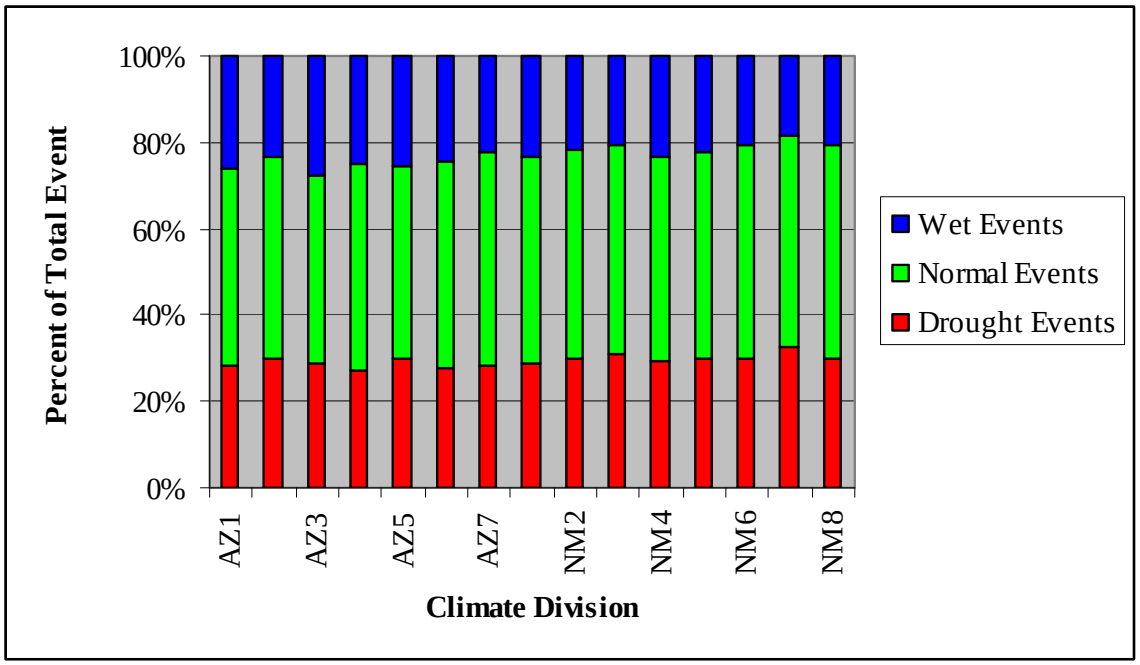


Figure 1-4. Comparison of the percent of events classified as drought, normal, and wet events for all climate divisions in the Southwest.

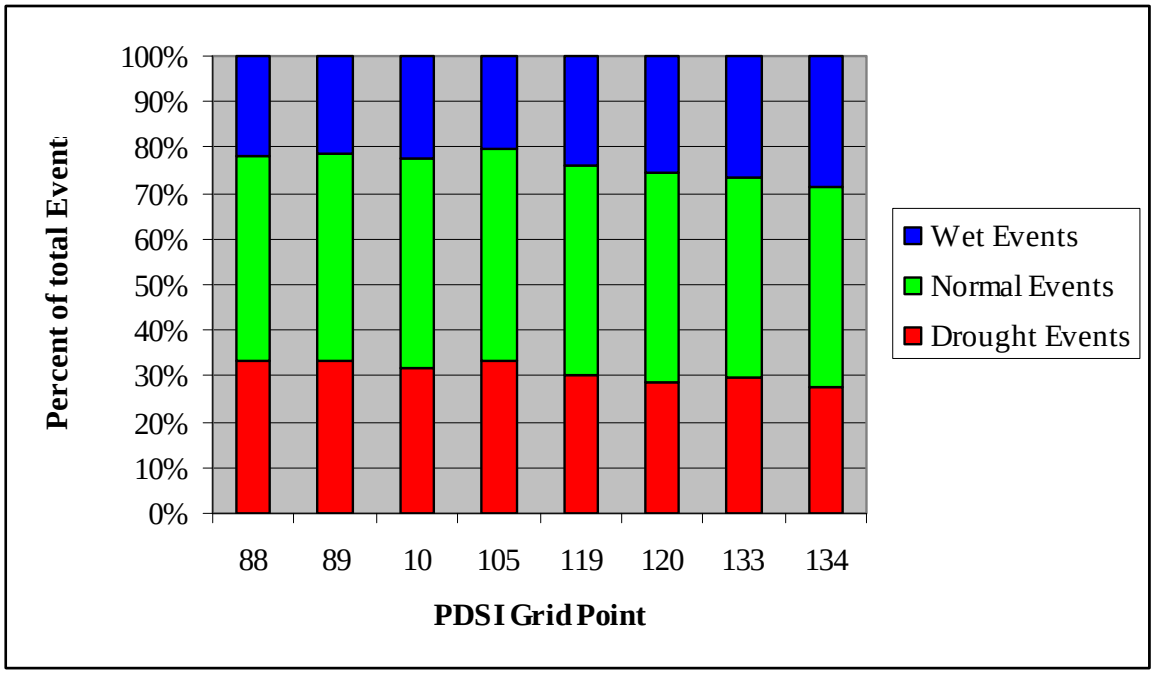


Figure 1-5. Comparison of the percent of events classified as drought, normal, and wet events for all PDSI grid locations in the Southwest.

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

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

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

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

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

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

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

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

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

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Chapter 12 - Pinyon-Juniper Woodland

12.1 General Description

Pinyon-juniper woodlands and savannas in Arizona and New Mexico generally occur between 1370 and 2290 meters (4500 and 7500 feet) in elevation and are bounded at lower elevation by grasslands or shrublands and at higher elevation by ponderosa pine and other montane forest associations (Bradley and others 1992; Daubenmire 1943; Gottfried 1987; Meeuwig and Bassett 1983; Moir and Carleton 1987; Ronco 1990; Woodbury 1947). Pinyon-juniper woodlands and savannas, i.e. those dominated by juniper and/or pinyon, (hereafter referred to in total as pinyon-juniper woodlands) occur on 7.5 million hectares (18.5 million acres) in Arizona and New Mexico of which 1.4 million hectares (3.5 million acres) are under US Forest Service management in Region 3 (USGS 2004); these woodlands also occur in nine other states across the West, occupying an additional 11.5 million hectares (28.4 million acres) outside of Arizona and New Mexico (Barnes 1983; Billings 1951; Meeuwig and Bassett 1983; Tueller and Clark 1975; West and others 1975; USGS 2004). Because the goal of this document is to summarize literature germane to the occurrence of pinyon-juniper woodland on US Forest Service Region 3 land, we restrict our review to studies conducted in Arizona, New Mexico, and portions of Utah and Colorado within the Colorado Plateau and Southern Rocky Mountains ecoregions. These ecoregions have been defined on the basis of similar soils, landforms, vegetation and climate patterns and extend into northern Arizona and New Mexico (Bailey 1995, 1998).

Climatic conditions vary considerably in pinyon-juniper woodlands within Arizona and New Mexico (West and others 1975). Mean annual precipitation ranges between 18 to 63 cm (7 to 25 inches) with northern Arizona receiving the least amount of total annual precipitation and southern Arizona and New Mexico the greatest amount of total annual precipitation (Ronco 1990; West and others 1975). More open stands (i.e. those with low pinyon-juniper cover) typically receive less than 40 cm (16 inches) of precipitation annually, while denser stands may receive more than 40 cm (O'Rourke and Ogden 1969; Springfield 1976). Most of the Southwest has a bimodal seasonal precipitation pattern, but there is a gradient of increasing summer precipitation relative to winter precipitation extending from northwestern Arizona, where winter precipitation dominates, to east-central and southeastern New Mexico, where most of the annual precipitation falls in summer and fall, and winters are cool and dry (Jurwitz and Kangieser 1978; Springfield 1976). Mean annual temperatures in these woodlands range from 40° to 61° F with cooler temperatures occurring in northern Arizona and New Mexico and warmer temperatures in the south (Ronco 1990).

Pinyon-juniper woodlands consist of relatively few tree species, but exhibit considerable diversity in understory plant composition in addition to tree composition (Aro 1971; Gottfried and others 1995; USDA 1997). In the Southwest, there are five common species of juniper and four species of pinyon which account for the pinyon-juniper woodland types (Daubenmire 1943). As the type extends westward, twoneedle pinyon (*Pinus edulis)* is replaced by singleleaf pinyon (*Pinus monophylla*) in western Arizona (Aro 1971; Daubenmire 1943) and as the type extends south, twoneedle is replaced by border pinyon (*Pinus discolor*) and Mexican pinyon *(Pinus cembroides)* in lower elevations (Dick-Peddie 1993a; Little 1971). While most juniper species

occur across the range of pinyon-juniper woodlands in the Southwest, species dominance among junipers appears to be related to elevation and precipitation patterns. Oneseed juniper (*Juniperus monosperma*) dominates at lower elevations on xeric sites in east-central Arizona and most of New Mexico (Ronco 1990). In southern Arizona and southwestern New Mexico, where annual precipitation comes primarily during the summer monsoon, alligator juniper (*Juniperus deppeana*) dominates in mesic sites often at higher elevations (Dick-Peddie 1993a; Ronco 1990; Springfield 1976). Where winter moisture is greater than summer moisture, Utah juniper (*Juniperus osteosperma*) dominates in dry, cold sites (Daubenmire 1943; Springfield 1976). Redberry juniper (*J. coahuilensis*) is common at lower elevations in central and southern Arizona (Aro 1971; Fowells 1965), and Rocky Mountain juniper (*Juniperus scopulorum*) occurs, usually as a codominant, in more mesic settings (Johnston 1987; Noble 1990). Usually one species of pinyon and up to three juniper species can be found in a stand, although pure stands of juniper are common, and pure stands of pinyon are sometimes found (Dick-Peddie 1993a).

The geographical distribution of pinyon and juniper species and their differential response to moisture, temperature and elevational gradients result in changing species' dominance patterns within woodlands (Martens and others 2001). In general, juniper is more abundant than pinyon in lower elevation woodlands and pinyon predominates at higher elevations, due to differential germination and mortality of the species along the elevational gradient (Martens and others 2001; Merkle 1952; Naylor 1964; Springfield 1976; Woodin and Lindsey 1954). One exception to this pattern is the prevalence of juniper in upper elevations in southern New Mexico, which may be related to the abundance of summer moisture (Ernest and others 1993; Hill 1990; Kennedy 1983; Pieper and others 1971; Potter 1957). Also, pinyon is generally more abundant than juniper in woodlands in northern Arizona and northern New Mexico (Hill 1990; Howell 1941; Merkle 1952; Naylor 1964; Perez 1978; Pieper and Lymbery 1987; Pieper and others 1971; Rasmussen 1941; Springfield 1976; Tueller and others 1979; Woodin and Lindsey 1954). Also, in recent years (1996 to 2003), high pinyon mortality resulting from regional drought and insect infestation have shifted many northern Arizona and New Mexico pinyon-juniper woodlands to juniper dominance (Breshears and others 2005; Mueller and others 2005a; Shaw and others 2005).

Soils supporting pinyon-juniper woodland vary in texture (stony and gravelly sandy loams to compacted clay), in depth (shallow to deep), in parent material, and in moisture availability (well drained fractures to poorly drained shallow soils with hardpan) (Springfield 1976; Woodbury 1947). These characteristics influence woodland distribution, overall productivity, tree density and cover, and competition between understory species and trees (Jameson 1967; Jameson and others 1962; Julius 1999; Pieper and others 1971; Ronco 1990; Thatcher and Hart 1974; West and others 1975). Pinyon-juniper soils are often alkaline, well-drained, shallow and rocky with low fertility (Budy and Meeuwig 1987; Evans 1988; Howell 1941; Meeuwig and Bassett 1983; Pieper 1977). However, on sites with finer, deeper, or more mesic soils, overall productivity is generally higher, leading to greater density and cover of understory shrubs and grasses (Larson 1980; Ronco 1990; West 1999). Pinyon-juniper woodlands display significant heterogeneity in physical and chemical soil properties associated with tree canopy and intercanopy spaces (Barth 1980; Davenport and others 1996; Shukla and others 2006).

Classification - The Southwest Region of the United States Forest Service (1997) delineates 51 associations within the pinyon and juniper series, which occur across a broad climatic and elevational gradient, producing considerable variation in understory composition and structure and associated disturbance regimes (Aro 1971; Moir 1979). Of the 51 associations, 29 are in the pinyon series and 19 are in the juniper series. We do not consider three associations here that are more appropriately described as either grassland (39) or shrubland (50, 51). A nearly verbatim description of these associations, their key criteria for identification, and their location within Region 3 is located in Appendix 12-A.

Moir and Carleton (1987) assigned each of these associations to one or more climate zones, which are also used in several other classifications (Barnes 1983; Jameson and others 1962; Moir 1979; Naylor 1964; Potter 1957; Springfield 1976; Thatcher and Hart 1974; West and Young 2000; Whittaker and Niering 1965). The climate zones delineate season of dominant precipitation (High Sun - summer, Low Sun - winter) and winter soil temperature regime (Cold, Mild):
Low Sun Cold – [Colorado Plateau: northern AZ, north-northwest NM]
1, 2, 3, 5, 7, 8, 11, 12, 13, 14, 15, 35, 40, 41, 42, 44, 45
High Sun Cold – [Southern Rockies: northern NM]
1, 4, 5, 9, 10, 14, 15, 18, 32, 36, 40, 44, 45, 46
Low Sun Mild – [Mogollon Rim: south-central AZ]
2, 19, 20, 21, 22, 23, 30, 37, 49
High Sun Mild – [Madrean: southern AZ, southwestern NM]
4, 5, 12, 14, 15, 24, 25, 26, 27, 28, 29, 31, 33, 34, 37, 43, 44, 45, 47

A number of other classifications have been developed for pinyon-juniper woodlands in specific geographic areas including different National Forests based on understory composition, overstory species dominance, soils, climate, and/or elevation (Barnes 1983; Brown 1982; Dick-Peddie 1993; Donart and others 1978; Hill and others 1992; Jameson 1967; Johnston 1987; Kennedy 1983; NatureServe 2006; NRCS 2006; West and others 1975; Woodin and Lindsey 1954).

An Alternative Classification - Romme and others (2003) identified three pinyon-juniper types based on canopy structure, understory composition, and historic fire regime, summarizing available published information and providing a set of hypotheses for further testing. The types are: pinyon-juniper grass **savanna**, pinyon-juniper **shrub woodland**, and pinyon-juniper forest (hereafter referred to as **persistent woodland**). Pinyon-juniper grass savanna is characterized by sparse juniper and/or pinyon, scattered shrubs, and dense herbaceous growth including perennial grasses, forbs and annuals. The type occurs on deep, fine-textured soils on gentle broad valley bottoms and on gently rolling hills with few barriers to fire spread; it may also occur on rockier sites where productivity is high and understory grasses form a more-or-less continuous fuel layer. The type is common in southern and central Arizona and New Mexico where most of the annual precipitation occurs in the summer, and extends into northern Arizona and New Mexico and elsewhere on the Colorado Plateau where summer and winter precipitation co-dominate (bimodal rainfall pattern).

Pinyon-juniper shrub woodland normally occurs as a mix of trees and shrubs with sparse herbaceous cover but can be characterized by a series of vegetation states that move from herbaceous-dominated to shrub-dominated to tree-dominated over time, unless interrupted by a high severity stand-replacing fire or a mixed-severity fire. The latter returns severely-burned vegetation patches to an herbaceous-dominated state and reduces tree and shrub densities in less severely-burned vegetation patches. The type (including its various vegetation states) occurs on deep, fine-textured soils in valley bottoms and on gentle plains with few barriers to fire spread and on mixed to coarse substrates of variable depth in foothill and lower montane settings. It is common in areas where most of the annual precipitation comes in the winter including northern Arizona and New Mexico but is not restricted to this precipitation regime. In Arizona and New Mexico, the type includes pinyon-juniper/big sage (*Artemisia tridentata*), pinyon-juniper/rabbitbrush (Chrysothamnus nauseosus), pinyon-juniper/chaparral and pinyon-juniper/oak (*Quercus undulata, Q. toumeyi,* or *Q. turbinella*) associations (Appendix 21-A).

Pinyon-juniper persistent woodland is characterized by a multi-age stand structure of pinyons and junipers including very old trees (>300 years old); tree density and canopy cover are high, shrubs are sparse and herbaceous cover is low and discontinuous. This type does not appear to be restricted to particular soil types or climatic conditions but occurs where soils are thin and rocky and do not support a continuous herbaceous or shrub cover and/or where the topography is rugged with significant barriers to fire spread (e.g., cliffs, canyons and extensive areas of exposed rock). Persistent woodland is scattered geographically throughout the Colorado Plateau, southern Rocky Mountains, New Mexico, and in central and northern Arizona. Additional information on historic fire regimes and post-settlement changes for each of these pinyon-juniper types is summarized in *Trends with and without Fire.* Because this classification is based on historic disturbance patterns and is consistent with vegetation composition and structure at relict sites, we will use it throughout this review to distinguish pinyon-juniper woodland types and as a basis for constructing quantitative models of vegetation change (Chapter 21).

Several knowledgeable experts from the US Forest Service have indicated that a fourth type exists in Region 3. The type is similar ecologically to pinyon-juniper grass savanna (e.g, dense herbaceous growth, shrubs absent to scattered) but with a greater density of trees and higher canopy cover (> 10% canopy cover), justifying its classification as an open woodland rather than savanna. Because this type, which we refer to as **pinyon-juniper grass open woodland**, has not been formally described in the literature and because historical tree density and canopy cover values for it have been reported in only a single study (Landis and Bailey 2005), we will adhere to Romme and others' (2003) classification. However, we emphasize that our results and conclusions for pinyon-juniper savanna likely apply to the pinyon-juniper grass open woodland type, especially at the upper ecotone with ponderosa pine forest where an open pinyon-juniper woodland rather than savanna may have occurred historically.

12.2 Historic Range of Variation of Ecological Processes

Vegetation Dynamics - All pinyon-juniper woodlands are affected by multiple disturbance processes including fire, climate, drought, insect infestations, pathogens, herbivory and dispersal. More detailed information about these historical disturbance processes follows this introductory discussion of vegetation dynamics. In this section, we describe succession following stand-

replacing fire (or other major disturbance) in the 3 types, and their fire regimes are described in general terms in the following section, *Trends with fire and without fire*.

Several models of pinyon-juniper succession after stand-replacing fire have been developed for persistent woodland and pinyon-juniper shrub woodlands based on observations of vegetation changes following stand-replacing fires. The studies that gave rise to these models were conducted in west-central Utah, northern Arizona, and southwest Colorado, and it is unclear if they can be applied to other sites in New Mexico and Arizona. After a stand-replacing fire, annuals are usually the first to recolonize the site and may be dominant through the third growing season after fire (Arnold and others 1964; Barney and Frischknecht 1974; Erdman 1970). The area then proceeds through a perennial grass/forb dominated stage by the fifth or sixth year, then to a perennial grass/forb/shrub stage that may persist up to 30 years after fire (Arnold and others 1964; Barney and Frischknecht 1974). Shrubs may appear as early as 11 years after fire if they are resprouters (and were present before) or if they have seeds that are stimulated by fire, such as shrub live oak (*Q. turbinella*), rabbitbrush (*Chrysothamnus viscidiflorus*), and snakeweed (*Gutierrezia sarothrae*; Barney and Frischknecht 1974). Shrubs continue to increase on the post-burn site until they become dominant 35 years after fire, with the establishment of non-resprouting shrubs like sagebrush and bitterbrush; shrub dominance can persist up to 100 years or more post-fire (Barney and Frischknecht 1974; Erdman 1970; Koniak 1985; Schott 1984; Young and Evans 1978). If another fire occurs during the shrub-dominated stage or the site did not previously support shrubs, perennial grasses will dominate until trees attain dominance (Arnold and others 1964; Barney and Frischknecht 1974; Schott and Pieper 1986; Thatcher and Hart 1974).

Junipers are usually the first trees to regenerate after a stand-replacing fire, appearing as early as 11 years post-burn; 45 to 65 years after a stand-replacing fire, shrub mortality begins as competition with juniper intensifies, and juniper continues to increase in density, achieving cover dominance as early as 70 years after fire (Barney and Frischknecht 1974; Barnes and Cunningham 1987; Blackburn and Tueller 1970; Erdman 1970; Floyd and others 2000; Gottfried and Severson 1993; Gottfried 1987; Schott 1984). Alligator juniper, which is common in southern Arizona and New Mexico, may achieve dominance on a post-fire site more rapidly, due to its ability to resprout after fire (Miller 1999). Tree recruitment is facilitated by the foraging and caching behavior of birds and some small mammals, which is discussed in more detail in the *Herbivory* section. Evidence from northern Arizona suggests that roughly 60-80 years after initial juniper establishment, the microclimate around the base of the trees and shrubs has become conducive to pinyon establishment (Landis and Bailey 2005; Padien and Lajtha 1992). Once pinyon establishment begins, their rate of increase commonly exceeds that of juniper (Blackburn and Tueller 1970; Howell 1941; Jameson 1965; Lymbery and Pieper 1983; Meagher 1943; Tausch and others 1981), such that pinyon eventually dominates and may restrict juniper recruitment and growth (Erdman 1970; Schott 1984; Tausch and Tueller 1977). Eventually, mature shrub woodland or persistent woodland forms, with an open or closed canopy and an understory of sparse to common shrubs, some grasses, and forbs (Arnold and others 1964; Erdman 1970). Tress and Klopatek (1987) estimated that it would take approximately 215 years for shrub woodlands to fully recover following stand-replacing fire in northern Arizona, while Erdman (1970) estimated about 300 years for a persistent woodland in southwestern Colorado.

Tree canopy cover values have been reported for the vegetation changes described above. Trees comprised less than 3% canopy cover in all sites surveyed up to 50 years after a fire (Goodrich and Barber 1999). Between 65-90 years post-burn, tree canopy cover was 5-17%, except at one site where *Artemisia tridentata* dominated 90 years after fire and tree cover was < 1% (Arnold and others 1964; Tress and Klopatek 1987). Older, mature pinyon-juniper woodlands with no evidence of recent fire had at least 30% to 43% tree canopy cover. Herbaceous cover and production generally declines with increasing tree canopy (Arnold and others 1964; Lymbery and Pieper 1983; Pieper 1993; Schott and Pieper 1986; Tress and Klopatek 1987; Tausch and West 1995) and with increasing density of woody species (Lymbery and Pieper 1983), although this pattern does not hold for cool-season grasses, which occur more frequently under tree canopies than in open interspaces (Armentrout and Pieper 1988; Clary and Morrison 1973; Pieper 1990).

Vegetation dynamics following fire, including recovery rates, can be altered by fire intensity, fire size, season of burn, pre-burn plant composition, the demographic response of plant species to fire, and past land use history (Barney and Frischknecht 1974; Everett 1987; Pickett 1976; Pieper and Wittie 1990; Zschaechner 1985). For example, if shrubs were present before the burn and perennial grasses were not, shrubs may follow the annual stage (Barney and Frischknecht 1974). Similarly, succession will be slower where the nearest seed source is the edge of a large burn, while burns smaller than 20 acres will regenerate more rapidly (Barney and Frischknecht 1974; Huber and others 1999). In general, successional processes are faster and tree density is greater on depositional sites with deeper mesic soils than on thin, dry soils (Graves 1917; Harper and Davis 1999). Timing of regeneration of shrubs depends on their patterns of post-fire survivorship; that is, rabbitbrush and oak can resprout after fire and rapidly colonizes sites whereas sagebrush and bitterbrush have limited resprouting capability and recolonize over longer time periods via seed establishment (Ward 1977; Wright 1972; Wright and others 1979). Finally, after a non-stand replacing fire, surviving pinyons and junipers, including top-killed junipers that resprout, can generally repopulate the site within 2 or 3 decades (Dwyer and Pieper 1967; Miller 1999; Schott and Pieper 1987; Tausch and Tueller 1977). More work over a broader geographic area is needed on successional patterns in pinyon-juniper shrub and persistent woodlands following severe fire, including how fire severity, time since the last fire, and fire size affect rates and patterns of vegetation recovery.

In contrast to pinyon juniper shrub woodlands and persistent woodlands, vegetation dynamics (succession) in pinyon-juniper savanna following a stand-initiating event, such as a severe fire or drought, have not been explicitly described. However, several studies discussed above indicate that if perennial grasses and forbs were prominent in the pre-burn plant community, as they would be in pinyon-juniper savannas, they will appear first in succession (because of their ability to resprout after fire) and will dominate the site for some period of time (Barney and Frischknecht 1974; Everett 1987; Schott and Pieper 1986). Pinyon-juniper savannas are generally thought to be maintained by frequent surface fire and stand-initiating events are presumably rare since tree ages can exceed 300 to 500 years (Leopold 1924; McPherson 1997; Wilkinson 1971). Tree recruitment occurs in protected sites where fine fuels are limited and/or during fire-free periods that permit young trees to reach a size/height where survivorship is high following surface fire (Dwyer and Pieper 1967; Jameson 1962; Leopold 1924; McPherson 1997). In the absence of fire, shrubs and trees will increase (Humphrey 1953, 1958; McPherson 1997).

These factors together suggest that the vegetation dynamics in pinyon-juniper savanna are similar to those described for shrub- and persistent woodlands with two exceptions: (1) the annual-dominated state is absent as resprouting perennials preempt space and prevent the germination of annuals, and (2) frequent low-intensity surface fires will maintain shrubs and trees at low densities creating an open savanna structure of fire resistant trees and an herbaceous understory.

Trends with fire and without fire - Pinyon-juniper grass savanna (and open woodland) is thought to have been maintained historically by frequent, low-severity surface fires that spread from and into adjacent vegetation types including semi-desert grassland, Madrean pine-oak woodland, and ponderosa-pine forest (Allen 1989; Baisan and Swetnam 1997; Brown and others 2001; Dwyer and Pieper 1967; Jameson 1962; Leopold 1924; Muldavin and others 2003; see HRV documents for other PNVT types). However, some savannas appear to have sparse tree cover due to inadequate soil moisture although more quantitative information is needed to fully support this hypothesis (Johnsen 1962; Romme and others 2007). Livestock grazing coupled later with active fire suppression, mostly in adjacent vegetation types, have reduced fire frequency in many historical savannas resulting in an increase in tree density and canopy cover and a reduced cover and abundance of herbaceous vegetation (Miller 1999). Current stand structures are often dominated by young trees (< 150 years old) with very old trees (> 300 years) present but not numerous. Present-day fires, when they do occur, can be high severity and stand-replacing (Allen 2001; Brockway and others 2002; Dick-Peddie 1993b; Jacobs and others 2002; Leopold 1924). Some extant pinyon-juniper savannas and woodlands were historically grasslands before 1900 (Cottam and Stewart 1940; Dwyer and Pieper 1967; Johnsen 1962; Leopold 1924; Miller 1999; Shaw 2006).

Pinyon-juniper shrub woodland developed after infrequent stand-replacing fire and was most likely maintained by patchy mixed-severity fires that occurred with moderate to low frequency (Arnold and others 1964; Despain and Mosley 1990; Huffman and others 2006a; Tausch and West 1988). The latter fires kept trees and associated shrubs sparse to moderately dense and herbaceous vegetation moderately dense to sparse depending on overstory canopy cover and time since the last fire. When fires did occur, many (to all) trees were killed under a low-frequency more intense fire regime but greater numbers of trees survived under a moderate frequency, moderate-severity regime so that old trees (> 300 years) were normally present but not numerous (Arnold and others 1964; Despain and Mosley 1990; Huffman and others 2006a; Koniak 1985; Miller and others 1995; Miller and Tausch 2001; Romme and others 2003; Tausch and others 1981; Tausch and West 1988). Livestock grazing and active fire suppression have reduced fire frequency in this type resulting in increased tree density and cover and a decrease in shrubs and herbaceous vegetation. Fuel loads have increased as has canopy closure and the connectivity of tree crowns with the result that recent fires “have probably been larger and more severe than those in the late 1800s” (Romme and others 2003).

Persistent woodland developed under an historic regime of infrequent high severity, stand-replacing fire (Floyd and others 2000, 2004; Miller 1999; Muldavin and others 2003; Romme and others 2003; Tress and Klopatek 1987). Recovery is slow following fire with vegetation trending through a series of seral states (Floyd and others 2004). Livestock grazing and active fire suppression have had little impact on fire frequency and severity and presumably little effect

on overstory vegetation composition, density, and structure; however, understory composition has been dramatically altered by past livestock grazing in many persistent woodlands (Floyd and others 2000, 2004; Romme and others 2003). Tree density and canopy cover may have increased in some persistent woodlands as a result of natural recovery from a past stand-replacing fire and/or climatic variation conducive to tree establishment (e.g., wet period in the early 1900's).

Although most persistent woodlands have had little change in overstory structure and composition, evidence that pinyon and juniper have invaded sagebrush and grassland areas and increased in pinyon-juniper savannas and shrub woodlands comes from multiple sources: pollen analyses of wetland sediment cores (Davis and Turner 1986); matched historic photographs including early aerial photos (Davis and Turner 1986; Hastings and Turner 1965; Johnsen and Elson 1979; Miller 1999; Phillips 1963; Rogers 1982; Shaw 2006); historic accounts coupled with stand age-structure information (Cottam and Stewart 1940), and fire-history reconstruction using fire-scarred trees, some coupled with stand structure information (Allen 1989; Brown and others 2001; Despain and Mosley 1990; Huffman and others 2006a; Muldavin 2003). All reconstructions show a cessation or dramatic reduction in the occurrence of fire after 1880 to 1900. Thus, based on local fire histories, this suggests that younger trees, which numerically dominate most current pinyon-juniper stands, have increased due to the lack of wildfire, a result of livestock grazing and/or drought (followed by severe soil erosion) that led to a loss of fine fuels needed to carry fire (Allen 2001; Arnold and others 1964; Brockway and others 2002; Dwyer and Pieper 1967; Gruell and others 1994; Jacobs and others 2002; Jameson 1962). Other hypotheses invoking climate variation and recovery from pre-settlement disturbances (drought, severe fire) have been proposed to explain the post-settlement structural changes observed in historical savannas and shrub woodlands, but these have not been adequately tested. More studies are needed to disentangle the effects of fire exclusion, livestock grazing, climate fluctuations, and recovery from past severe disturbances on the dynamics of tree expansion in the different pinyon-juniper woodland types (Romme and others 2003).

Disturbance Processes and Regimes

Climate

Climate and Pinyon-Juniper Distribution - Analyses of plant macrofossils in packrat middens show dramatic shifts in the geographic and elevational distribution of pinyon-juniper woodland and in its species' composition since the last (Wisconsin) ice age due to climate change. During the last glacial period, juniper and pinyon-juniper woodlands occupied what are now the Sonoran and Chihuahuan Desert lowlands of the Southwest from 300 to 1700 meters (1000 to 5600 feet) in elevation (Betancourt 1984; Betancourt and others 1993; Cole 1981; McAuliffe and Van Devender 1998; Spaulding 1984; Van Devender 1977; Van Devender 1990; Van Devender and Spaulding 1979; Wells 1966). To the north, montane and subalpine forest occupied the Colorado Plateau and Rocky Mountain regions. Beginning approximately 13,000 years ago, pinyons, junipers, and other species in these woodlands were displaced differentially on both latitudinal and elevational gradients in response to increased temperatures and reduced precipitation during the Holocene, producing novel plant associations at individual sites through immigration and local extinction. Range shifts, especially in the distribution of pinyon species, also occurred during the Holocene (Betancourt 1987; Betancourt and Van Devender 1981; Lanner and Van

Devender 1981). Pinyons were extirpated from desert elevations in Arizona and New Mexico at ca. 11,000 years B.P. while junipers and oaks were extirpated at ca. 8000 yrs B.P; all now persist at higher elevations (Betancourt 1987).

Significant expansions and contractions in the distribution of juniper-pinyon woodland also occurred over the last 4,000 years and have been documented by studies of packrat middens, dry caves, pollen cores, lake and wetland sediments and archaeological sites (Van Devender and others 1984; Mehringer and Wigand 1990; Gottfried and others 1995). In general, during periods when the climate became wetter, pinyon and juniper moved down mountain slopes into more xeric communities and grasses increased, and when the climate became drier and warmer, pinyon-juniper declined but grasses persisted. After 500 BP, increased winter precipitation led to a re-expansion of juniper-pinyon woodland that sharply increased after 1700 and again in the early 1900s (Davis and Turner 1986; Eisenhart 2004; Ghil and Vautgard 1991; Mehringer and Wigand 1990).

Climate and Mortality of Pinyon and Juniper - Superimposed on this longer-term climate pattern, interdecadal climate variability has produced multi-decade episodes of wet and dry conditions that have resulted in strong mortality effects (and establishment events, see below) on pinyon-juniper woodlands. The 1950s drought caused a massive die-off of pinyons and some junipers in the middle Rio Grande basin and elsewhere (Betancourt and others 1993). Pinyon mortality was especially high on older trees at drier, lower elevation sites in northern New Mexico (Allen 1989; Betancourt and others 1993; Swetnam and Betancourt 1998). In addition, due to ponderosa pine mortality, the ecotone boundary between ponderosa pine forest (*Pinus ponderosa*) and pinyon-juniper woodland shifted 2 km (1.2 mi) or more upslope in less than 5 years as a result of the 1950's drought in northern New Mexico (Allen and Breshears 1998). The effects of this drought have persisted for more than 40 years as ponderosa pine has failed to re-establish in areas converted to pinyon-pine woodland despite increased precipitation between 1960 and 1990 (Allen and Breshears 1998).

The recent drought, which began in the mid-1990s, has also resulted in regional die-offs of pinyon and, to a lesser extent, juniper during the extreme drought years of 1996 and 2002-2003 (Breshears and others 2005; Gitlin and others 2006; Mueller and others 2005a; Ogle and others 2000). Following an extended period of below-average precipitation (2000-2003) coupled with a regional outbreak of bark beetles (*Ips confusus*), Breshears and others (2005) documented a >20% decrease in the Normalized Difference Vegetation Index (NDVI), a remotely-sensed index of vegetation greenness, at their Mesita del Buey study site near Los Alamos in northern New Mexico and over more than 12,000 km^2 (3 million acres) in Arizona, New Mexico, Colorado and Utah; this decrease was associated with a > 90% mortality of *P. edulis* at Mesita del Buey and tree mortality rates of 40% to 80% elsewhere (Breshears and others 2005). Forest surveys documented pinyon mortality occurring across 8,500 km^2 (2.1 million acres) in Arizona and New Mexico between 2000-2003 and throughout the West; pinyon mortality approached 100% in some areas while other areas experienced little or no mortality (USDA 2005; Shaw and others 2005).

In northern Arizona, pinyon mortality ranged from < 5% to 93% depending on the site and was positively related to soil moisture stress (Gitlin and others 2006; Huffman and others 2006b;

Ogle and others 2000; Swaty and others 2004). Juniper (*Juniperus monosperma*) mortality was generally lower than pinyon mortality due to juniper's greater drought tolerance and competitive ability; juniper mortality rates ranged from 2% to 40% with higher mortality occurring at sites with lower soil moisture and with competing understory grasses (Breshears and others 1997; Breshears and others 2005; Cobb and others 1997; Gitlin and others 2006; Haskins and Gehring 2004; Swaty and others 1998, 2004; Teague and others 2001). In general, larger pinyons were more likely to die than smaller ones and reproductive individuals were more likely to die than non-reproductive ones (Huffman and others 2006b; Mueller and others 2005a). Stands that suffered high pinyon mortality in 1996 also did so in 2002 with mortality positively correlated with the percentage of trees that were reproductive. Differential mortality of pinyon individuals resulted in a decline in the size distribution, a reduction in basal area, a reduced percentage of reproductive trees, and a shift in species dominance in pinyon-juniper stands (Mueller and others 2005a).

Drought-induced changes in the composition and cover of pinyon-juniper woodlands may have a number of important ecological effects including significant changes in carbon stores and dynamics (Breshears and Allen 2002); increased solar radiation and ground temperatures (Martens and others 2000); increased water runoff and erosion (Allen and Breshears 1998; McAuliffe and others 2006); changes in the genetic structure of pinyon and other populations (Mitton and Duran 2004); unfavorable recruitment conditions for pinyon including a reduction in nurse plants, mycorrhizal fungal inoculum, and seed dispersal by birds (Christensen and Whitham 1991, 1993; Haskins and Gehring 2005; Mueller and others 2005a; Swaty and others 2004); and potentially large changes in the composition and function of associated biotic communities (Brown and others 2001; Mueller and others 2005b; Ruel and Whitham 2002; Whitham and others 2003).

Betancourt and others (1993) determined that the 1950s drought was a 200 to 500 year event based on climate reconstructions using the extensive network of tree-ring chronologies in the Southwest coupled with inferences of previous die-off events based on episodes of sparse tree establishment and low survival. Their analysis suggested a similar pinyon die-off associated with the 1580's drought (followed by abundant recruitment during the subsequent wet period) as well as the existence of prolonged droughts in 1667-1681 and 1730-1750 which may have had similar mortality effects. The recent drought (1996-2003) was wetter but warmer than the 1950s drought and in contrast to the earlier one, the recent vegetation die-off was documented at sites near the upper distributional limit of *P. edulis* where precipitation and water availability were generally greater (Breshears and others 2005). Tree mortality during the recent drought may have been exacerbated by the increased density of trees in pinyon-juniper woodlands resulting from increased precipitation between 1978 and 1995 and human-caused fire exclusion; in addition, higher temperatures may have increased moisture stress on trees and sustained pinyon Ips populations at high levels (Breshears and Allen 2002; Breshears and others 2005; Cobb and others 1997).

Climate and Reproduction/Establishment of Pinyon and Juniper - Cone production in pinyons is erratic with large cone crops occurring on average every 1-3 out of 10 years and small cone crops occurring in the intervening years (Barger and Ffolliott 1972; Forcella 1978, 1980; Meeuwig and Bassett 1983). This pattern apparently results from the climatic control of ovulate

cone production (Forcella 1980) with mast crops occurring when temperatures are below-average (> 1 standard deviation below mean) during the last week of August and the first two weeks of September. The masting habit of pinyons may function not only to satiate seed predators (Forcella 1980) but also to ensure successful seed dispersal since avian dispersers are preferentially attracted to individual pinyons and pinyon populations that produce more cones (Christensen and Whitham 1991; Vander Wall and Balda 1977). We could find no studies investigating the relationship between juniper seed production and climate although heavy crops of juniper seed are reported to occur every 2 to 5 years (Tueller and Clark 1975).

Scant information exists on climatic factors that favor seedling germination and establishment in pinyons and junipers. Arnold and others (1964) stated that a combination of good seed years and favorable moisture conditions were needed; in support of this, they observed that many juniper stands were initiated in 1905 and 1919 when annual precipitation was 1.5 times higher than average. Similarly, Eisenhart (2004) reconstructed historical stand-age structures across the Uncompaghre Plateau and identified a period of high tree establishment in the early to mid-1700s, a time marked by increased precipitation following a period of severe drought. Finally, Muldavin and others (2003) investigated pinyon recruitment patterns in pinyon-juniper woodland in the Oscuras Mountains, southern New Mexico, and found distinct cohorts of trees within stands that established during non-drought periods that coincided with fire-free intervals.

Fire - Baker and Shinneman (2004) provide a comprehensive review of the role fire plays in pinyon-juniper systems, including fire history and severity, in 11 western states; their review is structured around a series of questions. We briefly summarize information pertaining to a subset of those questions that are relevant to the historic range of variability of fire frequency and intensity, but we consider a more restricted geography (see *General Description*) and include new information.

Following Baker and Shinneman (2004), the questions are:

1) Do junipers and pinyons accurately record fires by means of fire scars?
Fifteen studies provide evidence on this question (Table 12-1). Thirteen of the fifteen found fire-scarred trees; three of these thirteen studies reported fire scars to be common and another three studies found large numbers of fire-scarred trees (> 20 scarred trees). Four of the latter six studies occurred in lower elevation or upper ecotone savanna-open woodland suggesting that surface fires (or mixed-severity fires with a strong surface fire component) may be more frequent in these settings. Four studies reported no or few (<2) fire scars, all of which occurred in shrub or persistent woodland sites in northern Arizona and southwest Colorado; two of these sites were located in an upper ecotone setting where there were significant barriers to fire spread (Despain and Mosley 1990; Floyd and others 2000; Romme and others 2003; Rowlands and Brian 2001). Shrub woodland sites in the upper ecotone that lacked significant barriers to fire spread, however, had large numbers of fire-scarred trees suggesting that surface (or mixed-severity) fires may be more frequent in this type when fires can spread (Huffman and others 2006a).

Variability in the abundance of fire scars may be related to the abundance of surface fires (Baker and Shinneman 2004). In support of this hypothesis, Floyd and others (2000) and Romme and others (2003) failed to find fire scars on junipers and pinyons as well as on nearby ponderosa

pines (which are known to be good recorders of surface fires) suggesting the absence of these fires at their site. However, Muldavin and others (2003) and Huffman and others (2006a) found fire-scarred pinyons to record fewer fires than adjacent ponderosa pines consistent with suggestions by Gottfried and others (1995) that pinyons may be poor(er) recorders of surface fire. Junipers appear to scar well, but due to the presence of false rings and difficulties with cross-dating they have not been used widely in fire history reconstructions (Brown and others 2001; Despain and Mosley 1990; Gottfried and others 1995). Baker and Shinneman (2004) conclude that without a more systematic study of how fire-scar evidence is left by contemporary fires on pinyon and juniper, the accuracy of fire scars in recording surface fires (and estimating their abundance) cannot be fully resolved.

2) *Did spreading low severity fires occur in pinyon-juniper woodlands and what is the fire return interval (FRI)?*

Evidence for low-severity surface fire comes from direct observations of fire behavior and fire history reconstructions using fire-scarred trees. There are 5 published studies reporting direct observations of low severity surface fire in pinyon-juniper woodland, mostly in savanna or upper ecotone settings. One was a recent, naturally-ignited fire (the South Canyon Fire) in western Colorado which began as a backing surface fire moving through scattered pinyon-junipers with a grassy understory and turned into a high severity fire as it moved upslope and up-canyon through thicker, closed canopy pinyon-juniper-wavy oak fuels (Butler and others 1998). Two other studies reported human-caused, low severity surface fires in savanna settings in central New Mexico and southern Arizona (Johnson and others 1962; Dwyer and Pieper 1967). In northern Arizona, Phillips and Mulford (1912) stated that “surface fires are the most common, but crown fires sometimes occur” in pinyon-juniper areas. Finally, on the Uncompahgre Plateau in western Colorado, Hoffman (1921) stated that “countless ground fires have run in past years.” Interestingly, Baker and Shinneman (2004) attempted to locate fire scar trees across the Uncompahgre Plateau and found them to be “rare or absent, even where trees several hundred years old were still present.”

Table 12-1. The occurrence of fire-scarred trees by species, where scars were found, and the setting, location, and elevation where studies were conducted. Data relevant to question 1 (see Fire section); table modified from Baker and Shinneman (2004) including new information. Abbreviations are: P = present; C = common.

Study	State	Elevation (m)	Number of scarred trees			Where fire scars were found			Type of setting		
			Juniperus sp.	Pinus edulis	PIPO	On burn edges	In shrub- or persistent woodlands	In savannas, lower elev.	Savanna lower elevation	Shrub- or persistent woodland	Upper ecotone
Allen (1989)	N NM	2016-2048	P	P	9						X
Arnold and others (1964)	N AZ	--		> 1		X				X	
Baisan and Swetnam (1997)	C NM	2225-2380	>1	>1	21 a.						X
Despain and Mosley (1990)	N AZ	2040	0	0	1		X			X	X b.
Floyd and others (2000); Romme and others (2003)	SW CO	2060-2485	0	0	0					X	X
Gottfried and others 1995	C NM	1900-2100					X			X	
Johnsen (1962)	N AZ	1430-1980	P					X	X		
Leopold (1924)	S AZ	--	C					X	X		
Miller (1999)	SW NM	1750-2983	C					X	X		
Rowlands and Brian (2001)	N AZ	1769-1867	0	0						X	
Tausch and West (1988)	SW UT	2000	26	2						X	
Wilkinson (1971); Brown and others (2001)	S NM	2400-2440	P	10							X
Muldavin and others (2003)	S NM	--		9	9		X			X	X
Huffman and others (2006a)	N AZ	2005-2073	6	18	96		X			X	X
Huffman and others (2006a)	N NM	2347-2438	14	23	23		X			X	X
Huffman and others (2006b)	N AZ	1900-1950	P	P							

a. No indication from authors whether fire-scarred cross-sections were collected from ponderosa pine, pinyons or junipers.
b. Study site dissected by a steep canyon that represents a significant topographic barrier to fire spread.

Fire history reconstructions using fire-scarred trees normally provide the strongest evidence for existence of low intensity surface fires (e.g., Covington and Moore 1994; Swetnam and Baisan 1996). Two components are needed to clearly establish that a fire event was a surface fire: a reliable fire date recorded by two or more trees (points) and stand-structure information between these trees or points indicating the fire did not kill most overstory trees (Baker and Shinneman 2004). Without multiple trees recording a fire date, it is difficult to know whether the fire event was a spreading surface fire or a patchy surface fire of small extent. Likewise, without stand structure information, it is difficult to determine whether the fire that caused the scarring was a low-severity surface fire or a mixed-severity fire particularly as the distance between fire-scarred trees increases; what is clear is that in the vicinity of the fire-scarred tree fire severity was low. However, despite its importance, most fire history reconstructions for higher-elevation forest systems where low intensity surface fires are believed to dominate the historic fire regime also lack spatially explicit stand-structure information (e.g., Fulé and Covington 1996, 1999; Swetnam and Baisan 1996; Swetnam and others 1992).

Five studies with cross-dated fires have estimated the frequency of surface (or mixed-severity) fires in upper ecotone settings, although all but one study lack age-structure information (Table 12-2). Allen (1989) found 13 cross-dated fires recorded in ponderosa pine that spatially bracketed intervening pinyon-juniper woodland, and Baisan and Swetnam (1997) recorded 6 or more fires that simultaneously scarred multiple trees (unspecified as to whether they were pinyon or ponderosa pine) across their study area. Similarly, Wilkinson (1971) and Brown and others (2001) found 19 fire dates recorded by individual pinyon trees, five of which coincided with widespread fire years from ponderosa pine and mixed conifer sites (> 25% of sites recording fire) but only 3 fires (fire dates) scarred more than an individual pinyon in their sample. Finally, Huffman and others (2006a) found 8 fires recorded by multiple pinyons and ponderosa pine at their Tusayan site prior to 1890 and only 2 such fires at their Canjilon study site; accompanying age structure information suggested that fires that spread into upland shrub woodland did not result in large patches of tree mortality at the former site whereas at the latter site, fires were "often small in extent and probably occurred as patchy surface fire to mixed-severity fire that killed groups of trees or small stands". Although many of the fires recorded in the above studies were likely low severity surface fires, in those studies without stand age structure information and fire dates recorded by multiple trees, the possibility of mixed-severity fires or patchy surface fires of limited extent cannot be ruled out (Baker and Shinneman 2004; Huffman and others 2006a).

Quantitative analyses of fire scarred trees have yielded variable estimates of the mean fire return interval (MFI) for low severity surface (or mixed-severity) fires in pinyon-juniper systems (Table 12-2). Some of this variation results from the use of different methods for estimating fire frequency. A composite estimate of MFI, derived from pooling all fires recorded on trees in the study area, assumes that the fire-scar record on trees is incomplete (i.e., there are unrecorded fires) and attempts to correct for this by "compositing" (Dieterich 1980). When the occurrence of unrecorded fires is high, the composite estimate of MFI may provide a close approximation to the population (true) MFI. However, because the probability of unrecorded fires is unknown, the composite estimate of MFI has an equivocal interpretation which is that a fire burned somewhere in the stand at a specified frequency (Baker and Ehle 2001). Also, in ponderosa pine and pinyon fire histories, most fires are recorded by 1 or 2 trees suggesting they are small but any fire, large or small, decreases the value of the fire interval by the same amount in a composite estimate. In addition, the composite estimate decreases with increasing sampling area and sample size as

more fires are found and t can be reduced if investigators target multiple-scarred trees and areas with a high density of fire scars (Arno and Petersen 1983; Baker and Ehle 2001). For all of these reasons (e.g., small fires, large sampling area and sample size, targeted tree sampling), the composite estimate of MFI may significantly overestimate mean fire frequency on a stand- or patch-level. A restricted composite ($\geq$ 25% or $\geq$ 10% of trees showing evidence of fire in a particular year) partially offsets these problems and likely provides a better estimate of the stand-level fire frequency. Finally, if all fires are recorded by fire scars, the individual tree estimate of MFI, a pooled average of mean fire return intervals calculated for individual trees (also referred to as the point mean fire interval, PMFI) provides an accurate estimate of MFI but if unrecorded fires occur, which is likely, it underestimates the stand-level frequency (Baker and Ehle 2001; Brown and others 2001; Romme and others 2003; Swetnam and Baisan 1996). In the face of this uncertainty, Baker and Ehle (2001) recommend bracketing estimates of MFI with the restricted composite (shorter fire interval estimate) and the individual tree estimate (longer interval estimate).

Keeping the above methodological and interpretational issues in mind, estimates of MFI can be made for each pinyon-juniper type. One study using an individual tree estimate (for a single fire-scarred tree) put the mean fire interval at 10 years for lower elevation juniper savanna (Table 12-2); Leopold (1924) also reported that fire scars were common in juniper savannas in southern Arizona foothills and argued that frequent, spreading fires kept brush down and tree density low. This view is supported by newspaper accounts of large fires that burned through juniper-oak communities in southern Arizona between 1859 and 1890, removing brush, thinning trees and maintaining the open savanna character of these landscapes (Bahre 1985, 1991). Four studies in upper ecotone pinyon-juniper savanna (or open woodland) report mean fire return intervals of 6 to 43 years with a range of 3 to 60 years for successive fires; one of these studies uses a composite, another uses a restricted composite, and two use an individual estimate (Allen 1989; Baisan and Swetnam 1997; Brown and others 2001; Muldavin and others 2003).

No studies have attempted to reconstruct the frequency of low severity surface fire in pinyon-juniper savannas and open woodlands away from either upper or lower ecotone. However, in southern Arizona and southwestern New Mexico, Madrean pinyon-juniper-oak woodlands typically support a perennial grass-dominated understory and are sandwiched in elevation between semi-desert grassland and Madrean pine-oak woodland, both of which sustained a mean fire return interval consistent with Leopold's (1924) estimate for juniper savanna (Humphrey 1953, 1958; Kaib and others 1996; Swetnam and others 1992; Swetnam and Baisan 1996). In addition, historic photographs for Madrean encinal and Madrean pine-oak woodlands show abundant grass in the understory and open canopy-woodlands (Schussman 2006). Based on local fire history and on the continuity of fuels (which still exist today), it is likely that fire burned into and through these juniper-pinyon-oak woodlands from the grassland or higher-elevation Madrean pine-oak woodlands at a mean FRI equal to or less than the 10 years suggested by Leopold (1924).

Table 12-2. Information pertaining to fire-history reconstructions in pinyon-juniper systems including type of setting, mean fire interval (MFI), and method of fire interval estimation: CR is restricted composite, C is composite, and I is individual tree estimate. Data relevant to question 2 (see *Fire* section); table modified from Baker and Shinneman (2004) including new information. Other abbreviations include: Y = yes; N = no; M = many; and U = unknown.

Study	State	Elevation (m)	Fire history reconstructions				Type of setting			MFI in years; (min, max FI)	MFI method
			Cross-dated fires	Age structure	No. of scarred trees	No. of spreading fires	Savanna lower elevation	Shrub- or persistent woodland	Upper ecotone		
Allen (1989)	N NM	2016-2048	Y	N	9	13			X	16 (3, 28)	CR
Baisan and Swetnam (1997)	C NM	2225-2380	Y	N	21	M			X	6-12	C
Despain and Mosley (1990)	N AZ	2040	Y	Y a.	1 a.	U		X	X	--	--
Floyd and others (2000); Romme and others (2003)	SW CO	2060-2485	N	Y	0	0		X	X	--	--
Leopold (1924)	S AZ	--	N	N	M	U	X			10	I
Rowlands and Brian (2001)	N AZ	1769-1867	N	N	0	0		X		--	--
Tausch and West (1988)	SW UT	2000	Y	Y	27	U					
Wilkinson (1971); Brown and others (2001)	S NM	2230-2440	N	N	8	U			X	27.5 (10, 49)	I
Muldavin and others (2003)	S NM	--	Y	Y b.	9	U		X	X	43 (30-60) c.	I
Huffman and others (2006a)	N AZ	2005-2073			43	M		X	X	11.6 , 41.6 d.	CR , I
Huffman and others (2006a)	N NM	2347-2438	Y	Y	24	M		X	X	22.5 , 81 d.	CR , I

a. Dated burned snags to determine fire dates and nearby trees to determine date of stand establishment relative to fire date.
b. Stand structure data is available for the Oscuras Mtns. site (persistent woodland setting) but not the San Andres Mtns. site (upper ecotone setting).
c. MFI is for the San Andres Mountains site.
d. Huffman and others (2006a) indicate that the composite estimate of MFI applies to ponderosa pine and pinyon growing in wetter drainage bottoms at their n. Arizona site; the mean individual estimate (PMFI) is also heavily influenced by ponderosa pine and pinyon in this setting (n = 16 trees out of 20 trees) but the mean sample interval for individual junipers ranged from 35 to 111 years (n = 4). In contrast at the n. New Mexico site, a more balanced sample of fire-scarred pinyons and ponderosa pine was collected and these were found at "various microsites, distributed more or less generally across the study area." Thus, the composite MFI and PMFI may be reasonable estimates of the historical fire frequency at the site as a whole.

In another study, Huffman and others (2006a) provide both a restricted composite (≥ 10% of trees scarred) and an individual tree estimate of the mean fire interval for pinyon-juniper shrub woodland at two upper ecotone sites (Table 12-2); the PMFI is 3.6 times longer than the restricted composite estimate at both study sites indicating how different the methods can be in their estimation of a stand-level mean fire return interval. In addition, Huffman and others (2006a) argue that their estimates of MFI for surface fires at the northern Arizona site (MFI = 11.6 years, 41.6 years) best apply to canyon bottoms where ponderosa pine and pinyon occur and that fires spread less frequently into upland areas where pinyon and Utah juniper dominated, as suggested by the relatively longer PMFI values for juniper there (35 to 111 years). Despite these longer intervals, Huffman and others (2006a) report that fires that historically carried into upland pinyon-juniper probably occurred as patchy low severity surface to mixed-severity fires that did not result in large patches of tree mortality, basing this conclusion on stand structure information. In contrast to their northern Arizona site, fire-scarred ponderosa pine and pinyon were distributed across the northern New Mexico study site and not restricted to particular microsites; the estimates of MFI for this site are 22.5 to 81 years (Table 12-2). Huffman and others (2006a) suggest that the fire regime at their northern New Mexico site was characterized by a combination of long-interval patchy crown fires which removed groups of trees and small stands and more severe mixed-severity fires compared to the northern Arizona site.

Fire-history reconstructions in pinyon-juniper systems that rely on ponderosa pine may bias estimates of MFI since ponderosa pine often occurs along drainage bottoms and in wetter sites than pinyon and especially juniper so fires that were recorded in the former settings may not have spread into adjacent upland/drier pinyon-juniper savanna or woodland sites (Baisan and Swetnam 1997; Baker and Shinneman 2004; Huffman and others 2006a). This concern probably does not apply to Allen (1989) who used a restricted composite estimate of MFI and reported that fire-scarred ponderosa spatially bracketed pinyon-juniper stands in his study area such that fires recorded by multiple trees likely had to burn through adjoining pinyon-juniper stands (Tables 12-1 and 12-2).

Additional evidence for mixed-severity fire comes from three studies at three sites although no estimates of MFI were made. Two of the three sites are pinyon-juniper shrub woodlands occurring in northern Arizona and southwest Utah while the remaining one is a persistent woodland in south-central New Mexico. Despain and Mosley (1990) analyzed cross-sections taken from dead, burned trees in combination with stand age-structure information and reported patchy fires that left surviving trees in northern Arizona. Similarly, Muldavin and others (2003) dated dead, fire-scarred trees that recorded fires in 1717, 1731, and 1798 in south-central New Mexico and found older, surviving trees within the burned stand as well as adjacent stands that were initiated after 1798. Tausch and West (1988) found junipers but no pinyons that survived an 1830 fire in southwestern Utah and estimated juniper survivorship at 37%; furthermore, an estimated 14% of junipers that had recruited prior to an earlier fire (1658) survived both fires. Finally, Huffman and others (2006a) combined analyses of fire scarred trees with maps of different aged stands and charred tree structures to argue that historic fires at their northern Arizona site occurred as patchy low severity surface- to mixed–severity fires that killed small groups of trees but left many survivors, whereas at their northern New Mexico site, the fire regime was characterized by a combination of long-interval crown fires and severe surface fires that left fewer surviving trees.

3) *Did high-severity fires occur in pinyon-juniper woodlands prior to Euro American settlement and what was the fire rotation for these fires?*

At least 31 pre-settlement fires were documented as high-severity in at least a portion of the burn area (Table 12-3). At least 22 of these were probably mixed-severity fires, that is, the fire was stand-replacing in some areas, killing groups of trees, but leaving surviving trees along with dateable charred snags or fire-scarred trees in others (Despain and Mosley 1990; Huffman and others 2006a; Muldavin and others 2003; Tausch and West 1988). Excluding the Huffman and others (2006a) study, which reports the majority of these fires (thereby strongly influencing the results), at least 16 pre-settlement high severity fires have been documented in other studies and at least 7 of these were probably mixed-severity fires. Pre-settlement, high severity (stand-replacing) fires have been reported in Mesa Verde National Park in southwestern Colorado; Sheeprock Mountain Range in central Utah; the Canjilon Ranger District in northern New Mexico; and the Sacramento, Oscuras, and San Andres Mountains in southern New Mexico. Pre-settlement mixed-severity fires have been documented in the Needle Range of southwestern Utah; Oscuras Mountains in southern New Mexico; the Canjilon Ranger District in northern New Mexico; and the Tusayan Ranger District, Walnut Canyon National Monument and Grand Canyon-Parashant National Monument near Mt. Turnbull in northern Arizona.

Many of the high-severity fires killed all trees within the burn area while others left islands of surviving trees in rocky areas with thin soils, on ridges, and in areas adjacent to topographic breaks (Table 12-3). Documentation of pre-settlement high severity fire has depended on establishing all of the following: 1) a fire date using fire-scarred trees at the burn perimeter or charred snags within the burn area, 2) the absence of older, surviving trees in the burn area, and 3) tree recruitment after the fire date (Baker and Shinneman 2004; references in Table 12-3). Historic photographs can also provide strong evidence of high-severity fires (Romme and others 2003; Figure 12-1).

A fire rotation or turnover time is defined as the time it takes to burn over an area equivalent to a particular landscape once; thus some areas in the landscape may burn more than once and others not burn at all during the time period defined as a rotation (Baker and Ehle 2001; Baker and Shinneman 2004). There is only a single study that estimates this parameter for high-severity stand-replacing fire in persistent woodland. Using reconstructed fire maps and the area of pinyon-juniper woodland burned from 1949 to 1999, Floyd and others (2004) estimated a turnover time (or frequency) of approximately 400 years for high-severity crown fires. Although this estimate was based on fires burning after 1880, the authors argued that stand replacing fires were the norm and hadn't changed in frequency prior to Euro American settlement based on: 1) extensive ageing of pinyons and junipers growing underneath or adjacent to charred snags to obtain an approximate fire date coupled with age structure information showing that stands had established after the fire; and 2) early photographs showing clear evidence of turn-of-the-century stand-replacing fires as well as a very dense forest structure that was indistinguishable from present-day stands.

Two studies permit estimates of the turnover rate for mixed-severity fires in shrub woodlands. Despain and Mosley (1990) reported fires in 1804, 1832, 1862, and shortly after 1880 for their study site in northern Arizona, stating that "fire occurred throughout most of the study area within the past two centuries in all cases"; the mapped extent of the 19th century fires and the fact that no fires were recorded after 1900 suggests that the turnover rate for mixed-severity fire was

100+ years. Huffman and others (2006a) suggested that PMFI values represent reasonable estimates for fire recurrence, putting the turnover rate for mixed-severity fire at 35 to 111 years for the northern Arizona site if only juniper samples are used and 81 years for the northern New Mexico site. The values for the northern Arizona site bracket the PMFI estimate (42 years) calculated by Huffman and others (2006a) using fire-scarred ponderosa pine, pinyon and juniper.

Synthesis – Because pinyons are poorer recorders of fire than ponderosa pine (e.g., lower post-fire survivorship, more unrecorded fires) and junipers, which appear to scar well, are difficult to age, fire history reconstructions in pinyon-juniper savannas and woodlands have proved challenging. In addition, a number of authors have commented on the rarity of fire scars in pinyon-juniper systems but there is little quantitative data documenting the abundance of fire scars in pinyon-juniper systems across the Southwest. Fire scars have been reported in 13 out of 15 studies surveyed with scarred trees tending to be common in lower elevation and upper ecotone pinyon-juniper grass savannas (or open woodlands) and in upper ecotone shrub woodlands with no topographic barriers fire spread. Fire scars were absent or uncommon in shrub or persistent woodland sites, often at the upper ecotone where there were significant topographic barriers to fire spread. Variability in the abundance of fire scars may indicate variation in the occurrence (frequency) of surface fires or mixed-severity fires that leaves surviving overstory trees or, alternatively, site-specific variation in the recordability of fires resulting from, for example, difference in understory fuels. There is a dearth of fire history information on surface or mixed-severity fires away from the ecotones.

Fire history reconstructions have provided a range of estimates for fire frequency but most studies have methodological problems that make inferences less certain. These include: i) the use of ponderosa pine growing in more mesic settings in reconstructions to infer fire frequency in drier pinyon-juniper stands (estimates likely biased toward shorter intervals); ii) use of composite estimates for MFI which are sensitive to sampling area, sample size and the occurrence of small fires that tend to bias estimates toward shorter fire intervals; iii) targeted sampling of multiple-scarred trees and areas with a high density of fire scares biasing estimates toward shorter intervals and iv) use of individual tree estimates which increase the likelihood of unrecorded fires biasing fire frequency estimates toward longer intervals. In addition, most fire history studies lack spatially explicit age-structure information so that it is uncertain whether the fire that caused a scar was a surface fire or a mixed-severity fire.

Given the methodological issues cited above, bracketing fire frequency estimates using those biased toward shorter intervals (restricted composite estimate) and longer intervals (individual tree estimate) has been recommended to span the limits of where the population MFI lies (Baker and Ehle 2001). Doing this, the mean fire interval for lower elevation pinyon-juniper woodland is 10 years (single study, single fire-scarred tree); for pinyon-juniper grass savanna (and open woodland), MFI ranges from 12 to 43 years while for shrub woodland, MFI ranges from 23 to 81 years (excluding the restricted composite MFI estimate for Huffman and others' (2006a) northern Arizona site, see Table 12-2 for explanation). Shorter MFI's suggest a fire regime characterized by low-severity surface fires while longer intervals suggest moderate intensity, mixed-severity fires that killed groups of trees or small stands but left surviving trees in less severely burned patches (Huffman and others 2006a). Evidence for low severity surface fires comes from direct observations of fire behavior in pinyon-juniper savanna-woodland settings in central New Mexico, southern and northern Arizona, and western Colorado while evidence for

mixed-severity fires come from fire-history and stand reconstruction studies in pinyon-juniper shrub woodlands settings in northern Arizona, northern New Mexico and southwestern Utah.

High severity pre-settlement fires have also been documented, but many of these were mixed-severity fires that burned with high severity in groups of trees or small stands but left many surviving trees in other areas. Pre-settlement high-severity fires that were largely or entirely stand-replacing have been reported in pinyon-juniper shrub and persistent woodlands in northern and southern New Mexico, northern Arizona, southwestern Colorado and central Utah. One study in persistent woodland estimated the turnover time for these high-severity fires at 400 years.

Figure 12-1. Photograph taken in 1934 in the western portion of Mesa Verde National Park, near an area that burned in that year on Wetherill Mesa. The photo is not of the 1934 burn, but shows an area that was burned at an unknown time prior to Park establishment in 1906. Fire history reconstructions (Floyd et al. 2000) suggest that the area in this photo probably burned in the 1880s. Note the edge of dense, unburned pinyon-juniper forest in the background. (Photo courtesy of Romme and others 2003).

Table 12-2. Studies reporting high severity and mixed-severity fires before Euro American settlement (1880). Data relevant to question 3 (see *Fire* section); table modified from Baker and Shinneman (2004) including new information. Abbreviations include: U = unknown

Study	State	Elevation (m)	Reconstructed	Observed	Pre-Euro/American settlement	Post-Euro/American settlement	High severity	Unburned islands	Mixed-severity
Arnold and others (1964)	N AZ	1800	16		2	14 a.	X	Yes	No
Aro (1971)	NW CO, NE UT			2		2	X	---	----
Barney and Frischknecht (1974)	C UT	1775-2375	28		5	23	X	No	No
Butler and others (1998)	C CO	1775-2100		1		1		No	Yes
Despain and Mosley (1990)	N AZ	1890-2075	4		3	1 a.			Yes
Erdman (1970); Floyd and others (2000); Romme and others (2003)	SW CO	2060-2485	3	8	3	8	X	No	No
Goodrich and Barber (1999)	NE UT	--	1	2		3 b.	X	Yes	---
Hester (1952)	W CO	--		1		1	X	---	---
Hoffman (1921)	W CO	---		U		U	X	---	---
McCulloch (1969)	N AZ	1950-2133		1		1	X	No	No
Phillips and Mulford (1912)	C AZ	---	U			U	X	---	---
Tausch and West (1988)	SW UT	2000	2		2				Yes
Wilkinson (1971), Brown and others (2001)	S NM	2400-2420	1			1	X	---	---
Muldavin and others (2003)	S NM	----	> 1 (46 c.)		> 1 (23 c.)	> 1 (23 c.)	X	---	Yes
Huffman and others (2006a)	N NM	2347-2438	15 d.		15 d.	8	X		Yes

a. The earliest of these post-settlement fires occurred in 1885 and likely represents historic fire behavior: stand-replacing in the case of Arnold and others (1964) and mixed-severity in the case of Despain and Mosley (1990).

b. The earliest of these post-settlement fires occurred in 1894 and likely represents historic fire behavior.

c. Number of distinct fire patches reconstructed from aerial photographs which likely overestimate the total number of stand-replacing fires; most of the patches were at lower elevations at the ecotone of pinyon-dominated woodlands and juniper savannas along foothill slopes and valley bottoms.

d. Huffman and others (2006a) reported evidence of at least one pre-settlement stand-replacing fire at their northern New Mexico site and this fire was less than 10 ha (D. Huffman, pers. comm.); the rest of the fires were likely mixed-severity fires based on fire history reconstruction and the distribution of PMFI's for fire-scarred trees; the authors also reported recurrent mixed-severity and surface fires historically at the northern Arizona site but because it is impossible to distinguish between these two types of fire from their data we did not include this information in the table.

Hydrology – We found no studies that documented hydrological processes such as flooding as important historical ecological determinants for pinyon-juniper woodland.

Herbivory – Birds and small mammals are the primary predators on juniper berries and pinyon nut crops (Arnold and others 1964; Balda 1987; Gottfried and others 1995; Johnsen 1962; McCulloch 1969; Noble 1990; Salomonson 1978; Scott and Boeker 1977; Short and others 1977). Mule deer (*Odocoileus hemionus*), white-tailed deer (*Odocoileus virginianus*) and elk (*Cervus canadensis*) consume leaves and seeds of both species and they browse associated woodland grasses, forbs and shrubs including mountain mahogany, Gambel's oak, wavyleaf oak, big sagebrush, and cliffrose (Martin and others 1961; Short and McCulloch 1977). We could find no studies that documented the historical effect of herbivores on vegetation structure and composition in pinyon-juniper systems. However, after the extirpation of wolves (*Canis lupus*) and grizzly bears (*Ursus arctos*) during the early 1900s, mule deer severely damaged shrub and tree species in a heavily-used wintering area in northern Arizona, reducing the density and vigor of juniper, cliffrose, sagebrush, and pinyon while increasing the abundance of grasses and other herbaceous species (Merkle 1952; Rasmussen 1941).

In addition to acting as predators, birds are considered the most important dispersal agents of juniper and pinyon, transporting seeds away from the parent plant to microsites that are suitable for germination. Juniper seeds that passed through the digestive tract of birds and other herbivores germinated faster than uneaten seeds (Johnsen 1962). Scrub jays (*Aphelocoma californica*), pinyon jays (*Gymnorhinus cyanocephalus*), Steller's jays (*Cyanocitta stelleri*) and Clark's nutcrackers (*Nucifraga columbiana*) are the primary dispersers of pinyon seeds, acting as seed predators when harvesting small cone crops but as dispersal agents during mast crop years, caching hundreds of thousands of pinyon seeds, many of which are never recovered (Balda and Bateman 1971; Ligon 1978; Vander Wall and Balda 1977). Scrub jays preferred to cache pinyon seeds in the soil near and under pinyon and juniper trees rather than in the open while pinon jays preferred to cache seeds in the open "where standing trees are few" (Hall and Balda 1988; Ligon 1978). Pinyon germination and survivorship were greater under the shade of a nurse plant or woody debris, but subsequent growth was faster for pinyons that established under shrubs compared to under trees (Callaway and others 1996; Meeuwig and Bassett 1983; Padien and Lajtha 1992). Pinyons have evolved a number of characteristics that increase the efficiency (and likelihood) of seed dispersal including mast cone crops that overwhelm invertebrate and vertebrate seed predators ensuring both dispersal and subsequent recruitment (Ligon 1978; Vander Wall and Balda 1977).

Cone production by pinyons varies between individuals within a population and between populations; avian seed dispersers respond to this variation, harvesting more cones and a greater percentage of cones from trees producing larger cone crops both in the laboratory and in the field (Christensen and Whitham 1991, 1993; Christensen and others 1991). In addition, small mammals, like cliff chipmunks (*Neotamias dorsalis*) and rock squirrels (*Spermophilus variegatus*), compete with birds and stem- and cone-boring insects (primarily *Dioryctria albovittella*) for pinyon cones and seeds (Christensen and Whitham 1993).

Predator/Prey extinction and introductions – We could find no studies that implicated predator/prey extinctions and introductions as important historical ecological determinants in pinyon-juniper woodlands (but see preceding section, *Herbivory*).

Insects/Pathogens – Ronco (1990), Rogers (1995), Negron (1995), Gottfried and others (1995), Weber and others (1999) and Shaw (2005) provide lists of insects, pathogens, and plant parasites that attack pinyon and juniper. For pinyon, they include pinyon stem- and cone-moth (*Dioryctria albovittella*); pinyon cone moth (*Eucosma bobana*); pinyon Ips; pinyon twig beetles (*Pityophthorus sp.* and *Pityogenes sp*.); pinyon needle miner (*Coleotechnites edulicola*); pinyon needle scale (*Matsucoccus acalyptus*); black stain root disease (*Leptographium wageneri*); and pinyon dwarf mistletoe (*Arceuthobium divaricatum*). For junipers, they include twig beetles (*Phloeosinus sp*.); twig pruners (*Stylox bicolor*); western cedar borer (*Trachykele blondeli*); juniper mistletoe (*Phoradendron juniperinum*); and rusts (*Gymnosporangium sp.*), the later causing galls, leaf damage and branch excrescences on most species of junipers. Although the life cycles of these pests are known in some detail, little information exists on their impacts to pinyon populations locally or regionally or on the environmental factors that control their populations, including outbreaks, except for a few species included below.

Pinyon Ips, pinyon twig beetles, pinyon needle miners, and pinyon needle scale are normally present in low numbers in pinyon-juniper woodlands but under conditions of drought-induced water stress outbreaks may occur causing defoliation and/or mortality at the watershed-, landscape- and regional scales (Allen and Breshears 1998; Furniss and Carolin 1977; Gottfried and other 1995; Hagle and others 2003; Rogers 1995; Waring and Cobb 1992; Wilson and Tkacz 1992). Most insect-related pinyon mortality in the West is caused by pinyon Ips (Rogers 1993). A severe outbreak of the species was reported at Bandelier National Monument in northern New Mexico during the 1950's drought, resulting in patchy mortality, mostly of trees > 100 years old, on drier, lower elevation woodland sites (Allen 1989; Swetnam and Betancourt 1998). More recent outbreaks of pinyon Ips associated with the extreme drought years of 1996 and 2000-2003 were reported in northern and eastern Arizona, northern New Mexico, southeastern Utah and western and southern Colorado, causing widespread pinyon mortality as reported in the *Climate* section.

Individual and stand-level attributes associated with the probability of infestation by pinyon Ips after the 1996 drought were investigated by Negron and Wilson (2003) in northern Arizona. Pinyon stand density index, a parameter that is a function of both pinyon density and tree size, was positively related to the probability of stand infestation whereas diameter at the root collar and level of mistletoe infestation were positively related to the probability of individual tree infestation within stands. After the outbreak had subsided, there was no difference in the total basal area of infested and un-infested stands, however the percent basal area that was pinyon was lower in infested stands (a consequence of greater mortality in pinyons than in junipers). These results suggest that increased stocking numbers and mistletoe infestation exacerbated the effects of drought-induced water stress on pinyons resulting in greater susceptibility to beetle infestation (Ehleringer and others 1985). Pinyon Ips outbreaks may also be associated with root disease, previous defoliation, and edaphic moisture stress (Hessburg and others 1995; McCambridge 1974; Skelly and Christopherson 2003).

A great deal of information exists on the impact of the stem- and cone-boring moth (*Dioryctria albovitella*) on pinyon (*P. edulis*) individuals and populations in northern Arizona. Chronic herbivory by the moth on terminal buds affected tree architecture, producing individuals with prostrate shrub-like crowns, and reduced growth rates, shoot production, cone production, seed mass, seed viability and mycorrhizal colonization relative to lightly infested or resistant trees

(Mueller and others 2005b; Whitham and Mopper 1985). Pinyon resistance was correlated with low juvenile tree growth rates and appears to be under genetic control (Mopper and others 1991a; Ruel and Whitham 2002; Whitham and Mopper 1985). Moth herbivory also increased the chemical qualities of litter (leaf fall) produced by pinyons as well as litter decomposition rates, potentially increasing net N mineralization rates beneath the canopy (Chapman and others 2003; Scott and Binkley 1997).

At the population level, *Dioryctria* damage varied between pinyon populations, ranging from a mean of 0.1% to 22% of all stems infested, and was associated with environmental stress at local and regional levels (Cobb and others 1997; Gehring and Whitham 1994; 1995; Mopper and others 1991b). Percentage of silt-clay content of the soil was highly correlated with soil moisture and nutrient levels locally and explained 56% of the variation in herbivore damage across 23 sites in northern Arizona covering approximately 10,000 km^2 (2.5 million acres) and a wide range of soil conditions. There is no direct information on how drought conditions, superimposed on site edaphic conditions, affect moth populations, although it is likely that the increased stress on trees leads to increased infestation levels.

Detailed ecological work also exists on the effects of the pinyon needle scale insect, (*Matsucoccus acalyptus*) on pinyon at the individual (but not population) level in northern Arizona. *Matsucoccus* infests the needles of juvenile pinyons (< 50 years old), causing needle abscission within a year of initial attack and resulting in a sparse, open canopy that retains only 2 years of needles compared to up to eight years of needles by un-infested trees. Trees chronically susceptible to scale insect attack had reduced stem growth, were less affected by drought, and had decreased mycorrhizal colonization (Gehring and Whitham 1991, 1994, 1995; Gehring and others 1997; Trotter and others 2002). Scale insect herbivory also increased the temperature and soil moisture beneath canopies, the chemical quality of the leaf litter, and litter decomposition rates, together enhancing potential nutrient mineralization and cycling (Meentemeyer 1978; Aerts 1997; Kochy and Wilson 1997; Conant and others 1998; Chapman and others 2003).

Dwarf mistletoe infection of pinyons and true mistletoe infection of junipers can kill individual trees or groups of trees but typically does not cause widespread damage (Rogers 1995). Parasitism rates of between 0.2% to 46.8% have been reported in *Pinus edulis* and juniper populations in northern Arizona; rates were higher in nutrient-poor soils with low water holding capacity suggesting that environmental (moisture) stress increases the incidence of parasitism (Hreha and Weber 1979). Fire that burned the crowns of trees was the most effective factor in reducing mistletoe infection and limiting its spread within juniper and pinyon populations (Weber and others 1999). Seedling recruitment was more than adequate to replace trees dying from mistletoe infection suggesting that mistletoe-induced mortality has little effect on juniper and pinyon populations except through interactions with other pests like pinyon Ips (Hreha and Weber 1979; Negron and Wilson 2003).

Nutrient Cycling – We found no studies that documented historic nutrient cycling processes or rates. However, present-day pinyon-juniper woodlands are generally nutrient limited, and pinyon and juniper trees influence site nutrient distribution by capturing soil and nutrients from intercanopy spaces and concentrating them beneath the tree canopy, forming “fertility islands” as they age (Barth 1980; Bunderson and others 1985; Davenport and others 1996; DeBano and Klopatek 1987; Garcia-Moya and McKell 1970; Klopatek 1987a, b; Weber and others 1999). As a result, amounts of litter, nutrient levels (C, N, P, K, Mg, Fe, Cu, etc.), rates of nitrogen (N)

mineralization and numbers of nitrifying bacteria are often significantly higher and pH is lower beneath juniper and pinyon canopies than in intercanopy spaces (Barth 1980; Davenport and others 1996; Klopatek 1987a, b; Klopatek and Klopatek 1987; Padien and Lajtha 1992, but also see DeBano and Klopatek 1987, Everett and others 1986, Tiedemann 1987 for contrasting results). In addition, increased soil moisture and reduced temperatures below tree canopies create a physical and chemical environment that facilitates numerous reactions and increases the rate of nutrient cycling (Breshears and others 1998, Davenport and others 1996, Klopatek 1987a, Wilcox and Breshears 1995; Young and Evans 1987); for example, soil moisture is the dominant force driving carbon flux in pinyon-juniper woodlands (Klopatek and others 1998). The spatial pattern of nutrient resources is apparently persistent, remaining at least 5 to 8 years after tree harvest (Thran and Everett 1987); however, based on the random pattern of tree establishment following a stand-replacing disturbance (fire or tree harvest), Klopatek and others (1998) suggested that the nutrient distribution becomes homogenized at some point after a disturbance until woody plants reoccupy the site and reconcentrate nutrients under their canopies.

Fire acts as a rapid mineralizing agent, making a small part of the nutrient pool readily available for a short period of time while volatilizing substantial amounts of nutrients, thereby impacting the storage and cycling of above- and below-ground nutrients (Klopatek 1987a,b). In general, recently burned sites showed reduced available C, reduced P cycling and organic P, and higher N mineralization potential (Spier and Ross 1978; Hedley and others 1982; Klopatek 1987a).

Windthrow – We found no studies that documented windthrow as an important historical ecological determinant for the pinyon-juniper vegetation type.

Avalanche – We found no studies that documented avalanche as an important historical ecological determinant for the pinyon-juniper vegetation type.

Erosion – Most pinyon-juniper woodlands in the Southwest have a high soil erosion potential (i.e. erosion rates are more sensitive to changing vegetative cover and can cross a threshold more easily than in regions with lower intensity precipitation events). Whereas erosion is minimal on sites with high herbaceous ground cover, erosion rates may increase dramatically with reductions in litter and herbaceous cover (Davenport and others 1998, Wilcox 1994; Wood and others 1987). Soil erosion potential is positively correlated with increasing slope (Davenport and others 1998); on low to moderately sloped sites, runoff and erosion decrease (per unit area) with increasing scale, as water and sediment may be redistributed but conserved within the site (Wilcox and others 2003).

McAuliffe and others (2006) estimated an average historical soil loss of 1.9 mm (0.07 in.) per year from a hillslope in pinyon-juniper woodland in northern Arizona over the last 400 years. However, erosion was highly episodic, tending to occur after lengthy drought periods that reduced herbaceous cover, followed by extended periods of above-average precipitation. This type of erosion normally occurs when infiltration capacity is exceeded during large, early summer thunderstorms (Wilcox 1994; Wilcox and others 2003).

Several studies have measured present-day erosion rates in pinyon-juniper woodlands, highlighting the importance of herbaceous cover in minimizing precipitation runoff and soil loss in pinyon-juniper woodlands. On sites with high woody canopy cover or reduced intercanopy vegetation and litter cover, soil losses from intercanopy spaces range from 4 to 4.7 mm/year

(0.15 to 0.18 in./year) have been reported (Jacobs and others 2002; Wilcox and others 1996). Given soil depths that average 1 to 12 dm (4 to 47 in.), these erosion rates are clearly not sustainable, and annual soil losses that are more than a few millimeters may result in nutrient loss and a reduction in site productivity (DeBano 1991). Davenport and others (1998) and Hastings and others (2003) suggest a threshold ground cover of 15-20% in intercanopy spaces in pinyon-juniper woodlands below which high-magnitude sediment yields would result. However, while one site with 93% intercanopy ground cover has soil loss rates comparable to historical estimates (1 mm/year; 0.039 in./year), another site with 51% intercanopy cover has erosion rates that are already at unsustainable levels: 4.7 mm/year (0.18 in./year; Jacobs and others 2002).

In northern New Mexico and Arizona, sites with high herbaceous cover produced annual sediment yields of between 300 and 493 kg/ha (268 and 440 lb/ac), whereas grazed plots or those with low herbaceous cover produced 3.3 to 13 times as much sediment annually (Bolton and others 1992; Wilcox and others 2003). Increased erosion on grazed plots continued at Mesita del Buey, northern New Mexico, for at least 11 years after livestock were removed, at which point the authors observed decreases in runoff and other signs of recovery. Accumulated changes in soil properties from erosion may lead to a threshold that, when crossed, limits plant establishment and prevents a system from recovering without management intervention (Jacobs and Gatewood 1999; Jacobs and others 2002; Wilcox and others 1996). Sites that are still rapidly eroding 50 years after a ponderosa pine die-off initiated accelerated soil loss may have exceeded such a threshold (Wilcox and others 2003).

Synthesis – Based on published literature, climate variation, insect outbreaks, fire and seed dispersal by birds and small mammals appear to be the most important natural disturbances that determined the historical structure of pinyon-juniper stands and the distribution and abundance of these stands or patches across the landscape. Regional droughts with a 200- to 500-year return interval coupled with stress-induced insect outbreaks (pinyon Ips) caused widespread mortality of pinyons and, to a much lesser extent, juniper affecting species dominance patterns, tree age structure, tree density, and canopy cover within pinyon-juniper woodlands. These, in turn, likely caused cascading ecological effects including potentially large changes in carbon stores and carbon dynamics, soil erosion rates, seed dispersal services by birds, and changes in the composition and function of associated biotic communities. Wet periods, on the other hand, provided opportunities for tree recruitment and growth especially in fire-maintained pinyon-juniper types when these favorable climatic conditions were accompanied by fire-free periods. Surface and mixed-severity fires in pinyon-juniper savannas (and open woodlands) and in shrub woodlands removed young and/or older trees depending on time since the last fire, maintaining an open canopy structure and a diverse herbaceous and/or shrub understory depending on pinyon-juniper type. These fires burned at a mean fire interval (MFI) of 10 to 43 years in pinyon-juniper grass savanna (and open woodland) and an MFI of 23 to 81 years in shrub woodland. Shrub woodlands and persistent woodlands were also subject to severe stand-replacing fire; one study in persistent woodland estimated the turnover time for these fires at 400 years. Stand-replacing fire or some other severe stand-initiating disturbance like drought likely occurred in pinyon-juniper grass savanna (and open woodland) since few stands exceed 350-400 years in age despite the fact that trees can live more than 700 years in the Southwest (Betancourt and others 1993). Studies investigating historical disturbance processes are limited in number and in geographical and elevational extent. Clearly more studies are needed across Arizona and New Mexico and across pinyon-juniper types on historical fire regimes as well as the effect of climate variation, fire, insect outbreaks and seed dispersal by birds on the historical structure,

composition, and vegetation dynamics in pinyon-juniper stands. In addition to studies in relict sites, investigations on how current disturbance processes affect pinyon-juniper stand structure and recovery dynamics hold particular promise for reconstructing (or gaining insights into) historical disturbance processes and vegetation patterns.

12.3 Historical Range of Variation of Vegetation Composition and Structure

Patch Composition of Vegetation

Overstory, **Understory**, and **Herbaceous Layer** - We found a number of studies describing the species composition in relict areas that were never subjected to livestock grazing or other human disturbance due to their isolation and inaccessibility. We have organized these relict sites by pinyon-juniper type, basing our determination on information provided in these studies including the relative abundance of trees, shrubs and herbaceous species and tree density; in cases where determination of type was ambiguous, we have noted this uncertainty accordingly. We assume that fire and other natural disturbances in these reference sites have been operating within their historical range of variation. However, many of these sites occur on mesa tops where there are barriers to fire spread from adjacent vegetation, potentially resulting in a lower fire frequency (and associated effects on vegetation composition and structure) compared to pinyon-juniper woodlands in larger, more continuous landscapes.

<u>The following relict sites are examples of juniper savannas</u>:

Spy Mesa, sites with granular sandy loam top layer (Figure 12-2; Thatcher and Hart 1974): 16 ha (40 acre) site in northern Arizona, dominated by winter moisture, with a maximum elevation of 1525 m (5000 ft).
***Overstory*:** *J. osteosperma*
***Understory*:** The most common shrubs were *Chrysothamnus viscidiflorus*, *Purshia mexicana*, *Ephedra viridis*, and *Gutierrezia lucida*; *Opuntia sp.*, *Rhus trilobata*, and *Shepherdia rotundifolia* occurred in lower abundance.
Herbaceous layer: Perennial grasses comprised a greater proportion of the vegetation than shrubs. *Hilaria jamesii* and *Stipa speciosa* were the dominant perennial grasses on this tree/grass site (Thatcher and Hart 1974); *Aristida fendleriana*, *Bouteloua gracilis*, *B. curtipendula* and *Tridens sp*. were present in trace amounts.

Williams Mesa (Baxter 1977): 4 ha (10 acre) site in northern Arizona, dominated by winter moisture, at an elevation of 1823 m (5980 ft).
***Overstory*:** *J. osteosperma*
***Understory*:** *Nolina microcarpa*, *Quercus turbinella*, *Cercocarpus montanus*, *Purshia mexicana*, and *Atriplex canescens*.
Herbaceous layer: Perennial grasses dominated the site with cool season species comprising 60% of the herbaceous community. *Poa fendleriana* was the most abundant grass species followed by *B. curtipendula*, *Sitanion hystrix*, *B. gracilis*, and *Hilaria jamesii* in decreasing proportions (Baxter 1977).

<u>The following relict sites are examples of pinyon-juniper shrub woodlands</u>:

Fishtail Mesa (Rowlands and Brian 2001): 311-ha (770 acre) site in northern Arizona, dominated by winter moisture, between 1769-1867 m (5800-6125 ft) in elevation.
Overstory: *J. osteosperma* and *P. edulis*

Understory: Shrub cover greatly exceeds grass cover; sagebrush (*Artemisia tridentata, A. bigelovii)* dominates with *Yucca bacata, Ephedra torreyana* and *Shepherdia rotundifolia* occurring in lower abundance (Rowlands and Brian 2001).
Herbaceous layer: *Poa fendleriana* is the dominant perennial grass with *Bouteloua gracilis* in lower abundance (Rowlands and Brian 2001); total grass cover has increased to only 4% since 1958.

No Man's Land Mesa, upland sand soils (Mason and others 1967): 725 ha (1790 acre) site in southern Utah, dominated by winter moisture, between 2010-2200 m (6593-7216 ft) in elevation.
***Overstory*:** *P. edulis* dominant, *J. osteosperma* subdominant
***Understory*:** Shrubs comprised 90% of the total annual understory/herbaceous production at the site. *Artemisia tridentata*, *Quercus gambelii*, and *Opuntia sp*. dominated, with *Arctostaphylos patula, Purshia tridentate*, and *Tetradymia canescens* comprising a minor component of the shrub understory.
Herbaceous layer: *Poa fendleriana, P. nevadensis* and *Oryzopsis hymenoides* were the most common perennial grasses; *Muhlenbergia torreyi* and *Stipa comata* occurred in lower abundance.

Comanche Canyon Mesa (Ernest and others 1993): 146 ha (361 acre) site in the Carson National Forest in north-central New Mexico, equal balance of summer and winter precipitation, between 2200-2350 m (7216-7708 ft) in elevation.
***Overstory*:** *P. edulis* dominant, *J. osteosperma* subdominant, and *J. monosperma* and *J. scopulorum* present
***Understory*:** *Cercocarpus montanus* and *Artemisia tridentata* were codominant with pinyon in separate patches on and alongside the mesa; *Opuntia sp.* and *Yucca bacata* were also present. On the steep mesa slopes, *Quercus gambelii* occurred in scattered patches.
Herbaceous layer: *B. gracilis*

<u>The following relict sites are examples of persistent woodland</u>:

Southern Mesa Verde National Park (Floyd and others 2000): 6600 ha (16,310 acre) site in southwestern Colorado, dominated by winter precipitation, between 2060-2485 m (6757-8151 ft) in elevation. This site had livestock grazing prior to 1930s, but the authors argue that this has had no impact on the fire regime or on overstory structure and composition.
***Overstory*:** *P. edulis, J. osteosperma, J. scopulorum*
***Understory*:** Shrubs reported from the site include: *Quercus gambelii, Amelanchier utahensis, Symphoricarpos oreophilus, Fendlera rupicola, Rhus trilobata, Artemisia nova, Artemisia tridentata, Purshia tridentata, Cercocarpus montanus*
Herbaceous layer: *Poa fendleriana* is common, but has low cover.

Largo Mesa (Ernest and others 1993):121 ha (300 acre) site in Apache-Sitgreaves National Forest in west-central New Mexico, dominated by summer moisture, between 2350-2440 m (7708-8003 ft) in elevation.
***Overstory*:** *J. monosperma* dominant with *P. edulis* subdominant
***Understory*:** Low shrub cover; *Cercocarpus montanus, Chrysothamnus nauseosus* and *Gutierrezia sarothrae* were the most common shrubs.
Herbaceous layer: *B. gracilis* sparse.

The following relict sites may be examples of either pinyon-juniper shrub woodland or persistent woodland.

Spy Mesa, sites with vesicular, massive, or platy surface layer (Figure 12-3; Thatcher and Hart 1974): 16 ha (40 acre) site in northern Arizona, dominated by winter moisture, with a maximum elevation of 1525 m (5000 ft).
***Overstory*:** *J. osteosperma* dominant and *P. edulis* subdominant
Understory: *Chrysothamnus viscidiflorus*, *Purshia mexicana*, *Ephedra viridis*, and *Gutierrezia lucida* were the most common shrubs; *Opuntia sp.*, *Rhus trilobata*, *Euphorbia setiloba* and *Shepherdia rotundifolia* occurred in lower abundance.
Herbaceous layer: *Aristida fendleriana* and *Stipa speciosa* occurred as the most common grasses on the tree/shrub sites (Thatcher and Hart 1974); *Bouteloua. gracilis*, *B. curtipendula*, *Hilaria jamesii* and *Tridens sp*. occurred in trace amounts.

No Man's Land Mesa, upland shallow breaks (Mason and others 1967): 725 ha (1790 acre) site in southern Utah, dominated by winter moisture, between 2010-2200 m (6593-7216 ft) in elevation.
Overstory: *P. edulis* dominant, *J. osteosperma* subdominant
Understory: *Mahonia fremontii*, *Ephedra viridis*, and *Cercocarpus montanus* were the dominant understory shrubs, with *Amelanchier utahensis* and *Petradoria pumila* occurring in lower abundance. On this site, shrubs and trees comprised 85-90% of the total annual production, while perennial grasses comprised 5-10%.
Herbaceous layer: *Stipa comata* was the most common perennial grass, with *Poa fendleriana* also present.

Synthesis – Aside from the structural characteristics that are discussed in the next section, three general observations about species composition in these relict areas can be made. First, cool season perennial grasses historically predominate in the herbaceous layer where winter rainfall exceeds summer rainfall while warm season grasses predominate when summer rainfall prevails (Barnes 1983; Ernest and others 1993). Second, on Spy Mesa, in northern Arizona, a tree-shrub community (persistent woodland or shrub woodland) occurs on soils with reduced infiltration while juniper savanna occurs on sandy loam soils (Thatcher and Hart 1974). Finally, relict shrub woodlands identified here were restricted to northern Arizona and southern Utah and were dominated by big sagebrush in the understory; they likely are not representative of shrub woodlands in other portions of Arizona and New Mexico.

Figure 12-2. Photograph taken circa 1970 at relict pinyon-juniper savanna site on sandy loam soil type on Spy Mesa in northern Arizona (Photo courtesy of Thatcher and Hart 1974). This burned recently as evidenced by burned stump near center of picture.

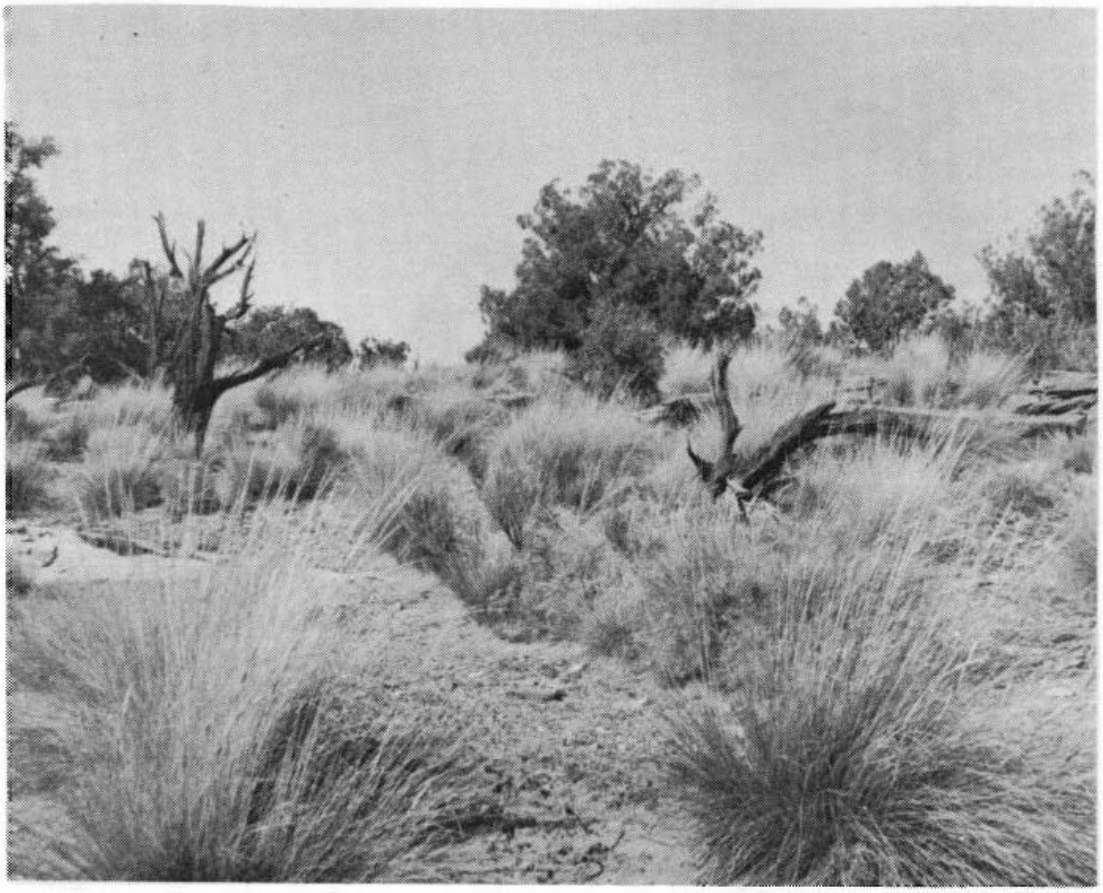

Figure 12-3. Photograph taken circa 1970 in pinyon-juniper shrub woodland relict site on vesicular, platy soil type on Spy Mesa in northern Arizona (Photo courtesy of Thatcher and Hart 1974). Vegetation in photo includes snakeweed, rabbitbrush, and juniper.

Patch or Stand Structure of Vegetation
In the following sections, reference sites and historic reconstructions using stand-age information collected in present day juniper woodlands were used to identify the historical range of variation in vegetation structure. As described earlier (see *Overstory, Understory and Herbaceous Layers* section), we attempted to assign each site to a specific pinyon-juniper type (sensu Romme and others 2003). This could not be determined if the study failed to provide information on the relative abundance or cover of trees, shrubs and herbaceous species; ambiguous cases were noted accordingly.

Canopy Cover Class (%) or Canopy Closure –Table 12-4 summarizes reported values for historical canopy cover for different pinyon-juniper woodland types based on relict sites and historical reconstructions. Reconstructed values may underestimate historical canopy cover due to post-settlement mortality of pre-settlement trees although age-specific mortality rates are not known except in cases of extreme drought (see *Climate* section).

Table 12-3. Canopy cover of trees in reference sites and in sites (Deadman Flat, Anderson Mesa) where pre-settlement canopy cover was reconstructed from stand-age information. Site codes are: DF – Deadman Flat; ANDB – Anderson mesa, basalt-derived soils; ANDS – Anderson mesa, sandstone-derived soils; ANDL – Anderson mesa, limestone-derived soils; FISH - Fishtail Mesa; MEVH – Southern Mesa Verde, high elevation; MEVL – Southern Mesa Verde, low elevation; NMSU - No Man's Land Mesa, sandy upland soils; NMSB - No Man's Land Mesa, shallow breaks soils).

Site	State	Elevation (m)	Tree Cover (%)	PJ Woodland Type	Source
DF	N AZ	1920	< 6	Savanna	Ffolliott and Gottfried (2002)
ANDB	N AZ	2073	4.5	Savanna [1]	Landis and Bailey (2005)
ANDS	N AZ	1920	11.5	Savanna [1]	Landis and Bailey (2005)
ANDL	N AZ	1920	14.7	Open woodland [1]	Landis and Bailey (2005)
NMSU	S UT	2012-2200	14	Shrub woodland	Mason and others (1967)
FISH	N AZ	1769-1867	18-20	Shrub woodland	Rowlands and Brian (2001)
NMSB	S UT	2012-2200	24	Shrub woodland or persistent woodland	Mason and others (1967)
MEVL	SW CO	1700-1848	10-65	Persistent woodland	Floyd (2003)
MEVH	SW CO	1848-2400	40-60	Persistent woodland	Floyd (2003)

[1] Landis and Bailey (2005) identified pinyon-juniper woodland type for each site on Anderson Mesa based on canopy cover values and the spatial distribution of trees; on limestone-derived soils, grass but not shrubs were reported in their woodland vegetation description, suggesting a pinyon-juniper grass open woodland.

Tree canopy cover in pinyon-juniper savanna reference sites ranged from 4% to 11.5% (Figures 12-4 and 12-5). Landis and Bailey (2005) argued that on limestone-derived soils on Anderson Mesa, historical vegetation had more of a woodland than savanna character based on the spatial distribution of pre-settlement trees, age of pinyons and junipers at the site and reconstructed canopy cover values. This then may be an example of the pinyon-juniper grass open woodland type which may extend upward in elevation to the ecotone with ponderosa pine forest. If so, then tree canopy cover in pinyon-juniper grass savanna and open woodland ranges from 4% to 14.7%.

Tree canopy cover in pinyon-juniper shrub woodland reference sites ranged from 14% to 24% (Figures 12-3 and 12-6), including the No Man's Land Mesa shallow breaks site, which alternatively may represent persistent woodland conditions. However, considering the tree density and canopy cover values reported for persistent woodlands at a similar elevation (Table 12-4 and 12-5), a tree cover value of 24% seems low for this woodland type suggesting that No Man's Land Mesa is a shrub woodland site.

Tree canopy cover in pinyon-juniper persistent woodlands ranged from 40 to 60% canopy cover in higher elevations, and 10 to 65% cover in lower elevation woodlands, which in some cases were dominated by a single species (Figures 12-7 and 12-8; Floyd 2003). All of the values for persistent woodland come from the Southern Mesa Verde site in southwestern Colorado.

Figure 12-4. Photograph taken in 1905 at El Paso and Southwestern railroad bed, southern NM of pinyon-juniper savanna. There is a large Ponderosa Pine tree in the foreground and open, scattered stands of Ponderosa Pine and pinyon, with grassy areas and patches of wavyleaf oak throughout the photographed area. (Photo courtesy of Fuchs 2002).

Figure 12-5. Photograph taken in March 2007 of pinyon-juniper savanna, east of the Gallinas Mountains in the Cibola National Forest, south of Corona, NM. Photo by Steven Yanoff.

Figure 12-6. Photograph taken by Timothy O'Sullivan in 1871 of pinyon-juniper shrub woodland near Truxton, Arizona prior to grazing. (Photo courtesy of Shaw 2006).

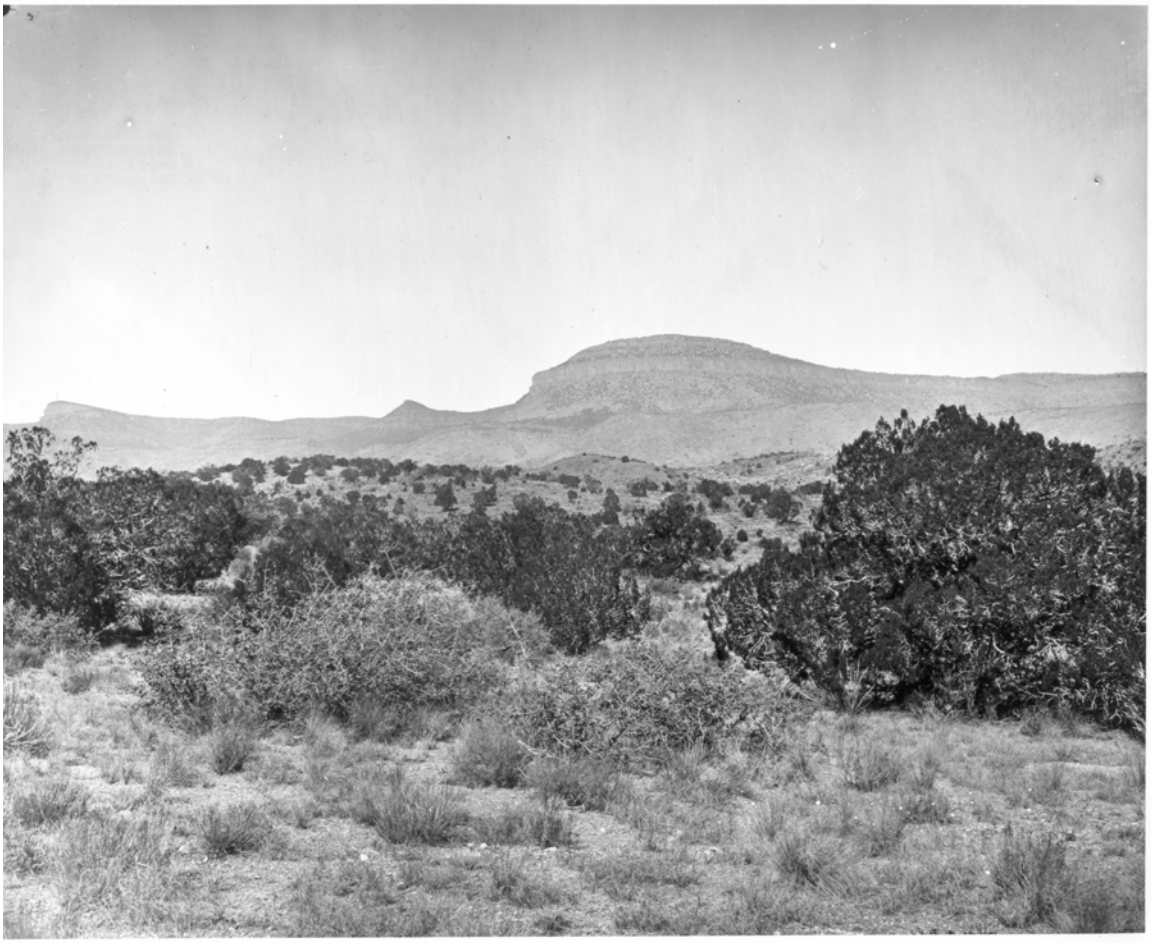

Figure 12-7. Photograph taken in 1929 of cliff dwellings in the southern portion of Mesa Verde National Park. Note the dense piñon-juniper forest on the rim above the ruins, a forest that does not look much different from the dense forests of today. (Photo courtesy of Romme and others 2003)

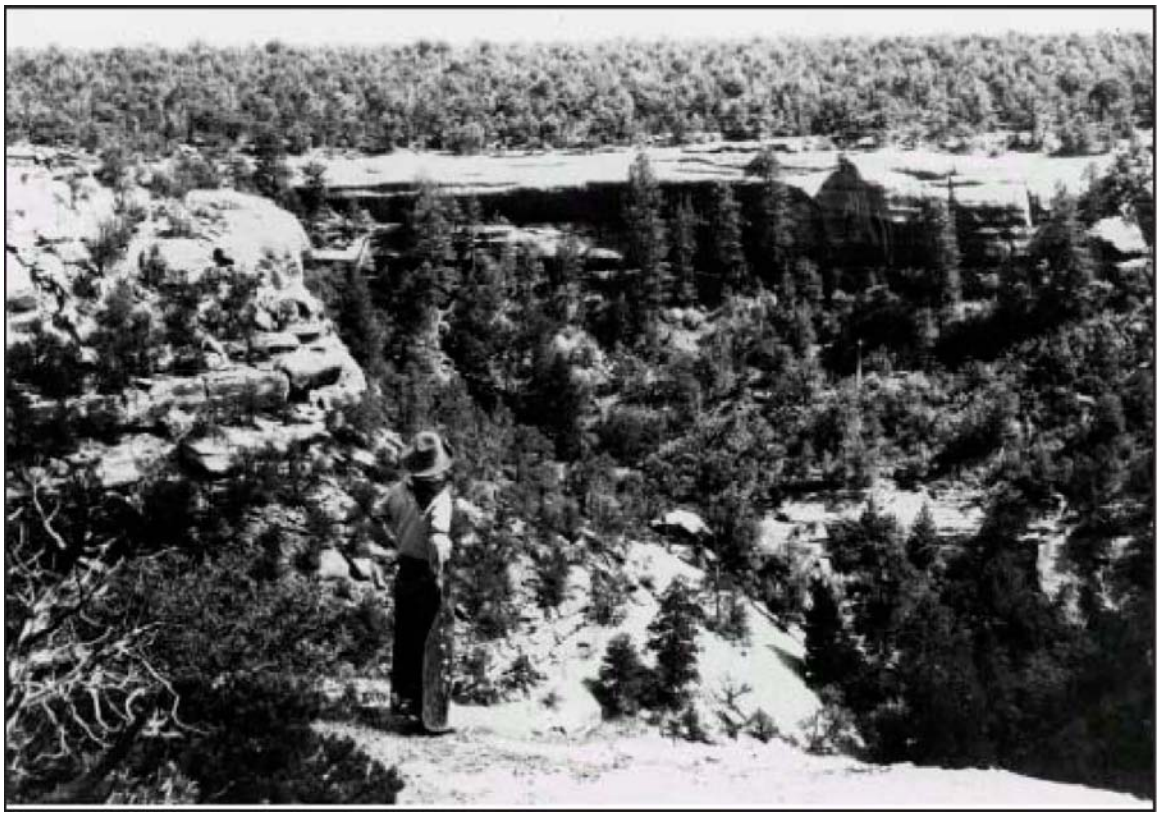

Figure 12-8. Photograph taken in 1934 of pinyon-juniper persistent woodland in the western portion of Mesa Verde National Park, near an area that burned in that year on Wetherill Mesa. The photo was taken to show the kind of forest that burned in that year. Note the high density of the stand in 1934, similar to the dense stands in this area today. (Photo courtesy of Romme and others 2003)

Structure Class (Size Class) – Tree size distributions are reported for three relict sites and are expressed in terms of number of individuals or proportion of total tree basal area in different size classes. On Fishtail Mesa in northern Arizona, pinyon and juniper trees ranged in size from <1 cm (0.4 in.) to over 40 cm (15.8 in.) in diameter (Rowlands and Brian 2001). Pinyons were more abundant than juniper in the < 15 cm (6 in.) size classes, and juniper was more abundant than pinyon in the larger size classes. The juniper size distribution was dominated by larger trees and showed pulses of recruitment such that most trees were between 1 to 5 cm (0.4 to 2 in.) and 15 and 35 cm (6 to 14 in.) in diameter (Rowlands and Brian 2001). Like that of juniper, the pinyon size class distribution was mixed size (age) but had a negative exponential shape suggesting high recent recruitment (individuals < 1 cm), high mortality rates when trees are small (young), and decreasing mortality rates as trees increase in size. Overall, pinyon displayed higher recent recruitment and a more evenly-distributed size class structure (fewer peaks and valleys) than did juniper. Ernest and others (1993) reported similar results for two pinyon-juniper woodland sites, Largo Mesa and Comanche Canyon Mesa, a persistent woodland and shrub woodland site in west-central and north-central New Mexico, respectively.

Across three sites in New Mexico–a relict shrub woodland, relict persistent woodland and historical data from a pinyon-juniper savanna—mean tree height values ranged between 3.6 m (11.8 ft.) and 4.9 m (16.1 ft.; Ernest and others 1993; Garrett and Garrett 2001; Plummer and others 1904). Mean diameter at the root collar (DRC) ranged between 12.7 cm (5 in.) and 35.6 cm (14 in.). In general, differences between sites were related to differences in their species dominance and topographic position. Mean tree height was higher and mean DRC was lower for pinyons compared to junipers. Also, the low values in the ranges came from a mesa slope site in north-central New Mexico, where trees were of smaller stature.

Despain and Mosley (1990), Garcia (1977), Huffman and others (2006a), and Martens and others (1997, 2001) reported regression equations (and R^2 values) for age-diameter relationships for pinyons-juniper stands at their sites. Diameter growth rates are dictated primarily by moisture availability; maximum diameter growth usually occurs around 50-60 years old and decreases or levels off as the tree ages (Meeuwig 1979, Ronco 1990, Howell 1941).

Life Form – We found no studies that documented the historical life form composition of pinyon-juniper woodlands.

Density – Based on information from relict sites and historical reconstructions of stand density, mean tree densities for pinyon-juniper savanna sites ranged from 22 to 122 trees/ha (8.9 to 49.4 trees/ac; Table 12-5); mean density for pinyon-juniper grass open woodland is 246 trees/ha (99.7 trees/ac); mean densities for pinyon-juniper shrub woodlands ranged from 215 trees/ha to 740 trees/ha (87 to 300 trees/ac); and mean densities for persistent woodland ranged from 948 to 3989 trees/ha (384 to 1614 trees/ac). Tree density on the proposed pinyon-juniper grass open woodland site was 246 trees/ha (100 trees/ac; Landis and Bailey 2005). These values are roughly consistent with how Dick-Peddie (1993b) described savannas (<314 trees/ha), woodlands (315-690 trees/ha) and forests (> 690 trees/ha). Values for pre-settlement tree density and canopy cover based on reconstructions in current stands may be low, due to post-settlement mortality effects. While the ranges for shrub woodland and persistent woodland are bracketed by relict sites, in which we have higher confidence, all the values for savanna except one are based on reconstructions, and therefore likely provide a conservative estimate of historical stand density.

Table 12-4. Average tree density (trees/ha) by site based on reconstructions of pre-settlement stand density or reference sites. Site codes are ANDB: Anderson Mesa basalt-derived soils, ANDS: Anderson Mesa sandstone-derived soils, ANDL: Anderson Mesa limestone-derived soils, BAJB: Bandelier National Monument, J. monosperma dominated, BPJB: Bandelier National Monument, pinyon-juniper with *B. gracilis* understory, BPJM: Bandelier National Monument, upper elevation pinyon-juniper with *M. montanus* understory, CANJ: Canjilon, Carson National Forest, COMA: Comanche Canyon Mesa, DWSA: Uncompaghre Plateau, Dominguez Wilderness Study Area, FISH: Fishtail Mesa, GGUP: Uncompaghre Plateau, Gunnison Gorge, LARG: Largo Mesa, LINC: Lincoln National Forest, MM: Mountain Meadow, MVNP: Mesa Verde National Park, NMSU: No Man's Land Mesa sandy upland soils, NMSB: No Man's Land Mesa shallow breaks soils, SMUP: Uncompaghre Plateau, Sims Mesa, TUSA: Tusayan, Kaibab National Forest, WACA: Walnut Canyon National Monument.

Site	State	Density (trees/ha)	Vegetation Type	Data Source	Source
GGUP	W CO	22	Savanna [2]	Reconstruction	Eisenhart 2004
LINC	S NM	47-62	Savanna	Historical data	Garrett and Garrett 2001
MM	SW UT	60	Savanna	Reconstruction	Cottam and Stewart 1940
SMUP	W CO	85	Savanna [2]	Reconstruction	Eisenhart 2004
PAJA	N NM	97	Savanna	Reconstruction	Martens and others 2001
ANDB	N AZ	110	Savanna [1]	Reconstruction	Landis and Bailey 2005
ANDS	N AZ	122	Savanna [1]	Reconstruction	Landis and Bailey 2005
ANDL	N AZ	246	Open Woodland [1]	Reconstruction	Landis and Bailey 2005
NMSU	S UT	215	Shrub woodland	Relict site	Mason and others 1967
WACA	N AZ	265	Shrub woodland	Reconstruction	Despain and Mosley 1990
CANJ	N NM	450	Shrub woodland [3]	Reconstruction	Huffman and others 2006a
TUSA	N AZ	621	Shrub woodland [3]	Reconstruction	Huffman and others 2006a
FISH	N AZ	740	Shrub woodland	Relict site	Rowlands and Brian 2001
NMSB	S UT	654	Shrub woodland OR persistent woodland	Relict site	Mason and others 1967
COMA	N NM	1422	Shrub woodland OR persistent woodland	Relict site	Ernest and others 1993
LARG	WC NM	948	Persistent woodland	Relict site	Ernest and others 1993
BPJM	N NM	1407	Persistent woodland	Reconstruction	Barnes 1983
MVNP	SW CO	3989	Persistent woodland	Relict site	Floyd and others 2005
DWSA	W CO	331	Unknown	Reconstruction	Eisenhart 2004
BAJB	N NM	624	Unknown	Reconstruction	Barnes 1983
BPJB	N NM	767	Unknown	Reconstruction	Barnes 1983

[1] Landis and Bailey (2005) identified pinyon-juniper woodland type for each site on Anderson Mesa based on canopy cover values and the spatial distribution of trees; on limestone-derived soils, grass but not shrubs were reported in their woodland vegetation description, suggesting a pinyon-juniper grass open woodland.
[2] Eisenhart identified savanna stands on the Uncompaghre Plateau based on pre-settlement tree density.
[3] Mean tree density (for pinyon, juniper) was calculated on a site-wide basis (i.e., includes stands dominated by ponderosa pine) and, therefore, may be a conservative estimate of pre-settlement tree density in pinyon-juniper stands.
[4] Mean tree density for 8 stands.

Age Structure – In addition to information cited in the **Structure Class** section, two studies have described pinyon-juniper stand age structure in relict sites. On Fishtail Mesa, a relict shrub woodland in northern Arizona, pinyon stands had a mixed-age structure on all sites sampled (Rowlands and Brian 2001), with an average of 12.5% pinyons in the stand identified as pre-settlement (>150 years old). In two undisturbed persistent woodland stands on Chapin and Wetherill Mesas in southwestern Colorado, Floyd and others (2000) found a negative exponential age distribution for pinyons and junipers suggesting constant recruitment and mortality rates and a more or less stable age class structure.

Several studies have attempted to reconstruct historic stand age structure and the timing of establishment events for pinyon-juniper woodlands in non-relict sites by looking at the age-distribution of pre-settlement trees. Problems with this approach include the possibility that i) pre-settlement trees may have died over the last 150 years especially younger trees skewing the distribution toward older age classes, and ii) inner rings may be lost due to decay resulting in uncertain establishment dates and potential gaps in the pre-settlement age structure (Johnson and others 1994; Landis and Bailey 2005). Nonetheless, on Anderson Mesa, Landis and Bailey (2005) found juniper-dominated savannas on sandstone- and basalt-derived soils had a multi-aged stand structure prior to 1860 with few gaps in establishment over multiple centuries. Eisenhart (2004) and Floyd (2003) reported similar results for pinyon and juniper in most savanna and persistent woodland stands in western and southwestern Colorado. The rate of establishment prior to 1880 ranged from 5 to 20 trees per hectare per decade in pinyon-juniper sites in northern Arizona, northern New Mexico, and southwestern Colorado (Eisenhart 2004; Julius 1999; Landis and Bailey 2005). Pulses of establishment were noted in all pre-settlement stand age reconstructions. On limestone-derived soils at Anderson Mesa, juniper establishment and survival increased starting in 1690, with other establishment pulses occurring between 1730-1750 and 1790-1810, such that by 1860, most trees in the stand were less than 150 years old. Establishment pulses over these same periods were also noted for pinyons at shrub woodland and persistent woodland sites in southwestern Colorado, northern Arizona and northern New Mexico (Floyd 2003; Huffman and others 2006a). In another study in southwestern Colorado, old-growth stands of undetermined woodland type showed one large pre-settlement pulse occurring between 1790 and 1800 (Eisenhart 2004). Similar to the limestone-derived soil type on Anderson Mesa, the preponderance of 50-150 year old trees was also recorded in Floyd (2003)'s and Huffman and others (2006a)'s studies.

Patch Dispersion – Landis and Bailey (2005) reconstructed the historic pattern of tree dispersion for pinyon-juniper woodlands on Anderson Mesa. On basalt- and sandstone-derived soil, juniper trees showed a highly clumped distribution at all scales in 1860, a pattern that was still discernable today. On limestone-derived soils, trees were also clumped at smaller spatial scales but the clumps or patches were randomly distributed across the landscape at larger spatial scales. The mean size of tree patches in 1860 was greater on basalt- and sandstone-derived soils than on limestone-derived soils. The pattern of tree dispersion on the former soils allowed for a more extensive and contiguous grass community than on limestone-derived soils (Landis and Bailey 2005).

Reference Sites Used

Reference sites, historical data, and non-relict sites where pre-settlement conditions were reconstructed are listed in Tables 12-4 and 12-5.

Limitations – As discussed earlier, most relict sites are small mesas that lack vegetation continuity with surrounding areas and therefore probably experienced lower historic fire frequencies compared to woodlands occurring in larger, more contiguous landscapes. This suggests that inferences about historic vegetation composition from relict sites should be applied cautiously to other sites. In addition, reconstructions of historical vegetation structure (and composition) in non-relict sites using pre-settlement trees have not attempted to account for post-settlement tree mortality, resulting in conservative estimates of historical tree density and biases in the age/size distribution if mortality is age/size dependent. Finally, the number and geographic range of relict and pre-settlement reconstruction sites from which to draw information is restricted, considering the extensive distribution of this vegetation type in Region 3. In particular, information regarding composition of shrub and persistent woodlands is solely available from studies in the Colorado plateau region, and canopy cover and density values for all types also come almost exclusively from studies in northern Arizona and northern New Mexico.

*Characteristics of Applicable Site**s*** – Ideally, reference sites would have intact historical disturbance processes including natural fire regimes and a lack of human disturbance (e.g. fuelwood cutting, livestock grazing, mining, and fire suppression). All of the relict sites were chosen based on their isolation from human disturbance.

Synthesis - Historical data, pre-settlement reconstructions, and studies conducted at relict sites together provide information on historical vegetation structure and composition in the three pinyon-juniper types. In general, the three woodland types differed with respect to understory composition and overstory structure such that the ranges of canopy cover and tree density values for each of the three types were exclusive and non-overlapping. Historically, juniper size distributions were discontinuous with greater numbers of trees in certain size classes and fewer trees in others (i.e., peaks and troughs), while pinyons showed a more even size distribution; stands were generally dominated numerically by smaller pinyon trees although among the larger size classes, junipers normally dominated. Recruitment by pinyons and junipers was relatively continuous over hundreds of years punctuated by establishment peaks presumably due to favorable climate conditions for recruitment (or survivorship). This recruitment pattern gave rise to mixed age stands across all pinyon-juniper types. Unfortunately, the information for historical savannas comes from a restricted number of low elevation sites, while information for shrub woodlands and persistent woodlands comes from a limited number of sites in a restricted geography (i.e., northern Arizona, northern New Mexico, southern Utah and southwestern Colorado); in the case of shrub woodlands, all had a sagebrush understory. More studies over a broader geographic area (and elevational range) are needed to fully describe the historical range of variation in composition and structure for shrub and persistent woodlands (and grass savannas).

12.4 Anthropogenic Disturbance Processes (or Disturbance Exclusion)

Herbivory – Domestic livestock were introduced into New Mexico when the Spanish settled the Rio Grande Valley in 1598 (Allen 2001; Baisan and Swetnam 1997; Gottfried and others 1995; Springfield 1976). Population expansion by Spanish settlers resulted in increasing numbers of sheep, goats, and cattle and intensive use of an expanding area for pasture. Adoption of sheep-herding by the Pueblo and Navajo people spread grazing impacts beyond the reach of Spanish settlements. By 1858, thousands of sheep and goats were observed grazing on the Rio Puerco

and elsewhere in northern New Mexico (Cozzens 1875; Denevan 1967; Dick-Peddie 1993a; Standley 1915). Grazing impacts varied *spatially* depending on the proximity to human settlement and water and *temporally* depending on ongoing political events such as the Pueblo revolt in 1680, Spanish re-colonization and expansion (1681-1784), and the ebb and flow of relationships between Spanish settlers and their Navajo, Apache, and Ute neighbors between 1600 and 1860. This spatial and temporal variation is reflected in local fire histories, with some sites showing a reduced frequency of surface fires and longer fire-free intervals during periods of expanded livestock use, and other, more isolated sites showing no evidence of grazing effects on the fire regime over a 300+ year (Allen 1989; Baisan and Swetnam 1997; Brown and others 2001; Muldavin and others 2003; Swetnam and Baisan 1996).

In Arizona the history of human impact is somewhat different. Spanish settlement and the introduction of livestock occurred in the 1690s, however marauding Apache prevented the expansion of livestock numbers and pasture use in central and southern Arizona, while in northern Arizona, the lack of surface water and conflict with the Navajo and Ute limited the reach and size of livestock herds until the 1880s (Bahre 1991).

In the 1870s and 1880s, the subjugation of Native Americans, the development of windmill technology to pump groundwater, and the completion of the railroads linking the Southwest to outside commercial markets created a ranching boom, resulting in millions of sheep and cattle grazing on public lands in Arizona and New Mexico (Allen 2001; Bahre 1991; Dick-Peddie 1993a). Livestock grazing continued in most places until the 1970s or 1980s, reducing herbaceous cover and essentially eliminating fire as a disturbance agent (Gottfried and others 1995; Allen 2001). By 1885, newspaper accounts of extensive fires in grassland and pinyon-juniper-oak woodlands had already ceased in southeastern Arizona (Bahre 1991) and fire-history reconstructions in pinyon-juniper savannas and shrub woodlands show little evidence of surface or mixed-severity fire after 1900 (Allen 1989; Baisan and Swetnam 1997; Brown and others 2001; Despain and Mosley 1990; Huffman and others 2006a).

The decline in herbaceous cover due to livestock grazing in concert with the disruption of the fire regime has been cited in a number of studies as the primary causative factor for the observed increase in tree densities in many pinyon-juniper systems over the last 120 years (Arnold and others 1964; Blackburn and Tueller 1970; Burkhardt and Tisdale 1976; Cottam and Stewart 1940; Dwyer and Pieper 1967; Gottfried 1987; Gottfried and others 1995; Huffman and others 2006a; Huffman and others 2006b; Johnsen 1962; Tausch and others 1981; West and others 1975). According to this argument, the increasing number of trees that survived in the absence of fire out-competed forbs and perennial grasses for limited soil moisture creating a “positive feedback loop”, e.g. coalescence of eroded patches, that facilitated continued tree invasion and loss of herbaceous ground cover (Allen 2001; Breshears and others 1997; Gottfried and others 1995; Jameson 1967). Subtle shifts in herbaceous composition also occurred with grazing, with warm-season perennial grasses being favored at the expense of cool-season species (Arnold and others 1964; Baxter 1977; Jameson and others 1962). The decline in herbaceous cover along with the direct effects of livestock grazing on soil compaction have also led to increased soil erosion rates at many sites (Allen 2001; Baxter 1977; Bogan and others 1998; Carrara and Carroll 1979; Gottfried and others 1995; Huffman and others 2006b; Wilcox and others 1996; Wood and others 1987). When erosion is severe, changes in the physical properties of soils and in the distribution and abundance of limiting nutrients can impede the re-establishment of herbaceous vegetation even when livestock grazing is eliminated (Allen 2001; Breshears and

Barnes 1999; Davenport and others 1998; Gottfried and others 1995; Klopatek 1987b; Klopatek and Klopatek 1987; Klopatek and others 1990; Koniak and Everett 1982; Laycock 1991). Alternative hypotheses implicating livestock grazing (without fire regime disruption), drought and climate variation in the post-settlement loss of grass cover, tree increases, and increased soil erosion have also been proposed. Additional studies are needed to disentangle the effects of these factors on the historical fire regime and on post-settlement structural changes in the pinyon-juniper types.

Perhaps the best-documented example of vegetation change following the introduction of livestock (and subsequent drought) is Mountain Meadow in southwestern Utah where the Mormons introduced livestock in 1862 (Cottam and Stewart 1940). At the time of settlement there was a wet meadow in the valley bottom, open grassland on meadow edges and on hillslopes and juniper woodland on the ridgetops. According to oral accounts, severe overgrazing caused a marked deterioration in range conditions that was further exacerbated by a drought lasting a decade in 1870s and early 1880s. The spring of 1884 brought heavy rains, which ran off the denuded hillslopes, causing down-cutting in the valley bottom and subsequent draining of the wet meadow. Over the next 55 years, the foothills were converted to sagebrush steppe and juniper woodland expanded into the former grassland from several foci. Invasion by junipers, followed by pinyon, began in the early 1900s when precipitation was above-average. In foothill areas, juniper recruitment exceeded that of sagebrush after 30 to 40 years in foothill areas, and by 1940, 55% of the sagebrush plants in the foothills were dead presumably due to competition with juniper (Cottam and Stewart 1940). Between 1862 and 1934 the extent of juniper woodland increased by 580% from 436 to 2,538 hectares (1,078 to 6,272 acres) and tree density within the original pinyon juniper stand increased six-fold.

A number of studies have investigated the effect of livestock exclusion for 10 to 28 years at pinyon-juniper sites across Arizona and New Mexico. Canopy cover of pinyon and juniper increased to the same extent or more on grazed plots than on exclosure plots (Arnold and others 1964; Potter and Krenetsky 1967) while shrubs, especially palatable species like sagebrush and cliffrose, and perennial grasses, especially cool-season species, increased on exclosure plots; herbaceous forage production was up to 2 times greater on excluded plots than on grazed ones (Springfield 1976; Arnold and others 1964; Pieper 1968; Potter and Krenetsky 1967). On one catchment studied in northern New Mexico, after 15 years of grazing exclusion, the erosion rates have persisted at rates 50 to 100 times higher than stable pinyon-juniper woodlands, and the intercanopy areas remain bare and unvegetated (Wilcox and others 1996).

Silviculture – Historically, pinyon-juniper woodlands were an important source of fuelwood, fenceposts, and building materials, and the current demand for these products has continued to increase (Ronco 1990; Dick-Peddie 1993a; Gottfried 1987; Gottfried and Severson 1993; Ffolliot and others 1979). A number of authors have suggested that the increased tree densities in pinyon-juniper systems since the early 1900s represent recovery from historical and prehistoric fuelwood cutting although this hypothesis has not been rigorously tested (Hack 1945; Lanner 1975; Betancourt 1987; Betancourt and van Devender 1981; Betancourt and others 1993; West 1984; Bahre 1991). However, in the Chaco Canyon area, the disappearance of pinyon from middens after 980 AD is consistent with a simulation model of long-term fuelwood harvest by Native Americans which shows woodland depletion within 200 years assuming 10th through 12th century population estimates (Samuels and Betancourt 1982; Hall 1988). In southern Arizona, fuelwood from evergreen woodlands (including pinyon-juniper woodlands) was the major source

of fuel for mining operations until the late 19th century and for domestic heating and cooking until after the 1940s (Bahre 1991). For example, fuelwood was so scarce near Tucson in 1892 that woodcutters had to go 20-30 miles away and still only brought back roots and stumps (Bahre 1991). In addition, local newspapers reported that the hills around the mining town of Bisbee were stripped within miles of town and that cordwood had become scarce throughout southeastern Arizona (cited in Bahre 1991). Bahre and Hutchinson (1985) estimated fuelwood use for mining and domestic heating/cooking in the Tombstone woodshed between 1878 and 1940 and found that the estimated amount of cordwood consumed was more than the total cordage currently reported by the Forest Service there. Although fuelwood was replaced by coal in the early 1900s as an energy source for mining, woodcutting for domestic heating and cooking continued. Even as late as 1940, 44% of the homes in Arizona still depended on fuelwood for heating and cooking (U.S. Bureau of Census 1943). Thus, fuelwood cutting for mining and domestic use prior to 1940 had significant impacts on pinyon-juniper woodlands near mining towns and population centers in southern Arizona and presumably in the rest of Arizona and New Mexico (Bahre 1991; Dick-Peddie 1993a). As fuelwood cutting was primarily a local phenomenon, however, recovery from silvicultural activities alone probably does not explain extensive regional expansion and "infill" of pinyon and juniper trees.

Current figures for the amount of wood harvested from pinyon-juniper woodlands in Arizona and New Mexico are lacking. However, in 1986 approximately 227,000 m^3 (8,018,370 ft^3) of pinyon and juniper fuelwood were harvested in New Mexico (McLain 1989) and up to 20,000 cords of wood (156 ft^3) were harvested annually on the Gila National Forest until 1985 (Fowler and others 1985). Most tribal and rural communities in Arizona and New Mexico depend on fuelwood as the primary source for heating and cooking and as a way to generate income, and there is a growing concern by National Forests that fuelwood demand will exceed supply in less than 50 years (Gottfried and others 1995). In response to this demand, National Forest districts have developed their own fuelwood policies in the absence of volume yield tables and sustained yield estimates (Cary 1980). Moreover, studies by the Forest Products Laboratory in Madison, Wisconsin, and others have demonstrated the potential for the development of new products made from wood and fiber of pinyon and juniper, which may greatly increase the future demand for trees (Murphy 1987; Ffolliott and others 1999; Gottfried 2004).

Roughly 88% of the juniper and pinyon-juniper woodlands in the Southwest have been identified as having the potential for growing wood products on a sustainable basis (Connor and others 1990; Van Hooser and others 1993). Bassett (1987) reviewed different silvicultural prescriptions that could be applied to juniper-woodland and discussed the tradeoffs in each, concluding that single-tree selection and two-step shelterwood methods would best sustain productivity. Other prescriptions such as three-step shelterwood, group selection and clearcutting are also being used in the Southwest (Gottfried and others 1995; Gottfried 2004). Demographic and growth information suitable for developing predictive models of sustained yield in these woodlands under different silvicultural practices is limited (Samuels and Betancourt 1982; Dixon 2006), although regressions between tree size (diameter, height) and volume are available for junipers and pinyons regionally (Chojnacky 1988, 1994; Clendenen 1979; Connor and others 1990). The Rocky Mountain Research Station is evaluating two ongoing silvicultural case studies in Arizona and New Mexico to provide additional information (Gottfried 2004).

Fragmentation – Construction of roads and a proliferation of primary and secondary homes have occurred in many pinyon-juniper systems throughout Arizona and New Mexico, although

we could find no studies that documented the impacts of fragmentation on pinyon-juniper savannas, shrub woodlands and persistent woodlands. However, in other vegetation types, fragmentation has been shown to affect fire regimes, fire control options, spread of non-native plants, wildlife movements and abundance, and local hydrologic cycle (Beier and Noss 1998; Brothers and Apingarn 1992; Curtin and others 2002; Forman 2003; Gelbard and Belnap 2003; Holdsworth 1997; Theobald 2003; With 2002).

Mining – Fuelwood that was harvested from southwestern woodlands and forests, including pinyon-juniper systems, was used in virtually every step in mining process from the 1870s until shortly after the turn of the century; consumption peaked during the 1890s. A discussion of mining impacts can be found in *Silviculture*, above.

Fire Management – The disruption of historical fire regimes that followed the introduction of livestock (and the 1890's drought) has been documented in historical accounts and fire-history reconstructions in pinyon-juniper savanna and open woodland (upper and lower ecotones) and in shrub woodland at the ecotone with ponderosa pine forest (Leopold 1924; Bahre 1985; Bahre 1991; Allen 1989, Despain and Mosley 1990; Kaib and others 1996; Swetnam and Baisan 1996; Baisan and Swetnam 1997; Brown and others 2001; Muldavin and others 2003; Huffman and others 2006a). These reconstructions show the virtual cessation of surface and mixed-severity fires in pinyon-juniper systems between 1890 and 1905. In contrast, there is little evidence that fire regimes in persistent pinyon-juniper woodland have been significantly altered by introduced livestock, and the alteration may be minimal or greater for shrub woodland and grass open woodland in non-ecotone settings, although fire-history information for the latter types/settings are lacking in the Southwest (Tausch and West 1988; Floyd and others 2000, 2004; Romme and others 2003).

In the early 1900s, a policy of active fire suppression was instituted by the federal government and involved the construction of fire lines and roads and later coordinated efforts with fire brigades and air tankers (Swetnam and Baisan 1996). Fire exclusion was very successful initially, but the accumulation of fuels, increased tree densities, and development of fuel "ladders" that could bring surface fires into the crowns and canopies of the woodland made fire suppression more difficult. As the number and size of fires has increased over the last century especially in the last 20-30 years (Dahms and Geils 1997; Crimmins and Comrie 2004; Westerling and others 2006), emphasis on the use of prescribed fire has increased within land management agencies, with varying levels of success due to complex social and climatic factors. Large stand-replacing fires that have burned with surprisingly high-intensity in pinyon-juniper and higher-elevation conifer systems in recent years [such as the La Mesa fire (1977), the South Canyon Fire (1994), the Cerro Grande fire (2000), the Rodeo-Chediski fire (2002), the Aspen fire (2003), and the El Rito fire (2005)] have underscored the need for active fire management to reduce fire risk as well as to restore the functionality of fire-adapted ecosystems (Frost 1998; Brown 2000; Hardy and others 2000). Restoration treatments, including fire management, should be based on an understanding of local stand history and the historical range of variability in disturbance regimes (Romme and others 2003; Baker and Shinneman 2004).

A number of studies conducted after prescribed burns and wildfires provide information on the post-fire survivorship of pinyon and juniper and are germane to the question of whether low-severity surface fire can maintain a savanna structure or low tree densities in woodlands. There are two studies that investigated the effects of low-severity surface fires in pinyon-juniper

savannas (Table 12-6), and two other studies that reported the effects of prescribed and natural fires in grasslands invaded by low to moderate numbers of pinyon and juniper. In all but one of these studies, fires killed a high percentage of small trees (Table 12-6). The exception, Alderete (1996), reported low mortality of junipers (3.6%) following a prescribed burn in grassland with moderate pinyon-juniper invasion; however, burning conditions were mild which may account for the low mortality rate. Miller and Tausch (2001) suggested that a fire every 45 to 90 years would be sufficient to maintain a savanna or open-canopy (low tree-density) woodland structure, based on the time it takes for a tree to reach a height of > 3 m (9.8 ft.) when the survival rate is high (Table 12-6). This frequency range is consistent with MFI estimates derived from fire history reconstructions in pinyon-juniper savanna (and open woodland) and some shrub woodland sites in the upper ecotone (Table 12-2). In sites where alligator juniper, a resprouting species, occurs more frequent fires would be required to suppress small trees and maintain a savanna structure (Miller (1999)

Table 12-5. Summary of the effects of post-settlement (1880) prescribed burns and wildfires on pinyon-juniper mortality as a function of pinyon-juniper setting; table from Baker and Shinneman (2004) including new information. Abbreviations are: pj = pinyon-juniper

Setting/Study	State	Elevation (m)	No. of prescribed fires	No. of wildfires	Trees/hectare Before	Trees/hectare After	Percent mortaility, other observations
Grassland with low pj							
Arnold and others (1964); Jameson (1962)	N AZ	1800	2	1	ca. 150		
January and March (prescribed) June (wildfire)							70-100% of trees < 1.2 m tall
January and March (prescribed) June (wildfire)							30-40% of trees 1.5 – 1.8 m tall
March (prescribed); June (wildfire)							60-90% of trees > 2.1 m tall a.
Grassland with moderate pj							
Alderete (1996)	S NM	1940	1				
					67	1	98.5% of all trees
Juniperus deppeana, J. monosperma					2427	2340	3.6% of all trees
Pinus edulis							
Pinyon-juniper savanna							
Dwyer and Pieper (1967)	S NM	1830-1980		1			100% of trees < 1.2 m tall
Juniperus monosperma (70%)							24.2% of all trees
Pinus edulis (30%)							13.5% of all trees
Johnson and others (1962)	S AZ	1525		1			
Juniperus deppeana							32% of 1-3 in. diameter trees
Juniperus deppeana							23% of 4-6 in. diameter trees
Juniperus deppeana							22% of 7-9 in. diameter trees
Juniperus deppeana							28% of all trees
Juniperus monosperma							79% of 1-3 in. diameter trees
Juniperus monosperma							73% of 4-6 in. diameter trees
Juniperus monosperma							77% of 7-9 in. diameter trees
Juniperus monosperma							76% of all trees
Closed pj woodland							
Alderete (1996)	S NM	1940	1				
					460	227	50.7% of all trees; notes poor fire spread
Juniperus deppeana, J. monosperma					1080	833	22.9% of all trees; notes poor fire spread
Pinus edulis							
Arnold and others (1964); Tress and Klopatek (1987) Pinus edulis, Juniperus osteosperma	N AZ	--		14 , 3	1025	0	100% of all trees except unburned islands where soils were rocky
Despain (1987) Pinus edulis, Juniperus osteosperma	NW AZ	--		9			ca. 22,000 acres burned; photos show 100% tree mortality
Despain and Mosley (1990)	N AZ			1			Survivorship figures unavailable, but fire left surviving overstory trees.

McColloch (1969) Pinus edulis, J. deppeanna, combined	N AZ	ca. 1500	Unknown	1	635	0	100% tree mortality
Aro (1971) Pinus edulis, J. osteosperma, J. monosperma	CO, UT		1	2			High, photos show 100% or close to 100% tree mortality
Barney and Frischknecht (1974) Pinus monophylla, Juniperus osteosperma	C UT	1930-2600		23	793	0	100% mortality of juniper, pinyon c.
Erskine and Goodrich (1999) Pinus edulis, Juniperus osteosperma	NE UT	--	7				Nearly 100% tree mortality on ca. 3000 acres
Goodrich and Barber (1999) Pinus edulis, Juniperus osteosperma	NE UT	--		3			Nearly 100% tree mortality
Floyd and others (2004); Romme and others (2003) Pinus edulis, Juniperus osteosperma	SW CO	2060-2485		8			100% tree mortality
Huffman and others (2006a) Pinus edulis, J. osteosperma, J. scopulorum	N AZ, N NM	2005-2438	1	22			Post-settlement wildfires scarred trees (surviving) in fire history sample; pinyon-juniper patches in prescribed burn area generally had 500-1000 trees/ha post-burn.

a. Mortality values for larger trees are the result of accumulated Russian thistle carrying fire into the tree crowns.
b. Burn conducted in mid-April following snowstorm; grassland portion of burn had only 750 kg/h of fine fuels to carry fire
c. Aerial photograph shows patchy nature of burns with extensive areas of closed-p-j woodland adjacent to burned areas.

In contrast to pinyon-juniper savannas, prescribed fires and wildfires in closed woodlands after 1880 have been stand-replacing in 71 out of 96 cases (74%) and one of these burns was reportedly conducted under cool, moist conditions (Table 12-6; Alderete 1996). In general, however, prescribed burns in these settings were intentionally ignited under extreme burning conditions to ensure fire spread with light surface fuels (Aro 1971; Despain 1987). Six studies at four sites documented the behavior of both pre- and post-settlement fires in closed woodlands and fires were high intensity and stand-replacing before and after Euro/American settlement in all cases (Table 12-3; Arnold and others 1964; Barney and Frischknecht 1974; Goodrich and Barber 1999; Erdman 1970; Floyd and others 2000; Romme and others 2003). Thus, fires appear to kill young trees when the understory is composed primarily of perennial grasses and herbaceous fuels and can maintain a savanna or open-woodland structure in these settings, however in dense pinyon-juniper woodlands (i.e., persistent woodlands as well as savannas and shrub woodlands with dense tree encroachment), current fires tend to be stand-replacing.

Exotic Introductions (Plant & Animal) – We could find no information on the distribution of exotic animals and their effects on pinyon-juniper systems, and we found limited information on non-native plants, specifically relating to their colonization of sites following fire. After recent fires in Mesa Verde, southwestern Colorado, invasive non-native species including cheat grass (*Bromus tectorum*), musk thistle (*Carduus nutans*), and creeping thistle (*Cirsium arvense*) dominate the understory (Floyd and others 2006; Romme and others 2003). Although cheatgrass is not yet abundant in recently burned sites or at Mesa Verde in general, the species has caused a dramatic change in the fire regime in the northern Great Basin, increasing both fire frequency and size there (Barber and Josephson 1987; Miller and Tausch 2001). In some situations, post-fire seeding efforts have used non-native grass species, facilitating the subsequent spread of certain species (Keeley 2004; Robichaud and others 2000). In addition, burned slash piles following fuelwood harvest became foci for the establishment of exotic plant species (Dickinson and Kirkpatrick 1987; Haskins and Gehring 2004). A variety of post-fire treatments, including aerial seeding of native grasses, mechanical removal, and herbicide application, have been effective in reducing weed densities but not in preventing weeds from becoming "major components of the post-fire plant community" (Floyd and others 2006; Romme and others 2003). In northern Arizona, accumulation of Russian thistle (*Salsola iberica*) beneath the canopy of large trees increased mortality during wildfires (up to 90%) reversing the advantage that size normally confers on survivorship (Arnold and others 1964).

Other Treatments – Other treatments besides prescribed fire have been applied to reduce the density of trees in pinyon-juniper systems with the goal of reversing tree encroachment, increasing forage production for livestock and wildlife, satisfying the demand for fuel wood production and improving watershed condition (Arnold and others 1964; Dalen and Snyder 1987; Gottfried and others 1995). The treatments have included cabling or chaining; bulldozing; clearing or thinning with hand axes or motorized saws; and herbicide application. Current estimates of the number of acres treated are lacking, although by 1961, 486,000 ha (1,201,000 ac) of pinyon-juniper woodland had been chained, bulldozed, thinned or cleared in Arizona (Cotner 1963); these treatments were also applied in New Mexico (Albert and others 2004; Gottfried and others 1995; Rippel and others 1983). The following narrative discusses the available literature for each of these treatments, focusing on their relative effectiveness in reducing tree density and achieving management objectives (e.g., increasing forage production).

Cabling or chaining has been identified as the most cost effective way to reduce the numbers of junipers and pinyon (Arnold and others 1964). However, this view is not universally held, as the remaining trees and the release from competition following treatment may result in even greater juniper-pinyon numbers later (Aro 1971; Rippel and others 1983); soil disturbance may temporarily reduce herbaceous ground cover and increase the spread of non-native species (Sheley and others 1999; Sieg and others 2003). Typically, the cable/chain slips over smaller trees (< 3.3 meters high) resulting in a kill rate of 30- 94% (Arnold and others 1964; Aro 1971; Cotner 1963; Jameson and Reid 1965). In general, kill rates vary with stand composition, age structure, and tree density (Steven 1999; Vallentine 1989). Doubling-chaining is more effective than single-chaining, and chaining followed by windrowing of trees increases the kill rate to near 100% (Aro 1971). Chaining has occurred on slopes of up to 65%, although it is normally applied on slopes < 50%. After chaining, as much as 50% of the ground may be covered with slash and debris; large trunks remain un-decomposed for decades, but smaller limbs and branches increase soil moisture and provide protected sites for grass establishment (Brockway and others 2002; Jacobs and others 2002). Broadcast burning has been applied after cabling to remove young trees, slash and debris (Arnold and others 1964), resulting in volatilization losses of N that could approach 13 percent of the total ecosystem N (Tiedemann 1987).

Other mechanical methods have also been used. Bulldozing has been most effectively applied to uproot individual trees in open stands that were too sparse to chain (Arnold and others 1964). Depending on the site, mortality can be close to 100% or many small trees may survive (Hessing and Johnson 1982; Ludwig and others 1997). Clearing or thinning with hand axes or motorized saws has proven more thorough than using heavy equipment, resulting in almost complete kill rates with minimal soil disturbance (Arnold and others 1964); this treatment has been best applied to grasslands being invaded by small trees. Although bulldozing and hand-thinning and clearing may be more effective than chaining in reducing pinyon-juniper densities, the per-acre costs are significantly higher (Arnold and others 1964).

Herbicide treatments, using tebuthiuron and picloram pellets, have been applied aerially or manually to pinyon-juniper woodlands to reduce tree density but have not been widely used (McDaniel and WhiteTrifaro 1987). Tree mortality rates ranged from 24% to 100% and varied by tree species, herbicide type, concentration of the active ingredient, stand structure, and soil type (Brock 1985; Clary and others 1985; Duncan and Scifres 1983; Johnsen 1987; Johnsen and Dalen 1984, 1990; McDaniel and WhiteTrifaro 1987; Wittie and McDaniel 1990). In general, kill rates were higher in pinyon than in juniper species and higher among smaller individuals than among larger ones. Response of understory herbaceous species also varied with herbicide type and application rate, although, on average, grass production increased by 1.4 to 2.3 times on treated areas within 2 to 3 years of treatment (Johnsen and Dalen 1990; Wittie and McDaniel 1990). Immediately after treatment, however, some grasses were killed under and near trees, especially at higher application rates, while forb production was reduced for at least 2 to 3 years. Site and soil characteristics should be carefully evaluated prior to herbicide use to determine the potential for increased herbaceous production (Romme and others 2003).

A number of studies have documented the response of vegetation, including changes in forage production, to mechanical treatment over time periods ranging from 1 to 29 years. In general, perennial grass cover increased in absolute terms by 4% to 31% following treatment, and forage production increased by 20 kg/ha (18 lb/ac) to 600 kg/ha (535 lb/ac) compared to adjacent untreated areas (Albert and others 2004; Arnold and others 1964; Aro 1971; Brockway and

others 2002; Clary and Jameson 1981; Jacobs and others 2002; Rippel and others 1983; Schott and Pieper 1987; Springfield 1976;). These changes persisted anywhere from 8 to 13 years after treatment, but by 20 to 28 years, perennial grass cover and production had returned to pre-treatment levels or lower in response to increasing tree canopy cover (Arnold and others 1964; Rippel and others 1983; Schott and Pieper 1987). In general, the degree to which forage production was enhanced depended on annual precipitation, pre- and post-treatment tree cover and soil nitrate-nitrogen concentration (Arnold and others 1964; Clary and Jameson 1981; Schott and Pieper 1986). The vegetation changes following mechanical treatment on these sites followed the generalized successional models of Arnold and others (1964), Erdman (1970), Barney and Frischknecht (1974) and Tress and Klopatek (1987), although the rate of succession was faster than these investigators observed after fire. However, in some cases, threshold effects may limit recovery of understory communities even when openings are created (Koniak and Everett 1982; Laycock 1991).

The effect of several slash treatment alternatives (removal, clustering, scattering) on vegetation response following mechanical treatment has also been investigated, and no effects were found after a 2-year period (Brockway and others 2002). However, slash burning resulted in a four-fold increase in the abundance of non-native forbs in a treatment area in northern Arizona (Haskins and Gehring 2004). Temporary increases in rates of nutrient mineralization and nitrification of soils have also been observed after slash burning, however N losses through volatilization may be significant (DeBano and others 1987; Tiedemann 1987). Over longer time periods, leaving slash on the ground to gradually decay permits a more extended period of nutrient release; retards the loss of nitrogen from the system; reduces runoff and protects the soil against erosion; aids in nutrient conservation by acting as concentration points for nutrients lost from other areas on the site; and provides recruitment sites for trees and herbaceous vegetation (Brockway and others 2002; Ernest and others 1993; Evans 1988; Gottfried and Severson 1994; Hastings and others 2003; Jacobs and others 2002; Ludwig and Tongway 1995; Wood and Javed 1992).

Less information exists on the role of mechanical and chemical treatments in improving wildlife habitat or water yields. Deer and elk use increased in areas where the tree overstory was removed or greatly reduced, as did small mammal densities, but there was no effect of mechanical treatment on songbird use (Albert and others 1995, 2004; Baker and Frischknecht 1971; O'Meara and others 1981; Sedgewick and Ryder 1987; Severson 1986).

Although many control programs were justified on the basis of increasing water yields, increased streamflow is unlikely on sites where annual precipitation is less than 46 cm (18 in.) and where annual precipitation exceeds potential evapotranspiration; these characteristics apply to most pinyon-juniper woodland sites (Hibbert 1979; Gottfried and others 1995). Research conducted at Beaver Creek and Corduroy Creek in Arizona failed to demonstrate any increase in streamflow following tree control treatments except in an area where annual precipitation was > 46 cm and dead trees were left standing following treatment which affected wind movements and reduced soil evaporation; increased water yields disappeared after dead trees were harvested (Baker 1984; Clary and others 1974; Collings and Myrick 1966). However, at Beaver Creek, removal of the tree overstory increased soil moistures relative to untreated areas when soil depth was > 30 cm (11.8 in.).

Synthesis – Livestock grazing removes fine fuels needed to carry surface and mixed-severity fires that likely maintained the structure and composition of pinyon-juniper savannas and shrub woodlands historically. Fire history reconstructions collected at a limited number of sites (representing these pinyon-juniper types) show the virtual elimination of surface and/or mixed-severity fire as a disturbance agent after 1880 when livestock numbers increased over most of Arizona and New Mexico. In addition, historical fuelwood cutting and, more recently, mechanical/chemical treatments have changed woodland age structure, tree density and cover values in all woodland types where they have been applied but there are no reliable estimates of the total number of acres treated or annual treatment levels. Present-day wildfires and fire treatments have facilitated the spread of non-native grasses and forbs; two of these species have already been shown to alter either fire effects (*Salsola iberica*) or fire frequency (*Bromus tectorum*) in pinyon-juniper woodlands. Climate variation, including droughts and wet periods, have likely interacted with human-caused disturbances, intensifying or confounding their effects on herbaceous vegetation cover, soil erosion rates, historical fire regimes, and tree densities.

12.5 Effects of Anthropogenic Disturbance

Patch Composition of Vegetation

Overstory – Numerous studies provide evidence that pinyon-juniper woodlands have increased in density and extent over the last 100 years (Figures 12-9 and 12-10; Cottam and Stewart 1940; Davis and Turner 1986; Gottfried and Ffolliott 1995; Huffman and others 2006b; Landis and Bailey 2005; Leopold 1924; Springfield 1976; West and others 1975). Pinyon and juniper have invaded grasslands and former pinyon-juniper savanna at both high and low elevations (Arnold and others 1964; Blackburn and Tueller 1970; Cottam and Stewart 1940; Johnsen 1962). Watson (1912) noted advances of juniper savanna at the expense of grassland as early as the turn of the century. In existing savannas and woodlands, trees have replaced formerly more abundant shrubs and/or perennial grasses, leaving skeletons of dead shrubs in the understory and/or large areas of bare soil in intercanopy spaces that are susceptible to soil erosion (Allen 2001; Brockway and others 2002; Jacobs and others 2002; Leopold 1924; West and others 1975). Comparing aerial photographs from 1935 and 1991, Miller (1999) documented a greater than 69% loss of grasslands and an 80% loss of juniper savannas to pinyon-juniper woodlands in the Negrito Creek watershed in west-central New Mexico; in addition, there was a 26% loss of open canopy pinyon-juniper woodlands to closed canopy woodlands. In northern Arizona, Ffolliott and Gottfried (2002) observed an increase in the abundance of *P. edulis* and *J. monosperma* in pinyon-juniper woodlands between 1938 and 2001, with pinyon increasing at a higher rate than juniper and now comprising 36% of the tree canopy, an increase from 24% in 1938.

Once established, pinyon generally increases in numbers and size faster than juniper, and several authors have argued that increases in tree density in woodlands over the last 150 years have been due primarily to pinyon increases (Blackburn and Tueller 1970; Huffman and others 2006b; Jameson 1965; Tausch and others 1981); this hypothesis has not been tested regionally. Other authors have suggested that tree density increases in pinyon-juniper woodlands on the Colorado Plateau may be part of a natural successional process, giving way (or not) to a more open canopy as trees mature (Eisenhart 2004; Floyd and others 2004).

Figure 12-9. Repeat photography sequence taken in 1912 (top) and 1997 (bottom) at Carrizo Mountain foothills near Carrizozo, New Mexico. Photograph depicts landscape-wide increase in density in pinyon-juniper stands (Photographs courtesy of Hollis Fuchs 2002).

Figure 12-10. Repeat photography sequence taken in 1936 (top) and 1995 (bottom) from area of Whipple's expedition, near Ash Fork, Arizona. Although juniper density had already increased at the time of the first photo, the foreground and the ridge in the midground in the second photo depict extensive infilling by trees and loss of grass over the last 60 years (Photo courtesy of Shaw 2006).

Understory – Increased overstory canopy cover in pinyon-juniper woodlands has led to a reduction in understory plant cover (shrubs, forbs, and grasses) and productivity (Arnold and others 1964; Blackburn and Tueller 1970; Huffman and others 2006b; West 1984) as well as declines in plant species richness and diversity (Huffman and others 2006b; Tress and Klopatek 1987). These decreases have been greatest on sites with shallow soils (40-60 cm; 16-24 in.) and/or on southerly aspects and are least noticeable in areas where tree overstory exceeded 30%

canopy cover prior to 1860 (Cottam and Stewart 1940; Koniak and Everett 1982; Springfield 1959; Tress and Klopatek 1987).

In some areas of northern New Mexico, the advance of sagebrush upslope has converted juniper savanna to open juniper-sagebrush woodland (Dick-Peddie 1993a).

Herbaceous layer – In addition to information in the **Overstory** section, blue grama, a warm-season perennial grass, has increased in northern Arizona at the expense of cool-season perennial species and now dominates in areas where it was not dominant before European settlement (Daniel and others 1966).

Patch or Stand Structure of Vegetation

Canopy Cover Class (%) or Canopy Closure – Current canopy cover values for areas that were historically savannas range from 4% to 30% (Baxter 1977; Landis and Bailey 2005; Moir 1979). On Anderson Mesa, for example, savanna stands increased in canopy cover on two soil types over the last 145 years to approximately 30%, a change from pre-settlement conditions when canopy cover was estimated at 4.5% and 11.5% (Landis and Bailey 2005). On Anderson Mesa, historical grass open woodland stands (on limestone-derived soils) currently have a canopy cover of 44%, a sizeable increase from the 14.7% cover value that Landis and Bailey (2005) estimated for the pre-settlement woodland.

Canopy cover in present-day pinyon-juniper shrub-woodlands ranges between 26% and 33% (Lymbery and Pieper 1983; Naylor 1964), which is outside the range of pre-settlement values.

In persistent woodlands, current canopy cover values range between 28.6% and 60% (Grier and others 1992, Merkle 1952). In a study by Grier and others (1992), young stands (mean tree age < 100 years) had a mean canopy cover of 28.6% (17.8% for pinyon, 10.8% for juniper), while mature stands (mean tree age > 250 years) had a mean canopy cover of 40.7% (20% for pinyon, 20.7% for juniper). The current canopy cover for persistent woodlands is within the range of historical canopy cover for this type. Persistent woodlands in all of the above studies occurred on shallow, rocky soils and had low shrub and herbaceous cover.

Other pinyon-juniper woodlands whose historical type was not distinguished by study authors or that we could not distinguish from information provided in the studies have current canopy cover values ranging from 35% to 80% (Gottfried 2004; Julius 1999; Moir 1979).

Finally, in a landscape change study conducted in southwestern New Mexico, Miller (1999) found that 41% of the Negrito Creek watershed was composed of closed canopy pinyon-juniper woodland (> 40% canopy cover) in 1991, an increase from only 30% of the watershed in 1935; pinyon-juniper savanna (<10% canopy cover) and open woodland (10% to 40% canopy cover) decreased over the same period from 23% to 14.5%. Grasslands also declined in the study area from 10% in 1935 to only 1% in 1991.

Structure Class (Size Class) – We found one study that documented both the size-class and age-class distributions for a pinyon-juniper woodland in northern Arizona; this study is summarized in the **Age Class** section. In addition, six other studies provide size-class distributions for present-day woodlands. On Deadman Flat, most junipers (*J. monosperma*) and pinyons (*P. edulis*) are in the larger size classes (>1 m, or 3.28 ft., in height), but pinyons are

proportionately (and in absolute terms) more abundant in smaller size classes, indicating recent pulses of establishment (Ffolliott and Gottfried 2002). The overall size (height) distribution is bell-shaped for junipers with most individuals between 1.5 and 2.7 m tall (4.9 and 8.9 ft tall), while the pinyon height distribution is flatter with trees < 2.4 m (7.9 ft) tall more or less evenly distributed in all size classes down to 0.3 m (1 ft); below this, pinyons are rare. Mean tree height is 1.5 m and 2 m (4.9 and 6.6 ft.) for pinyon and juniper, respectively, with only a few trees > 4 m tall (13 ft) in either species (Ffolliott and Gottfried 2002). In earlier studies on this site (e.g., 1938, 1948, 1958), junipers were more abundant than pinyons in almost all size classes, including seedlings and taller trees, suggesting that juniper established first and more prolifically on the site up until 1958. The current size class distribution on Deadman Flat is significantly different from size class distributions derived from relict sites and reconstructions for this type, in showing episodic recruitment (and/or high survival) in contrast to the historical pattern of constant recruitment (and/or survival). Mean tree size in present-day stands is also significantly smaller than in historical stands, suggesting that the present-day stands are dominated much more heavily by smaller trees than historical stands were.

At a site near Los Alamos on the Pajarito Plateau, pinyon (*P. edulis*) and oneseed juniper (*J. monosperma*) show size class distributions that mimic an inverse-J shaped curve, suggesting a relatively constant rate of regeneration and mortality; trees less than 5 cm (2 in.) diameter at the base represent the most abundant class (Martens and others 2001). This pattern is also observed in southern New Mexico, in northern Arizona and in east central Arizona, where current pinyon-juniper stands are dominated by trees less than 12 cm (4.7 in.) diameter at the base or 7 cm (2.8 in.) in diameter at breast height (Garrett and Garrett 2001; Gottfried 2004; Gottfried and Ffolliott 1995). This pattern is similar to the historical size distributions observed at reference sites and in historical reconstructions.

Three studies report mean tree heights and/or diameters in present-day woodlands in Arizona and New Mexico but did not provide complete size distributions. On these three sites (north-central Arizona, Coconino Plateau, Ft. Stanton Experimental Ranch), mean tree heights range from 2.9 m to 4.9 m (9.5 to 16.1 ft) for pinyon (*P. edulis*) and from 1.27 m to 5.4 m (6.9 to 17.7 ft) for juniper (*J. monosperma, J. deppeana*); mean diameters range from 8.4 cm to 25 cm (3.3 to 9.8 in.) for pinyon and from 8.7 cm to 36.6 cm (5.3 to 14.4 in.) for juniper (Grier and others 1992; Jameson 1965; Pieper and others 1971). Generally, most values are within the range of historical mean tree heights and diameters, but data for both pinyon and alligator juniper (*J. deppeana*) from north-central Arizona and Ft. Stanton sites fall below the minimum values in the historical range for mean tree height and mean tree diameter (3.6 m and 12.7 cm, respectively). The smaller mean tree sizes potentially indicate sites that are more heavily dominated by young trees than occurred historically.

Life Form – We found no information regarding the impact of anthropogenic disturbances on life form for pinyon-juniper woodlands.

Density – Numerous studies have documented an increased number of trees in pinyon-juniper woodlands after the introduction of domestic livestock and the 1890's drought by comparing current and pre-settlement (before 1850-1880) tree densities. Table 12-7 summarizes current tree densities in historical savanna, shrub and open woodland, and persistent woodland as well as pre-settlement densities when available. Current tree density in historical savannas and open woodlands ranges from 175 to 1154 trees/ha (71 to 467 trees/ac), an increase of 2.1 to 33.3 times

over the pre-settlement density. By comparison, current tree density in historical shrub or persistent woodland ranges from 325 to 2120 trees/ha (132 to 858 trees/ac), an increase of 1.4 to 3.3 times over pre-settlement densities. The density change value for persistent woodlands is based on information from a single site. There is no statistical difference in the current mean tree density in historical savannas compared to historical shrub, open and persistent woodlands (Mann-Whitney U test, U = 18, p = 0.14; mean tree density = 590.1 trees/ha in savanna vs.902.2 in woodland), however the increase over pre-settlement densities was significantly greater in historical savannas than in woodlands (U = 45, p = 0.007).

Table 12-6. Density of extant and pre-settlement trees in present-day pinyon-juniper woodlands by historical woodland type (Romme and others 2003). Site codes are ANDB: Anderson Mesa, basalt-derived soils, ANDS: Anderson Mesa, sandstone-derived soils, ANDL: Anderson Mesa, limestone-derived soils, BNMPA: Bandelier National Monument, pumice-argillic soils, BNMPNA: Bandelier National Monument, pumice, non-argillic soils, BNMNPNA: Bandelier National Monument, upper elevation, non-pumice, non-argillic soils, BPJM: Bandelier National Monument, upper elevation pinyon-juniper with *M. montanus* understory, CANJ: Canjilon, Carson National Forest, DF: Deadman Flat, FSER: Ft. Stanton Experimental Ranch, GGUP: Uncompaghre Plateau, Gunnison Gorge, KVTK: KV Tank, n. Arizona, LINC: Lincoln National Forest, SMUP: Uncompaghre Plateau, Sims Mesa, TUSA: Tusayan, Kaibab National Forest, WACA: Walnut Canyon National Monument.

See text for further details. Relict sites are not included in the table.

Site	Density, current (trees/ha)	Density, pre-settlement (trees/ha)	Increase (x times)	Woodland type	Source
GGUP	175	22	8.0	Savanna [1]	Eisenhart 2004
ANDS	261	122	2.1	Savanna [2]	Landis and Bailey 2005
SMUP	488 [3]	85 [3]	5.7	Savanna [1]	Eisenhart 2004
MM	506	60	8.4	Savanna	Cottam and Stewart 1940
LINC	618	25	24.7	Savanna	Garrett and Garrett 2001
ANDB	653	110	5.9	Savanna [2]	Landis and Bailey 2005
MEDB	664	--		Savanna	Martens and others 1997
DF	717	25	28.7	Savanna [4]	Ffolliott and Gottfried 2002
BNMPA	767	23	33.3	Savanna	Julius 1999
BNMPNA	1154	120	9.6	Savanna	Julius 1999
ANDL	654	246	2.7	Open woodland [2]	Landis and Bailey 2005
SM	325	---		Shrub woodland	Naylor 1964
FSER	612	---		Shrub woodland	Lymbery and Pieper 1983
TUSA	730	621	1.2	Shrub woodland [5]	Huffman and others 2006a
CANJ	735	450	1.6	Shrub woodland [5]	Huffman and others 2006a
WACA	870	265	3.3	Shrub woodland	Despain and Mosley 1990
BNMNPNA	1296	393	3.3	Shrub woodland or persistent woodland	Julius 1999
KVTK	332	--		Persistent woodland	Grier and others 1992
BPJM	2120	1407	1.5	Persistent woodland	Barnes 1983

[1] Eisenhart (1994) identified as savanna based on 1880 tree densities.
[2] Landis and Bailey (2005) identified pinyon-juniper woodland type for each site on Anderson Mesa based on canopy cover values and the spatial distribution of trees; on limestone-derived soils, grass but not shrubs were reported in their woodland vegetation description suggesting a pinyon-juniper grass open woodland.
[3] Mean value for 12 stands.
[4] Jameson (1965) identified the site as a savanna based on density and canopy cover of pre-settlement trees (>5 ft tall).
[5] Mean tree densities (for pinyon, juniper) were calculated on a site-wide basis (i.e., include stands dominated by ponderosa pine) and, therefore, may be conservative estimates of pre- and post-settlement tree density in pinyon-juniper stands.

The pattern observed across sites of greater increases in tree density in stands supporting fewer trees pre-settlement can also be observed within sites. Eisenhart (2004) reported that stands with > 200 trees/ha (81 trees/ac) but less than 320 trees/ha (130 trees/ac) in 1880 have actually decreased in density over the last 120 years on the Uncompaghre Plateau, most likely due to competition and density-dependent thinning of trees . In contrast, historical savannas (< 200 trees/ha) there have increased in density by 5.7 to 8 times since 1880. Combining savanna and woodland stands on the Uncompaghre Plateau, the vast majority of trees established between 1700 and 1850 during periodic wet periods that followed the 1500's drought, making the post-1880 increase in tree density for savannas less significant by comparison (Eisenhart 2004). Julius (1999) also found that since 1880 less dense (and younger) pinyon-juniper stands have increased more in density than denser (older) stands have at Bandelier National Monument.

Others studies also report current tree densities in pinyon-juniper woodland in Arizona, New Mexico, southern Utah and southern Colorado, but these studies did not provide sufficient information for us to identify woodland type, nor did the authors attempt to do this. These studies also did not provide estimates of pre-settlement tree density (Barnes 1983; Eisenhart 2004; Fowells 1965; Gottfried 2004; Gottfried and Ffolliott 1995; Huffman and others 2006b; Jameson 1965; Kennedy 1983; Moir and Carlton 1987). Current tree densities at these sites ranged from 420 to 1400 trees/ha (170 to 567 trees/ac).

Age Structure – A number of studies document age-structure distributions for present-day pinyon and juniper stands in northern Arizona and New Mexico, southwestern Colorado and Utah, and western Colorado. These distributions show considerable variability in stand initiation dates (1340 to 1840), shape (flat, truncated-normal, weakly to strongly J-shaped), and the location of peaks associated with periods of enhanced establishment (Betancourt and others 1993; Cottam and Stewart 1940; Despain and Mosley 1990; Eisenhart 2004; Floyd and others 2004; Huffman and others 2006b; Julius 1999; Landis and Bailey 2005; Romme and others 2003; Tausch and West 1988). For example, Floyd and others (2004) found that persistent woodland stands dating from 1740 and earlier at Mesa Verde showed a roughly J-shaped distribution that they interpreted as "representing a quasi-steady state condition with no strong directional trends in age or density", although without more demographic information on recruitment and age-specific mortality this claim cannot be substantiated. An establishment peak between 1900 and 1940 was evident in most age-structure reconstructions regionally (Cottam and Stewart 1940; Eisenhart 2004; Floyd and others 2004; Julius 1999; Romme and others 2003; Tausch and West 1988), although the initiation of enhanced recruitment began as early as 1860 to 1880 for some sites and stands (Cottam and Stewart 1940; Despain and Mosley 1990; Huffman and others 2006b; Julius 1999; Landis and Bailey 2005; Tausch and others 1981). This wave of recruitment may reflect moister climatic conditions in the early part of the 20th century and/or the lack of wildfire following the increase in livestock numbers in the 1870s and 1880s (Covington and Moore 1994, Grissino-Mayer 1995, Miller and Wigand 1994; Miller and Tausch 2001; Tausch and others 1981). In most cases, it has resulted in post-settlement trees dominating stands (see **Size Class** and **Density** sections). Although not as widespread, a number of pinyon-juniper stands also showed a marked increase in numbers (recruitment) in the early- to mid-1700s, a time of increased precipitation following a period of severe drought (Eisenhart 2004; Landis and Bailey 2005). Interestingly, the 1950's drought, a 200- to 500-year return event, was evident in some but not the majority of pinyon-juniper stands studied (Betancourt and others 1993; Julius 1999; Landis and Bailey 2005; Tausch and West 1988).

Patch Dispersion – The distribution of trees and dispersion of patches in present-day pinyon-juniper woodlands have been described in several studies. Ffolliott and Gottfried (2002) provided a qualitative description of tree distribution at Deadman Flat in northern Arizona that includes groups of same-aged trees with crowns touching, isolated trees at least one tree length away from other trees or groups of trees; and small trees growing under larger trees.

Landis and Bailey (2005) found that juniper trees were highly clumped at all spatial scales on 3 different soil types on Anderson Mesa, northern Arizona. However, on limestone-derived soils, the change from a random distribution of tree clumps at larger spatial scales in 1860 to the current clumped pattern suggests that the site "was comprised of smaller sized clumps, and through time, encroaching trees filled in the interspaces, causing clumps to reach maximum spatial scales". In another study, Huffman and others (2006a) provided estimates of the landscape covered by different-aged pinyon stands at two sites as well as estimates of mean patch size for these stands. At Tusayan in northern Arizona, approximately 80% of the landscape was comprised of pinyon stands that were > 200 years old with 250-300 year old stands covering the greatest proportion of the study area (36%); most stand age classes had mean patch sizes < 5 ha (12.4 ac) whereas 250-300 year old stands had a mean patch size of 30 ha (74 ac). In contrast, at Canjilon in northern New Mexico, patches were generally larger with a mean patch size of 24 ha (59 ac) and 79 ha (195 ac) for 200-250 year old stands and 250-300 year old stands, respectively; younger (< 200 years) and older (> 300 years) stands had a mean patch size of 10 ha. In addition, 200-250 year and 250-300 year stands covered 71% of the Canjilon site and were located predominantly along the southern portion of the study area.

Finally, Muldavin and others (2003) provided patch size information for pinyon-juniper stands that were burned < 60 years ago, 60-100 years ago, and > 100 years ago and that were discernable from surrounding vegetation based on differences in canopy cover. At the San Andres Mountain site, mean patch size ranged from 61 to 104 ha (151 to 257 ac), fire patches comprised 56% of the landscape and most fire patches were located at lower elevations at the ecotone of pinyon-dominated woodlands and juniper savanna. At the Oscuras Mountain site, mean patch size ranged from 81 to 143 ha (200 to 353 ac), fire patches comprised only 23 percent of the landscape (i.e., lower fire frequency); and most fire patches were located at lower elevations adjacent to juniper savannas.

Synthesis - Information on current vegetation structure and composition in pinyon-juniper woodlands comes from sites across Arizona and New Mexico and indicates that pinyon-juniper woodlands have generally increased in density and extent over the last 100 to 140 years. Tree density and canopy cover have increased in historical pinyon-juniper savannas and shrub woodlands, resulting in a loss of understory cover. Woodlands with historically higher tree density have exhibited little to no increase in density and canopy cover over the last 100 years. Ranges for tree density values in pinyon-juniper woodlands are no longer mutually exclusive; that is, savannas, shrub woodlands, and persistent woodlands now show overlapping ranges, making identification of historical pinyon-juniper type difficult based on current tree density information. Shifts in herbaceous species composition toward greater abundance of warm-season perennial grasses such as blue grama at the expense cool season perennial species have also occurred in pinyon-juniper woodlands and this have been attributed to post-settlement grazing pressure.

Age structure distributions generally show a wave of recruitment, starting as early as 1860 and continuing to as late as 1940, in most stands. Some sites show tree size and age distributions that are similar to historical distributions (e.g., young trees dominating the site, mean tree sizes within the historical range, and a pattern of constant regeneration). Other sites show evidence of episodic recruitment and/or have smaller mean tree sizes than have been documented historically, indicating a disproportionately greater number of young or small trees. This variation does not seem to sort out by pinyon-juniper woodland type, but rather seems to occur across a variety of sites. Age and size distribution information is not available for any sites in southern Arizona or southern New Mexico, suggesting that further research is needed in these areas to better understand current changes in vegetation structure.

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Appendix 21-A: USDA Forest Service (1997) Woodland Associations
PINYON series:
1) Twoneedle pinyon pine (*Pinus edulis*)/ sand bluestem (*Andropogon hallii*). KEY CRITERIA: The pinyon-juniper overstory occurs on sandy soils. The understory may be grassy and sand bluestem and/or sandhill muhly (*Muhlenbergia pungens*) are common to abundant. Or if the understory is shrubby, a dominant shrub is sand sagebrush (*Artemisia filifolia*). LOCATION: Occurs locally in the landscape in central and northern New Mexico on valley plains with deep, sandy soils. Typical soil is a Typic Ustipsamments.
2) Twoneedle pinyon pine/manzanita (*Arctostaphylos pungens*). KEY CRITERIA: This plant association exhibits a chaparralic expression of shrubs (i.e. dense shrubs), but relatively minor herbs. Pointleaf manzanita is often well represented or abundant. LOCATION: Known from Grand Canyon National Park, north of the Colorado River from Shivwits Plateau to Naukoweep Valley. Also on the Globe Ranger District (RD), Tonto National Forest (NF).
3) Twoneedle pinyon pine/ big sagebrush (*Artemisia tridentata)*. KEY CRITERIA: This plant association has big sagebrush in the understory and a pinyon-juniper overstory. LOCATION: This plant association is found on highly variable soils and topography from 6,000' to 7,400' (1830 to 2255 meters). The Utah juniper phase occurs from southwest Colorado and southern Utah into northern Arizona and north-central New Mexico. The oneseed and Rocky Mountain juniper phases occur from north-central New Mexico into southern Colorado. Mean annual precipitation is about 16" per year (Erdman and others 1969).
4) Twoneedle pinyon pine/blue grama (*Bouteloua gracilis)*. KEY CRITERIA: Understory is essentially grassy with blue grama as a dominant grass, and mountain muhly (*Muhlenbergia montana*) is scarce or absent. Generally warm season grasses are more prevalent. Shrubs may be scarce to well-represented, but oaks are not common. LOCATION: Widespread in New Mexico, Arizona, Colorado and Utah. Occurs in valleys or on elevated plains, piedmont slopes, and mountain slopes. Elevations range from 5100' to 7600' (1550 to 2320 m) depending on aspect and soils. Occurs on a wide variety of soil and parent materials. Mean annual precipitation is approx. 15-18" per year.
5) Twoneedle pinyon pine/ mountain mahogany (*Cercocarpus montanus*). KEY CRITERIA: This plant association exhibits a chaparralic expression of shrubs (i.e. dense shrubs), but relatively minor herbs. True mountain mahogany is common, often well represented or abundant. Gray oak may be well-represented, but other oaks are poorly represented. Tree cover is generally light to moderate. LOCATION: Found from southeastern Arizona and south-central New Mexico, north to southern Colorado. Generally occurs on steep to gentle slopes from 5,200' to 7,600' (1585 to 2315 m). Soils are often Udic or Lithic Ustochrepts, and surface is usually rocky (30-70% surface cover with cobbles). Mean annual precipitation is about 18" per year. Mean annual air temperature is 53° F.
6) Twoneedle pinyon pine/ rabbitbrush-Apache plume (*Chrysothamnus nauseosus-Fallugia paradoxa)*. KEY CRITERIA: Rubber rabbitbrush and/or Apache plume are abundant along washes; trees present include pinyon and juniper. LOCATION: Widespread geographically, but often occurs very locally in the landscape in intermittent washes and river terraces. Often between 6300' to 7500' (1920 to 2290 m). Common soils include Typic Ustifluvents, Fluventic Haplustolls, and Fluventic Ustocherpts; these are often incised with arroyos or gullies. Also found on deep cindery soils. Site specific determination of soils may be required.
7) Twoneedle pinyon pine/blackbrush (*Coleogyne ramosissima)*. KEY CRITERIA: Blackbrush is well represented as a shrub; pinyon and Utah juniper make up the overstory and grasses and forbs are common. LOCATION: Known from the Grand Canyon National Park in northern Arizona where it occurs on elevated plains and benches, 3,500' to 6, 200' (1070 to 1890 m). Soils

are generally shallow (lithic) and stony and may develop from a wide variety of parent materials.
8) Twoneedle pinyon pine/Arizona fescue (*Festuca arizonica)*. KEY CRITERIA: This grassy woodland often has an overstory of tall twoneedle pinyon pine and juniper. Arizona fescue is present, and usually at least common. LOCATION: Occurs in northern Arizona and west central New Mexico (Mt. Taylor RD, Cibola NF and Quemado RD, Gila NF). Mean annual precipitation is 18" per year.
9) Twoneedle pinyon pine/pine muhly (*Muhlenbergia dubia)*. KEY CRITERIA: A savanna (grassy) woodland with an overstory dominanted by alligator juniper and twoneedle pinyon pine. Pine muhly dominates the grass understory, but other grasses are present. There is a sparse shrub understory, primarily of wavyleaf oak. LOCATION: Presently known from the Sacramento and Guadalupe Mountains of south-central New Mexico where it occurs on moderate slopes of predominately southeastern exposures from 6,000' to 7,300' (1830 to 2225 m). Often found on slightly to moderately rocky sites.
10) Twoneedle pinyon pine/New Mexico muhly (*Muhlenbergia pauciflora)*. KEY CRITERIA: A savanna (grassy) woodland with an overstory of twoneedle pinyon pine and oneseed juniper. New Mexico muhly is usually part of the grass understory, but not necessarily the dominant grass. This may be one of the drier pinyon/grass plant associations. LOCATION: Presently known from the Sacramento and Capitan Mountains, and White Sands Missile Range, New Mexico, on upper slopes and ridges, gentle to moderate, south-facing slopes, and on steep north- to west-facing slopes. 6,200' to 7,300' (1890 to 2225 m).
11) Twoneedle pinyon pine/Muttongrass (*Poa fendleriana)*. KEY CRITERIA: This woodland often has an overstory of tall twoneedle pinyon pine and juniper, with a grassy understory. Muttongrass is common, but Arizona fescue is absent. LOCATION: Occurs in northern Arizona, southern Utah, southern Colorado, and central and northern New Mexico (including the Sandia, Jemez, and Chuska Mountains, and White Sands Missile Range). In the Jemez Mountains, elevations range from 6,500' to 7,100' (1980 to 2165 m) on north and east slopes. In the Sandia Mountains, this type can be found up to 8,400' (2560 m) on south-facing slopes. Loamy soils are generally noncalcaeous with high silt and clay content. Mean annual precipitation is 18" per yr and mean annual air temperature is 47° F.
12) Twoneedle pinyon pine/Stansbury cliffrose (*Purshia stansburiana*). KEY CRITERIA: The overstory consists of pinyon pine and Utah juniper and occasionally Gambel oak. The shrubby understory includes Stansbury cliffrose; antelope bitterbrush and usually mountain mahogany are scarce or absent while oaks are poorly represented. LOCATION: Occurs on plains and hillslopes from central Arizona to southern Utah and southwestern Colorado and locally in western New Mexico. General elevation range is 6,000' to 6,800' (1825 to 2075 m). Soils are frequently Lithic Haplustolls or Lithic Ustochrepts on calcareous parent materials. Mean annual precipitation is 14" to 16" per year.
13) Twoneedle pinyon pine/antelope bitterbrush (*Purshia tridentata*). KEY CRITERIA: The overstory consists of pinyon pine, Utah juniper and occasionally Gambel oak. The shrubby understory includes antelope bitterbrush; big sagebrush is scarce or absent. Cover of grasses and forbs is usually very sparse. LOCATION: Known from northwestern New Mexico and southwestern Colorado where it occurs on mesa and scarps, 6,900' to 7,500' (2100 to 2290 m). Soils are fine sandy loams to sandy loams, with shales and sandstones as parent rock. This plant association is often associated with the "San Jose Formation". Mean annual precipitation is 9” to 14" per year.
14) Twoneedle pinyon pine/Gambel oak (*Quercus gambelii*). KEY CRITERIA: Must have at least 5% cover of Gambel oak; ponderosa pine may be accidental. LOCATION: Local in southern New Mexico, becoming more widespread in central and northern New Mexico, and

north of the Mogollon Rim in Arizona. Usually occurs on moderate and steep mountain slopes, 6,300' to 8,000' (19202400 m) on cool, wet sites such as draws of north slopes. Mean annual precipitation is about 18” per year. Mean annual temperature is about 48° F.
15) Twoneedle pinyon pine/ wavyleaf oak (*Quercus x pauciloba*). KEY CRITERIA: Wavyleaf oak is generally abundant (>25%), pinyon is in the tallest stratum. Herbs are usually poorly represented. LOCATION: Found in southern (Sacramento Mountains, Lincoln NF and Mescalero Apache Reservation), central New Mexico, and locally in northern New Mexico (including northeastern mesas); 6,000' to 8,000' (1,830 to 2,440 m) on moderate to steep mountain slopes, often on lithic skeletal soils.
16) Twoneedle pinyon pine/rockland. KEY CRITERIA: Pinyon trees growing on rock with very little soil. LOCATION: Scattered locations throughout New Mexico and Arizona, including the malpais area near the Zuni Mountains of west-central New Mexico and the Peloncillo Mountains of southwestern New Mexico. Occurs on lava flows (malpais) or soils that are < 4" to bedrock.
17) Twoneedle pinyon pine/sparse. KEY CRITERIA: Understory is sparse, although annual plants may be well represented. Tree cover of pinyon and juniper is usually dense, often forming a closed canopy. LOCATION: Widespread geographically, but often occurs locally in the landscape (i.e. not usually extensive). Often between 6,500' to 7,300' (1980 to 2225 m) on basaltic mesas or hillslopes; soils are widely variable.
18) Twoneedle pinyon pine/Dore needlegrass (*Stipa nelsoni var. dorei*). KEY CRITERIA: Pinyon dominates the overstory and grasses dominant the understory; Arizona fescue is absent while Dore needlegrass or Schribner needlegrass (*Stipa schribneri)* are common to well represented. Alligator juniper may be accidental. A distinct litter layer is also usually present. LOCATION: Known from the Sacramento Mountains, Jicarilla Mountains, and White Sands Missile Range, and Rowe Mesa (Pecos RD, Santa Fe NF). Occurs on moderate to gentle slopes, 6,200' to 7,300' (1890 to 2225 m). Generally not found on rocky sites.
19) Singleleaf pinyon pine (*Pinus monophylla*)/pointleaf manzanita (*Arctostaphylos pungens*). KEY CRITERIA: This central Arizona plant association exhibits a chaparralic expression of shrubs (i.e. dense shrubs), but relatively minor herbs. Pointleaf manzanita and shrub live oak (*Quercus turbinella*) are at least common, often well represented or abundant; rucifixion thorn (*Canotia holacantha*) is absent. LOCATION: Known from central Arizona below the Mogollon Rim, north in Oak Creek Canyon to Sedona. Elevations are mostly between 4,800' to 6,000' (1,470 to 1,830 m) on a wide variety of slopes, aspects, landforms, and soils. Mean annual precipitation is 20” per year, with a hot, dry season during May and June.
20) Singleleaf pinyon pine/blue grama (*Bouteloua gracilis*). KEY CRITERIA: A pinyon juniper woodland with a rich understory of grasses, usually including blue grama. Singleleaf pinyon is the dominant tree, along with either alligator juniper or Utah juniper. LOCATION: Primarily known from central Arizona south of the Mogollon Rim (Prescott and Tonto NF’s and Ft. Apache Reservation). Occurs on elevated plains and alluvial valley plains. Elevations range from 4,900' to 5,600' (1,495 to 1,705 m). Mean annual precipitation is around 22" per year; mean annual air temperature is 52° to 56° F.
21) Singleleaf pinyon pine/ crucifixion thorn (*Canotia holacantha*). KEY CRITERIA: A pinyon-juniper woodland amid a shrubby and grassy matrix containing crucifixion thorn. LOCATION: Found in central Arizona south of the Mogollon Rim (including Prescott and Tonto NF’s, Fort Apache and San Carlos Apache Reservations), this association occurs on dissected, erosional escarpments and hills from 3,500' to 4,000' (1,075 to 1,225 m). Mean annual precipitation is 20" per year. Mean annual air temperature is 59° to 61° F.
22) Singleleaf pinyon/shrub live oak (*Quercus turbinella*). KEY CRITERIA: Singleleaf pinyon, Utah juniper, and oneseed juniper are found in the overstory of this shrubby woodland. Shrub

live oak is well represented and often abundant, crucifixion thorn is absent, mountain mahogany is poorly represented, and manzanita (*Arcostaphylos sp.*) is scarce or absent. The cliffrose phase is on calcareous soils. LOCATION: Primarily found in central Arizona mostly south of the Mogollon Rim tapering to occasional stands near the New Mexico border, this association occurs on a wide variety of soils and landforms. This association may represent the lowest elevational limits of singleleaf pinyon.

23) Singleleaf pinyon pine/banana yucca (*Yucca baccata*). KEY CRITERIA: Tree cover is luxuriant with an overstory of singleleaf pinyon, Utah juniper and possibly oneseed juniper. Herbs are scarce, primarily annuals, and shrubs are common. LOCATION: Presently known from Ft. Apache Reservation where it occurs on steep south or west slopes around 6,200' (1,890 m).

24) Border pinyon (*Pinus discolor*)/Mexican orange (*Choisya dumosa var. arizonica*). KEY CRITERIA: This woodland is usually well stocked with border pinyon, alligator juniper and occasional Arizona white oak (*Quercus arizonica)* in the canopy. The shrub dominated understory includes Mexican orange, but oaks or mountain mahogany are poorly represented (<5%). LOCATION: Known from the Dragoon Mountains in southeastern Arizona, this type has been found or steep, north-facing slopes around 6,500' (1,980 m). Parent materials are limestone and altered limestone. Mean annual precipitation is 20” per year.

25) Border pinyon/bullgrass (*Muhlenbergia emersleyi*). KEY CRITERIA: A grassy woodland on moderate to steep slopes occurring in southeastern Arizona and southwestern New Mexico. Bullgrass is usually present, although it may be lacking in some locations. Border pinyon and alligator juniper dominate the overstory, and oaks are present but scarce in the overstory. LOCATION: Presently known from southeastern Arizona, and southwestern New Mexico, but probably also occurs in northern Mexico. Usually on moderate to steep, north-facing colluvial slopes from 5,800' to 6,600' (1,770 to 2,010 m). Soils are erosional and may be very shallow (<5") and interrupted by exposed bedrock. Mean annual precipitation is 18” to 19" per year.

26) Border pinyon/pinyon ricegrass (*Piptochaetium fimbriatum*). KEY CRITERIA: This woodland is found in washes, drainages, and other alluvial settings. Border pinyon is the dominant tree species. The understory is dominated by grasses and may include pinyon ricegrass, although it is not always present. The shrub layer may be minor or significant, and includes oaks and yuccas. LOCATION: Occurs in southeastern Arizona and central and southwestern New Mexico. Elevations range from 5,500' to 6,000' (1,680 to1,830 m) often on north slopes. Soils may be Typic Ustifluvents and Cumulic and Typic Ustochrepts. Mean annual precipitation is 18” to 19" per year.

27) Border pinyon/silverleaf oak (*Quercus hypoleucoides*). KEY CRITERIA: A shrub-dominated woodland on moderate to steep slopes occurring in southeastern Arizona and southwestern New Mexico. Shrubs include a mix of oaks, manzanita, and others, but silverleaf oak is at least common. Border pinyon and alligator juniper dominate the overstory, and ponderosa pine (*Pinus ponderosa*) and Chihuahua pine (*Pinus leiophylla*) may be occasional on microsites. LOCATION: Presently known from southeastern Arizona in the Chiricahua and Santa Catalina Mountains and on the Clifton Ranger District near the New Mexico border, and in extreme southwestern New Mexico in the Animas Mountains. Often on steep, upper slopes and ridgetops, and elevated plains from 6,200' to 7,000' (1,890 to 2,130 m). Soils are extremely rocky, or shallow and rocky, often broken by rock outcrops. Mean annual precipitation is 20” to 21" per year; mean annual air temperature is 53° F with relatively mild winters.

28) Border pinyon/Toumey oak (*Quercus toumeyi*). KEY CRITERIA: A shrubby woodland on rhyolite parent materials occurring in southeastern Arizona and southwestern New Mexico. Border pinyon, alligator juniper, and redberry juniper (*Juniperus erythrocarpa*) dominate the

overstory; Toumey oak or its hybrids are present. LOCATION: Presently known from southeastern Arizona and extreme southwestern New Mexico (Animas Mountains), but probably also occurs in northern Mexico. On rhyolite parent materials, usually from 5,900' to 6,100' (1,800 to 1,860 m). Mean annual precipitation is 19" per year; mean annual air temperature is 58° F.
29) Border pinyon/evergreen sumac (*Rhus virens var. choriophylla*). KEY CRITERIA: A shrubby pinyon-juniper woodland occurring in southeastern Arizona. Mountain mahogany is well-represented and evergreen sumac is usually present to well-represented; oaks are not a significant part of the shrub mix. Border pinyon and redberry juniper dominate the overstory. LOCATION: Presently known from southeastern Arizona (Mule and Huachuca Mountains). Found on limestone parent materials from around 5,500' (1675 m) on north slopes to 6,500' (1980 m) on south slopes. Mean annual precipitation is 19" per year; mean annual air temperature is 55° F; mean January air temperature is 46° F (Fort Huachuca).

JUNIPER series:
30) Alligator juniper (*Juniperus deppeana*)/pointleaf manzanita (*Arctostaphylos pungens*). KEY CRITERIA: A juniper woodland where alligator juniper is dominant with an abundant (>25% cover) shrubby understory . LOCATION: Known from a single location at the foot of the Bradshaw Mountains at approximately 5,300 feet (1,610 m) on Typic Haplustalfs on a variety of slopes.
31) Alligator juniper/blue grama (*Boutelous gracilis*). KEY CRITERIA: A juniper woodland where alligator juniper is dominant with a scarce (<1% cover) or common (>1% cover) shrubby understory; gray oak is scarce (< 1% cover). LOCATION: Known from southern New Mexico and Arizona south of the Mogollon Rim, at approximately 5,200' (1,600 m) on north aspects and to 6,600' (2,610 m) on south aspects. *J. deppeana*, *B. gracilis*, *Prosopis glandulosa* phase is presently known only from the New Mexico-Arizona border between Glenwood, New Mexico and Clifton, Arizona.
32) Alligator juniper/desert ceanothus (*Ceanothus greggii*). KEY CRITERIA: A juniper woodland where alligator juniper and oneseed juniper are codominant with a well represented (>5% cover) shrubby understory with mountain mahogany or desert ceanothus common (>1% cover). Twoneedle pinyon may occur as an accidental tree. LOCATION: Sacramento and Guadalupe Mountains, NM; at elevations of 6,000' to 6,500' (1,824 to 1,975 m) on south slopes with limestone parent materials.
33) Alligator juniper/bullgrass (*Muhlenbergia emersleyi*). KEY CRITERIA: A juniper woodland where alligator juniper is dominant with a well represented (>5% cover) shrubby understory; gray oak is common (>1% cover) and bullgrass with its associates produce abundant (>25%) cover. LOCATION: Known only from Guadalupe Mountains, NM.
34) Alligator juniper/skunkbush sumac (*Rhus trilobata*). KEY CRITERIA: A juniper woodland where alligator juniper and oneseed juniper are codominant with a well represented (>5% cover) shrubby understory with true mountain mahogany or desert ceanothus scarce (<1% cover). LOCATION: Moderately steep and steep hill and mountain slopes, at elevations of 4,600' to 6,900' (1,400 to 2,100 m) on gravelly or cobbly soils. Known from southern New Mexico in winter-mild climates and from Guadalupe Mountains in the vicinity of Glenwood, New Mexico and adjoining Arizona.
35) Utah juniper (*Juniperus osteosperma*)/big sagebrush (*Artemisia tridentata*). KEY CRITERIA: This plant association has big sagebrush in the understory, and a Utah juniper and oneseed juniper overstory which seldom exceeds 15% canopy cover. LOCATION: This plant association occurs from northern Arizona and northern New Mexico to SW Colorado, Utah, Nevada and Wyoming. Typically found at elevations between 5,700' to 7,000' (1740-2130 m) on

a wide range of slopes from level to steeply sloping piedmont plains; soils often on gullied alluvium. Mean annual precipitation is 10” to14” per year, much of this as winter snow.
36) Utah juniper/blue grama (*Bouteloua gracilis*). KEY CRITERIA: A juniper savanna with a rich understory of grasses, usually including blue grama. Utah juniper is the dominant tree, although pinyon pine may be present, but is usually confined to microsites. LOCATION: Primarily known from central and northern Arizona where it occurs in valleys and on elevated plains and piedmont alluvial fans. Elevations range from 5,000' to 6,000' (1525 to 1825 m).
37) Utah juniper/tobosagrass (*Hilaria mutica*). KEY CRITERIA: A juniper savannah, often on heavy clay soils. Tobosagrass, curly-mesquite (*Hilaria belangeri*), and/or panic grass (*Panicum obtusum*) are present among an abundant cover of herbs. Juniper trees dominate the overstory, but rarely reach over 10% cover. Singleleaf pinyon pine may be present in the P. monophylla phase, but is usually only occasional or a minor climax species. LOCATION: Widespread south of the Mogollon Rim, this plant association is typically found on elevated or valley plains, from 4,300' to 5,900' (1315 to 1800 m). Soils generally have a heavy clay content. Mean annual precipitation is approximately 1718" per year (to 20" per year in the *P. monophylla* phase). Mean annual air temperature is 55° to 61° F.
38) Utah juniper-oneseed juniper (*Juniperus monosperma*)/sparse. KEY CRITERIA: Understory is sparse, although annual plants may be well represented; juniper overstory is well represented to abundant. Existing plants may be on pedestals, providing evidence of recent erosion. LOCATION: Widespread in New Mexico and Arizona where it commonly occurs between 5,000' to 6400' (1525 to 1950 m) on a wide variety of soils and parent materials, often adjoining grasslands of valley plains or piedmont slopes; can occur on special sites such as erosional badlands or gypsum soils. Mean annual precipitation is 12” to 16” per year.
40) Oneseed juniper (*Juniperus monosperma*)/sand bluestem (*Andropogon hallii*). KEY CRITERIA: This juniper woodland has a grassy understory which includes sand bluestem and/or sandhill muhly. The shrub broom dalea (*Psorothamnus scoparius*) is also present. LOCATION: Occurs locally in the landscape in central and northern New Mexico on valley plains with deep, sandy soils. Typical soil is Typic Ustipsamments.
41) Oneseed juniper/Bigelow sagebrush (*Artemisia bigelovii*). KEY CRITERIA: This plant association has Bigelow sagebrush in the understory, and a oneseed juniper overstory which seldom exceeds 10% canopy cover. Twoneedle pinyon may be accidental. LOCATION: This plant association occurs locally in northern Arizona and possibly northern New Mexico, southern Utah, and southwestern Colorado. Found on limestone mesas and hillslopes, on very shallow rocky soils (Lithic Ustochrepts and Lithic Ustorthents) from 5,000' to 7,000' (1520 to 2130 m). Mean annual precipitation is about 14" per year.
42) Oneseed juniper/big sagebrush (*Artemisia tridentata*). KEY CRITERIA: This plant association has big sagebrush in the understory, and a oneseed juniper overstory which seldom exceeds 10% canopy cover. LOCATION: This plant association is found in northern New Mexico on elevated and piedmont plains from 6,600' to 6,800' (2010-2070 m). It may occur on a wide variety of soils including calcareous Typic Ustochrepts and Typic Haplustalfs. Mean annual precipitation is about 14" per year.
43) Oneseed juniper/sideoats grama (*Bouteloua curtipendula*). KEY CRITERIA: A juniper woodland, often on steep, rocky slopes. Oneseed juniper is the dominant tree, although pinyon pine may be present, but is usually only occasional or minor climax species. Sideoats grama is common. LOCATION: From southern New Mexico and southeastern Arizona (NOMI phase) into southern Colorado (typic phase). Typically on steep, colluvial slopes of escarpments, and hill or mountainsides with >15% slope; soils, from a wide variety of parent materials, are often stony or rocky, and may be interrupted by rock outcrops. Elevations range from 4,900' to 6,400'

(1500 to 1950 m). Mean annual precipitation is approx. 15” to 19” per year. Mean annual air temperature is 55° to 57° F.
44) Oneseed juniper/blue grama (*Bouteloua* gracilis). KEY CRITERIA: A juniper savanna with a rich understory of grasses, usually including blue grama; sideoats grama is scarce or absent. Oneseed juniper is the dominant tree, although pinyon pine may be present, but is usually confined to microsites; twoneedle pinyon is accidental. LOCATION: Widespread in New Mexico, Arizona, and southern Colorado where it occurs in valley plains, piedmont alluvial fans on a wide variety of soil and parent materials. Elevations range from 5,500' to 7,000' (1675 to 2130 m). Mean annual precipitation is approximately 14” to 16” per year.
45) Oneseed juniper/ rabbitbrush-Apache plume (*Chrysothamnus nauseosus-Fallugia paradoxa*). KEY CRITERIA: Rubber rabbitbrush and/or Apache plume are abundant along washes, streamsides and terraces. Trees include oneseed juniper, Rocky Mountain juniper, and in northern Arizona, Utah juniper; an infrequent or occasional narrowleaf cottonwood may be present. In High Sun Mild (mild w/ summer moisture) climates, gray oak may also be occasional. LOCATION: Widespread geographically, but often occurs very locally in the landscape along streamsides and river terraces of intermittent washes, often between 4,300' to 6,500' (1315 to 1980 m). Common soils include Typic Ustifluvents, Fluventic Haplustolls, and Fluventic Ustochrepts (site specific determination of soils may be required); soils are often cut by gullies and arroyos.
46) Oneseed juniper/winterfat(*Krascheninnikovia lanata*). KEY CRITERIA: The soils are calcareous and the plant association has winterfat present. The overstory consists of oneseed juniper. LOCATION: This plant association is known from western and central New Mexico where it occurs in localized settings (i.e. not extensive) on valley plains from 6,000' to 6,500' (1830 to 1980 m). Soils are calcareous.
47) Oneseed juniper/ sacahuista-lechuguilla (*Nolina microcarpa-Agave lechuguilla*). KEY CRITERIA: An open cover of oneseed juniper with a strong shrubby component, consisting primarily of sacahuista and lechuguilla, with a grassy understory. LOCATION: Known from the Guadalupe Mountains and the southern portion of the Sacramento Mountains in southern New Mexico, this association occurs on limestone slopes, 4,300' to 4,600' (1315 to 1400 m).
48) Oneseed juniper/wavyleaf oak (*Quercus x pauciloba*). KEY CRITERIA: A chaparral woodland association where shrubs are generally abundant (>25%) and dominated by wavyleaf oak. Junipers are of low stature (<16' or 5 m). LOCATION: Found in southern and central New Mexico, and locally in northern New Mexico where it occurs on rocky slopes between 15-40% slopes, intergrading to scarp woodland with increasing steepness and rocky outcrop terrain, 6,000' to 6,500' (1830 to 1980 m).
49) Redberry juniper(*Juniperus erthryocarpa*)/crucifixion thorn (*Canotia holacantha*). KEY CRITERIA: A juniper woodland of redberry juniper and Utah juniper amid a shrubby and grassy matrix containing crucifixion thorn. LOCATION: Found in central Arizona south of the Mogollon Rim (including Prescott and Tonto NF’s, Fort Apache and San Carlos Apache Reservations), this association occurs on dissected elevated plains, eroding breaks of valley fill alluvia, and steep, erosional hills. Soils are of calcareous parent materials, and in the thermic (mean annual soil temperature is 59° to 72° F) soil temperature regime. Mean annual precipitation is 16” to 20” per year. Mean annual air temperature is 59° 63° F.

Chapter 13 - Vegetation Models for Southwest Vegetation

13.1 Introduction

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

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

13.2 Methodology

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

"Historic Range of Variation for Potential Natural Vegetation Types of the Southwest" (Schussman and Smith 2006).

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

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

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

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

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

In order to minimize the variability in model output that arises from variation in sample size (i.e., the number of pixels modeled) and to standardize models for all PNVTs, we conducted a sensitivity analysis of a "simple" grassland model to determine the appropriate number of sampling units (pixels) and model runs (simulations) to use in scenario analysis. The "simple" grassland model is a 4 box model that includes 3 transitions (fire, drought, and plant growth) (Figure 13-1). Results of the sensitivity analysis showed that variation due to sample size was minimized when 1,000 or more sample units were used (Table 13-1). Based on this result we set the modeled landscape at 1000 pixels and ran each scenario for a total of 10 runs (simulations) in order to calculate a mean and standard deviation value for each modeled state. This analysis also highlighted the need to perform a sensitivity test on the range of values identified for the probability of a transition in each model, as seemingly small differences in the probability of a transition had large impacts on model output when the transitions are very **frequent** yet had little impact on model output when transitions are very **infrequent** (Tables 13-2 and 13-3). Given these results and the fact that information from different studies of the same PNVT yielded a range of values for the frequency of transitions, we decided to use sensitivity analysis to determine the impact of imprecise information on all models for which a range of values was identified in the literature. Specifically, when a range of values was given for a transition, we ran the model using the average value, as well as the high and low ends of the value range and reported the results from all three model runs.

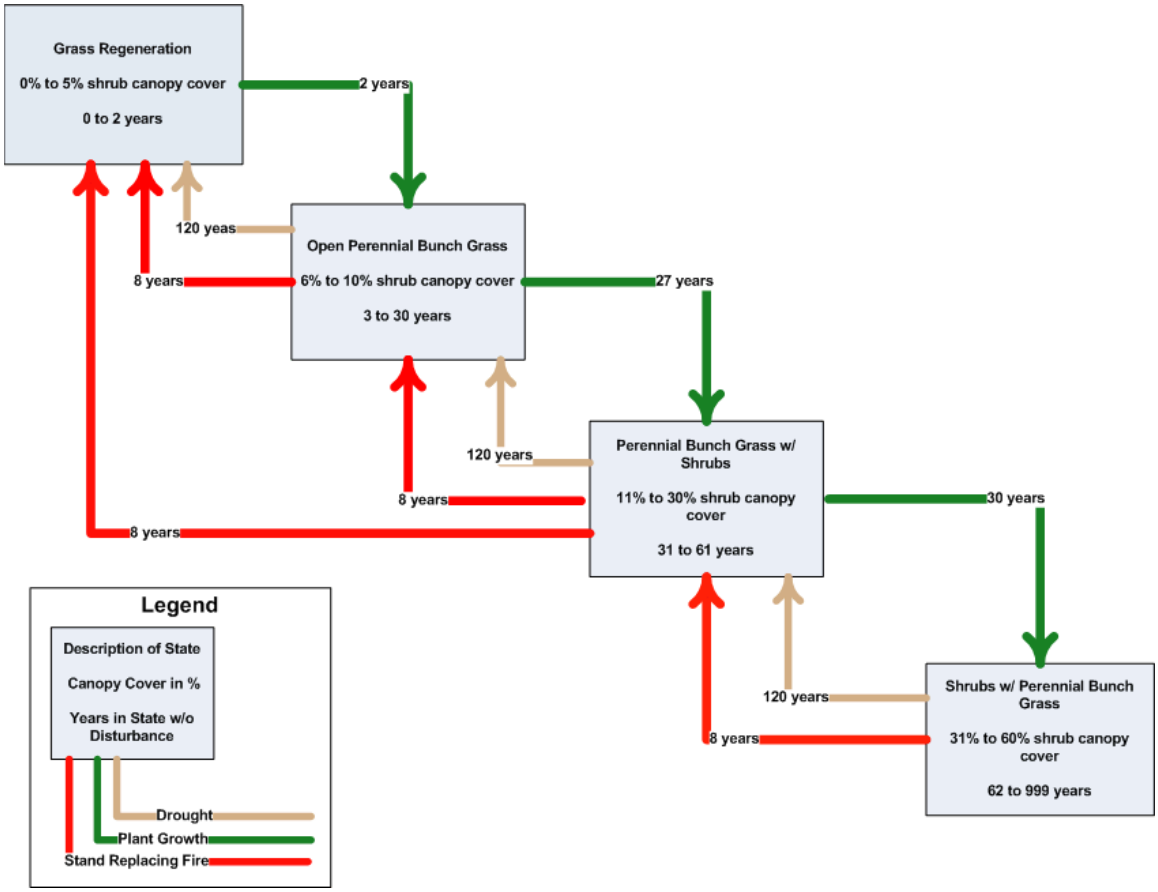

Figure 13-1. Simple grassland model used in sensitivity testing of VDDT software

Table 13-1. Sensitivity analysis showing the stabilization of model output, as indicated by average percent of the modeled landscape in each vegetation state and average standard deviation, when model is run at or above 1,000 sample units.

Sample Number	State A (%)	Standard Deviation (%)	State B (%)	Standard Deviation (%)	State C (%)	Standard Deviation (%)	State D (%)	Standard Deviation (%)
10	14.0	10.6	54.2	16.1	17.8	11.0	14.0	11.8
100	15.1	3.8	56.6	5.3	17.2	3.3	13.1	3.0
1000	13.5	1.0	57.4	1.4	16.5	1.0	12.5	1.1
10000	13.7	0.4	57.3	0.6	16.4	0.4	12.6	0.4

Table 13-2. Sensitivity analysis showing dramatic changes in the average percent of the landscape in each state when the frequency of the fire transition (every 8 years) is multiplied by a range of values between 0 and 2. Increasing the frequency of fire by a factor of 2 drastically changed the average percent of states A, C, and D. Similarly, decreasing the frequency by roughly a half (Every 20 years) also drastically changed the average percent of most of the states.

Fire Frequency Multiplier	Fire Frequency	State A (%)	State B (%)	State C (%)	State D (%)
0.0	none	0.0	0.0	0.0	100
0.4	Every 20 years	1.1	18.1	22.2	58.6
0.8	Every 10 years	8.6	48.5	20.1	22.8
1.0	Every 8 years	13.7	57.6	16.2	12.5
1.2	Every 7 years	15.7	66.3	11.8	6.2
1.6	Every 5 years	26.9	66.0	5.2	1.9
2.0	Every 4 years	31.5	65.9	1.9	0.0

Table 13-3. Sensitivity analysis showing little change in the average percent of the landscape in each state when the frequency of the drought transition (every 120 years) is multiplied by 0, 1, and 2. Increasing the frequency of drought by a factor of 2 increased the average percent of state A by only 5%, while state B saw a change of 6%. Decreasing the probability to 0 decreased A by about 4% and B by 2.5%, increased D by 5% and had little effect on state C.

Drought Frequency Multiplier	Drought Frequency	State A (%)	State B (%)	State C (%)	State D (%)
0.0	None	16.3	56.4	14.5	12.8
1.0	Every 120 years	20.4	59.0	13.2	7.4
2.0	Every 60 years	15.9	65.3	13.0	5.8

We ran the historic models for 1000 years, as this temporal span corresponds with the widest frame of reference offered by the scientific literature. Additionally, 1000 year long runs allowed for infrequent transitions, such as stand replacing fires in the spruce fir PNVT and extreme drought events in all PNVTs, to occur several times within each simulation. Ultimately, this level of temporal depth makes for a robust historic model that allows for multiple replicates of infrequent events while not over reaching the bounds of our historic knowledge. Current models were run for 120 years as this corresponds to the post-European settlement era when large scale changes to historic fire, flooding and grazing regimes in the Southwest were first documented.

We began all historic model runs with equal proportions of the modeled landscape in each state. For example if the model had 4 states then the historic model would start the 1000-year simulation with each state making up 25% of the landscape. However, for the current models, we began the 120-year simulations with the proportions of each state equal to the output values (900-year averages) from the historic model runs. This allowed us to simulate how the last 120 years of management has changed the historic proportions of the vegetative states.

Variability - One of the main concerns with vegetation models is the use of mean values to model the frequency of events that are variable in space and time. This is a valid concern and criticism as the mean value is not a metric for describing variability. For example, in the Madrean pine oak woodland, mean fire return interval (MFRI) for all fires, at 15 sites located in Arizona and northern Mexico, ranged between 3 and 7 years, while the MFRI for fires that scarred 25% of the trees ranged between 5 and 13.2 years (Fulé and Covington 1998; Fulé and others 2005; Kaib and other 1996; Swetnam and Baisan 1996; Swetnam and others 1992). Additionally, the minimum and maximum number of years between any given fire was between 1 and 38 years (Fulé and others 2005; Kaib and other 1996; Swetnam and Baisan 1996; Swetnam and others 1992).

Given concern over the use of mean values and the variability in the frequency of Southwest transitions we investigated the ability of VDDT to model variability in vegetation dynamics. Specifically, we analyzed year to year variability in our simple grassland model. Results of this analysis showed there to be little variability from year 10 to 1000 (13- 2). This was due to the consistency with which the probability of the transitions occurred (i.e., every year, each sample unit in which fire could occur had a probability of 0.12 of having that fire) as well as the large number of sampling units.

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

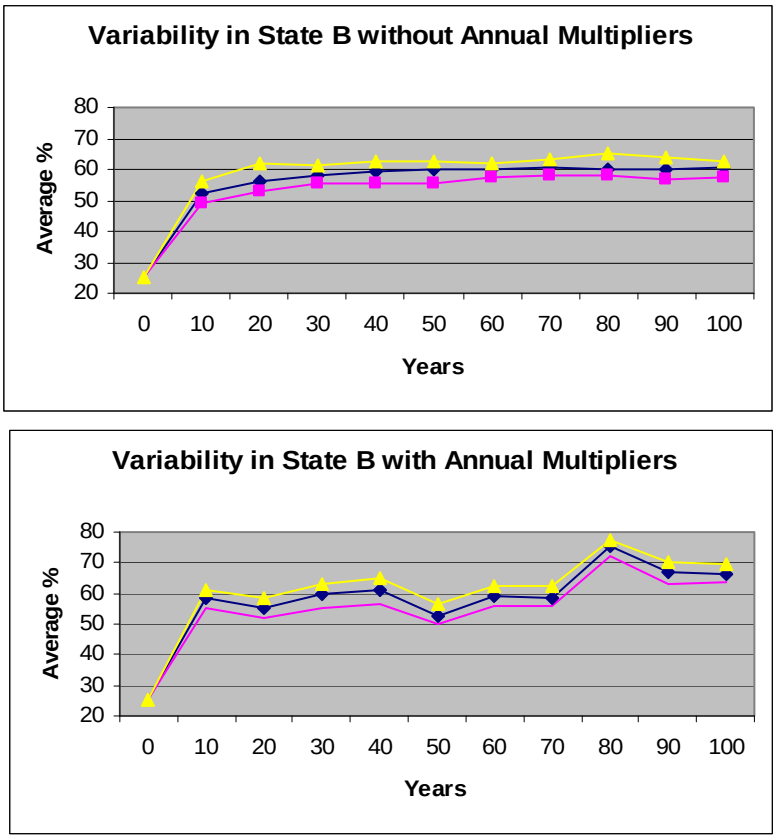

Figure 13-2. Comparison of year to year variability in state B of the simple grassland VDDT model with and without the use of annual multipliers. Maximum values in yellow, average values in blue, and minimum values in pink.

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

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

Fire Variability – The connection between fire occurrence and climate in the Southwest has been well established (Crimmins and Comrie 2004; Swetnam and Betancourt 1998). Based on this knowledge and our understanding of modeling year to year variability with VDDT, we modeled climate-mediated fire transitions using the annual multiplier function. To run the annual multiplier function we needed to identify the frequency of year types that increased and/or decreased fire occurrence as well as identify the magnitude of the effect. We obtained this information by analyzing the percent of regional fires that occurred in each year type using contingency table analysis (for an

example see (Table 13-5). The regional fires were identified by Swetnam and Betancourt (1998) on the basis of having been recorded at two thirds of all sites, 41 of 63 sites, with fire history reconstructions in the Southwest; these fires occurred between1709 and 1879. The year types (severe drought, drought, normal, wet, and extremely wet) were identified from an in-depth analysis of Ni and others' (2002) 989-year winter precipitation reconstruction. Details of this analysis are described in a companion document entitled "Assessing Low, Moderate, and High Severity Drought and Wet Events Across the Southwestern United States from Year 1000 to 1988" (Schussman 2006).

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

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

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

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

Model Reporting –We developed a descriptive state and transition diagram for historic and current conditions as well as a current photographic diagram for each PNVT. For all historic transitions, the historic frequency, or range of frequencies, of each transition is identified. Additionally, all possible transitions for which there was some level of information are included in the state and transition model. However, only those transitions for which the transition impacted the majority of the vegetation within a

PNVT and for which information regarding the frequency and effect of the transition on the vegetation was consistently identified were included into the quantitative VDDT models. Identification of the frequency of transitions, source(s) used to identify transitions, and assumptions made in identifying the frequency or effect of transitions are detailed in tabular form for both historic and current models, for each PNVT separately in the following chapters.

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

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Chapter 21 - Pinyon-Juniper Model

In this chapter, we present state-and-transition models for each of the three pinyon-juniper types identified by Romme and others (2003): pinyon-juniper grass savanna, pinyon-juniper shrub woodland, and pinyon-juniper persistent woodland. Models depicting historic (pre-1880) and current (1880 to present) vegetation dynamics for each type were developed based on published information summarized in the Pinyon-juniper Historical Range of Variation (HRV; Chapter 12) for use in the VDDT (Vegetation Dynamics Development Tool) program. The VDDT software allows the user to model succession as a series of vegetation states that differ in structure, composition, and cover and to specify the amount of time it takes to move from one vegetation state to another in the absence of disturbance. Various disturbance agents affecting the movement of vegetation between states can then be incorporated if sufficient information exists on their frequency and effects on vegetation (e.g., surface fires, stand-replacing fires, grazing, insect outbreaks). By varying the types and rates of disturbance across the landscape, the effects of different management treatments, such as wildland fire use, fire suppression, prescribed burning, grazing practices, and mechanical fuel treatments, on future vegetation can be investigated. While VDDT models can be used to "game play" with different management scenarios, the models we ran in this analysis only include states and transitions for which there is published information to support their inclusion within the model.

21.1 Pinyon-Juniper Savanna Vegetation Dynamics – Pinyon-juniper savanna, most often occurring as juniper savanna, is characterized by sparse juniper and/or pinyon, scattered shrubs, and dense herbaceous growth including perennial grasses, forbs and annuals. Pinyon-juniper savanna occurs on deep, fine-textured soils on broad valley bottoms and on rolling hills with few barriers to fire spread; it may also occur on rockier sites where productivity is high and understory grasses form a more-or-less continuous fuel layer. The type is common where most of the annual precipitation comes in the summer such as in southern and central Arizona and New Mexico, but may also occur in the upper ecotone with ponderosa pine forest in northern New Mexico where conditions are relatively mesic and a strong summer precipitation component exists.

Pinyon-juniper savanna was historically maintained by frequent low severity surface fires. Spread of these predominantly lightning-initiated fires was supported by a dense, continuous layer of herbaceous vegetation, primarily perennial grasses and forbs. Based on relict sites and pre-settlement reconstructions, tree densities range from 60 to 122 trees per hectare and canopy cover values range from 5% to 12% (Table 12-5; Fflolliott and Gottfried 2002; Landis and Bailey 2005). While there is one recorded site near the upper ecotone with ponderosa pine forest where pre-settlement canopy cover values up to 20% occurred and pre-settlement tree density exceeded that of other documented historical savannas (Landis and Bailey 2005; Chapter 12), there has been no formal identification of a separate type in the literature. Knowledgeable individuals recommend calling this pinyon-juniper grass open woodland because of its greater canopy cover and tree density (still to be determined) compared to savanna (J. Youtz, pers. comm.). Like historic savannas, these open canopy woodlands also supported a dense, continuous growth of perennial grasses and forbs and low tree densities and their open woodland structure also may have beenwas maintained by frequent surface fire. In this document,

we will refer to pinyon-juniper savannas based on the published information that supports this designation, but our model results and conclusions will apply equally well to these open canopy woodlands (with a perennial grass dominated understory). Climate influences production of herbaceous fuels and fuel moisture, and therefore likely affected historic fire regimes of pinyon-juniper savannas. In ponderosa pine forests, fire years were correlated with drought, especially when preceded by one to three years of high precipitation, while years with few fires were correlated with wet years (Swetnam and Baisan 1996; Swetnam and Betancourt 1990).

Since 1880, natural surface fires have essentially been eliminated from pinyon-juniper savannas. Disruption of the historical fire regime has been attributed to livestock grazing and/or severe drought coupled with soil erosion which reduced/removed surface fuels needed for fire spread. This has resulted in a dramatic increase in tree densities in historic savanna and potentially also in open canopy woodlands at the upper ecotone (see Table 12-7, Pinyon-juniper HRV).

Various insect species, including pinyon Ips, are capable of killing pinyons and junipers and are endemic to the Southwest; outbreaks of these pests have been closely tied to drought conditions (Breshears and others 2005; Waring and Cobb 1992; Wilson and Tkacz 1992) but their historic frequency, population impacts and areal extent of infestation are not well understood. Two outbreaks of pinyon Ips have been reasonably well documented in the last century, permitting some conservative estimates of their historic return interval and ecological impacts (described below). However, the extent of these outbreaks and tree mortality rates may have been exacerbated by post-settlement disruption of the surface fire regime and the resulting increase in tree density in this pinyon-juniper type.

Vegetation Models - Historic (pre-1880) and current (1880 to present) models for pinyon-juniper savanna are shown in Figures 21-1 and 21-2. Additionally, we used information in the HRV (Chapter 12) to estimate parameter values for transitions between model states (succession) and disturbance frequencies, allowing us to develop quantitative VDDT models. We discuss model parameters, output, and analysis for pinyon-juniper savanna below (Tables 21-1 through 21-4; Figure 21-3).

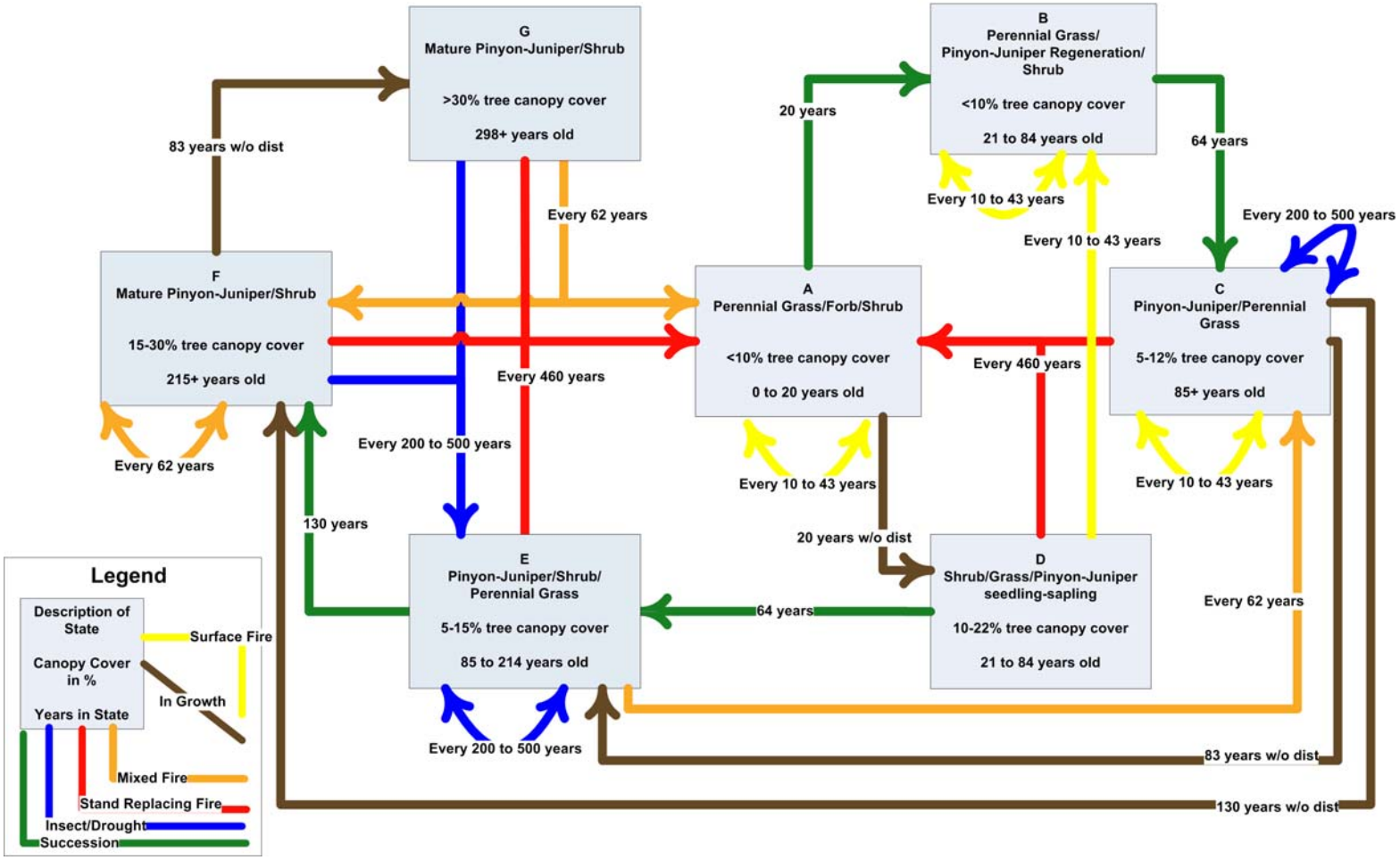

Figure 21-1. Conceptual Historic state and transition model for the pinyon-juniper savanna vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

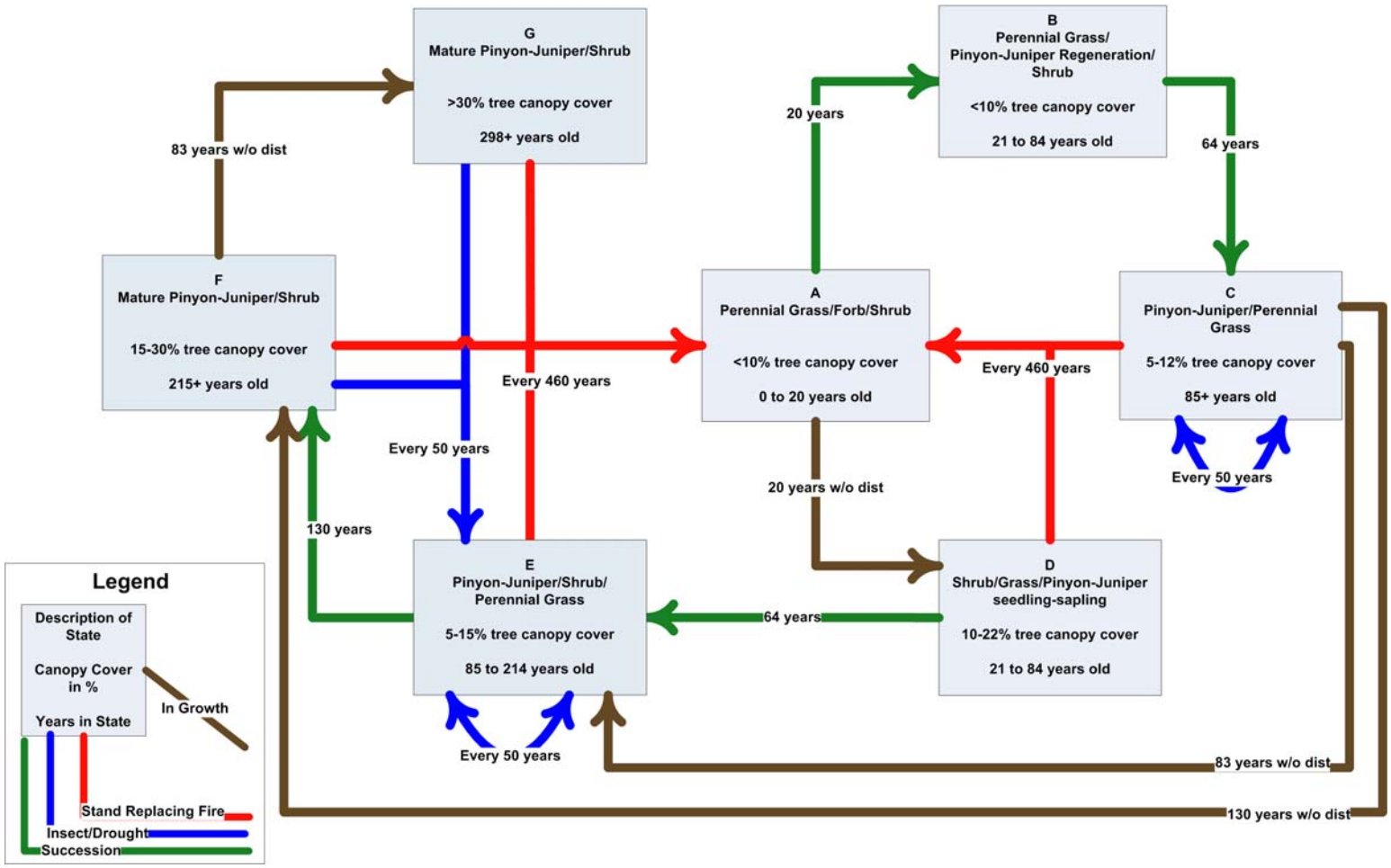

Figure 21-2. Conceptual Current state and transition model for pinyon-juniper savanna vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Model Parameters

In Tables 21-1 and 21-2 below, we describe the parameters included or not included within the Historic and Current VDDT models, as well as the sources of information and any assumptions made in creating model parameters.

Table 21-1. Identification of historic transition types, transition frequency or length, sources of information and assumptions made in developing the Pinyon-juniper savanna VDDT model.

Transition Type	Transition Frequency or Length	Source	Assumptions
Plant Growth (20 years, 64 years)	After 20 years, 64 years without a stand replacing fire (or disturbance event)	Allen 1989; Arnold and others 1964; Baisan and Swetnam 1997; Barnes and Cunningham 1987; Barney and Frishknecht 1974; Blackburn and Tueller 1970; Brown and others 2001; Erdman 1970; Floyd and others 2000; Gottfried and others 1993; Huffman and others 2003; Koniak 1985; Muldavin and others 2003; Schott 1984; Wilkinson 1997; Young and Evans 1978	Plant growth refers to the growth and ageing process of a stand in the absence of a stand-replacing fire. There are no fire history reconstructions that document historic stand-replacing fire (SRF) or succession after SRF in pinyon-juniper savanna. We assume that i) SRF's (or some other stand-replacing disturbance) occurred but were rare events as they were not recorded in savanna woodland fire histories that date back as early as 1400; and ii) succession in savannas is similar to that described for shrub woodlands, but that with surface fire the savanna understory stays primarily grassy. We base the latter assumption on the fact that i) pinyon-juniper savannas and shrub woodlands share many of the same perennial grass and shrub species, the latter increasing at the expense of the former in the absence of surface fire (USDA 1997); ii) historic savannas have increased in tree and shrub densities post-settlement due in large part to human alteration of the fire regime (Allen 2001; Brockway and others 2002; Humphrey 1953, 1958; Jacobs and others 2002; Leopold 1924; Miller 1999; McPherson 1997); and iii) present day tree densities in historical savannas and shrub woodlands are similar (Table 12-7). Studies of post-fire vegetation succession in shrub- or persistent woodland indicate that annual grasses and forbs dominate for the first 3 years.

			Perennial grasses become dominant on sites 5 to 6 years post-fire while shrubs become dominant (in terms of cover) 12 to 30 years post-fire; shrub dominance persists until trees become dominant 70-100 years after SRF (see sources). We took the mean value of the latter two time ranges to estimate the amount of time vegetation was in the perennial grass/forb/shrub and shrub/grass/tree seedling-sapling states, 20 years and 64 years, respectively. We have not included an annual stage in the savanna model because perennial grasses will resprout after fire and preclude annuals from becoming an important component of the herbaceous community. With a regime of frequent surface fires, shrubs will be continually knocked back and young trees will be thinned, resulting in a perennial grass/tree seedling-sapling state following the perennial grass/forb/shrub state along the main successional pathway (Arnold and others 1964; Leopold 1924; McPherson 1995).
In Growth	83 years	Arnold and others 1964; Howell 1941; Pieper and Wittie 1990	In growth refers to the infill of trees that would not have survived with recurrent surface or mixed severity fire and occurs after an estimated period of 83 years without fire. We arrived at this time value based on the following information. Howell (1941) estimated that it takes up to 75 years for a juniper trees to grow > 1.2 m tall, a size where post-fire survivorship is high (Pieper and Wittie 1990). Similarly, Miller and Tausch (2001) found that it takes pinyons and junipers up to 90 years to reach > 3 m tall, a height at which they found post-fire survivorship to be high; 83 years is the mean of these two estimates. If there is no surface fire during this period, tree recruitment and growth will likely increase tree density and canopy cover to levels characteristic of historic pinyon-juniper shrub woodland [see Table 12.7 for tree density increases in the absence of fire; tree density vs. canopy cover regression (cover = 0.031(tree density) + 10.3) derived from the following studies: Grier and others 1992; Landis and Bailey 2005; Lymbery and Pieper 1983; Rowlands and Brian 2001; Mason and others 1967]. At this

			point, we assume a mixed-severity fire is needed to reduce tree density and canopy cover, restoring a savanna structure.
Shrub Accumulation + In Growth (130 years)	After 83 years, then 47 years without fire	Tress and Klopatek 1987	Tress and Klopatek (1987) estimated that it took approximately 215 years for a shrub-woodland to fully recover in tree density, canopy cover and species composition following a SRF. This indicates an additional 47-year time period to move from an open canopy pinyon-juniper/shrub/perennial grass state to a mature, open canopy pinyon-juniper/shrub state after 83 years without fire.
Surface Fire	Every 10 to 43 years	Allen 1989; Baisan and Swetnam 1997; Brown and others 2001; Huffman and others 2006; Leopold 1924; Wilkinson 1997; Muldavin and others 2003	Mean fire return intervals (MFRI) for pinyon-juniper savanna based on analyses of fire-scarred trees range from 10 to 43 years; most estimates are point mean fire intervals (PMFI) based on analyses of individual trees which likely underestimate the stand-level MFRI. Restricted composite estimates all fall within the above range (Allen 1989; Huffman and others 2006); however, Baisan and Swetnam (1997) report composite estimates of the MFRI of between 6 and 11.2 years for a northern New Mexico site. We did not consider these latter estimates because they likely overestimate the stand-level fire occurrence (Baker and Ehle 2001; Baker and Shinneman 2004). In general, surface fires kill or topkill shrubs and remove young trees < 1.2 m but mortality of young trees is not always 100% (Arnold and others 1966, Dwyer and Pieper 1967; Johnson and others 1962). For example, if fire occurs in the shrub/perennial grass/tree seedling-sapling state, shrub mortality will create an opportunity for grass to rebound in the understory after fire, shifting the stand into the perennial grass/tree seedling-sapling/shrub state (Arnold and others 1964).
Mixed Severity Fire	Every 62 years	Huffman and others 2006	Historic PMFIs for mixed severity fire in shrub-woodland at the upper ecotone ranged from 42 to 81 years (mean value = 62 years). The effect of mixed severity fire in low and open cover shrub woodland stands will be a reduction in cover and mortality in younger trees, more similar to a surface fire than to a

			replacement fire, and in stands with shrub and grass understory, a fire will result in a shift to grass-dominated understory. In mature stands with greater than 30% canopy cover, we assume that a mixed severity fire will have both a stand replacing component (50%) and surface fire component (50%), in the latter case thinning trees and reducing canopy cover and in the former, sending the vegetation back to the perennial grass/forb/shrub state. In support of this, several studies found trees surviving successive fires that were > 100 years apart (Despain and Mosley 1990; Huffman and others 2006; Tausch and West 1985) while other authors report stand-replacing fires in pre-settlement closed woodlands (Table 12-3, HRV). We chose the 50-50 split because in the absence of better information, this break minimizes estimation error (Sokal and Rohlf 1994).
Stand Replacing Fire	Every 460 years	Allen 1989; Baisan and Swetnam 1997; Brown and others 2001; Huffman and others 2006; Leopold 1924; Wilkinson 1997; Muldavin and others 2003	Fire history reconstructions using fire-scarred trees provide a continuous record dating from 1400 to 1684, a period of roughly 320 to 600 years before present (mean = 460 years). Assuming that these stands initiated after a stand-replacing fire, this gives a conservative estimate of its frequency of (0.002).
Drought/Insect-Caused Tree Die-off.	Every 200 to 500 years	Allen and Breshears 1998; Betancourt and others 1993; Breshears and others 2005	Betancourt and others (1993) estimated that the 1950's drought was a 200- to 500-year return interval event; the 1996-2003 drought was wetter but warmer than the 1950's drought (Breshears and others 2005). The relationship between drought, insect outbreaks, and tree die-off in pinyon-juniper woodlands is well established (Allen and Breshears 1998; Gottfried and others 1995; Breshears and others 2005; Wilson and Tkacz 1992) however there is only one estimate of the amount (hectares) of pinyon-juniper woodland affected regionally and this is for the recent 1996-2003 drought (Breshears and others 2005). Since not all stands were affected during this drought (Mueller and others 2005; Negron and Wilson 2003), we adjusted the drought frequency by the estimated proportion of woodland in Arizona

			and New Mexico with discernable tree die-off (0.07) based on aerial surveys (Breshears and others 2005) to derive a patch-level probability of tree die-off during drought. Pinyon mortality was greater than juniper mortality, older individuals had higher mortality than younger ones, and denser stands were more susceptible to insect infestation and tree die-off was more severe than in less dense stands (Breshears and others 2005; Mueller and others 2005; Negron and Wilson 2003). Based on this information, drought/insect effects in tree-dominated states were modeled to move stands from higher canopy cover states to lower cover ones and to reset stand-age to the youngest age value for that state.

Table 21-2. Identification of current transition types, frequency of transitions, sources of information and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the Pinyon-juniper savanna (or open-canopy persistent woodland) VDDT model. Unless otherwise indicated (see below), we used the same transition types, and frequency or length of transitions as in the historic model (Table 21-1).

Transition Type	Transition Frequency or Length	Sources	Assumptions
Surface Fire, Mixed Severity Fire	Not Used in Current model	Allen 1989, 2001; Baisan and Swetnam 1997; Brown and others 2001; Huffman and others 2006; Leopold 1924; Wilkinson 1971; Muldavin and others 2003	Based on fire-history reconstructions and direct observation, we assume that surface fire and mixed severity fire have essentially ceased at the scale of pinyon-juniper occurrence in Arizona and New Mexico (7.5 million ha). Occasional surface fires and mixed severity fires do occur, but not at the same scale, either because fine fuels are too sparse for fire spread or enough fuels have accumulated in many areas to quickly shift surface fires to stand replacing ones (Allen 2001; Butler and others 1998). Prescribed fires and fire use are occurring in some areas at some times, but there is no published information on the amount of pinyon-juniper savanna (and former savanna) affected and it is likely that this treatment level is not within the historic range of variability for this system.
Stand Replacing Fire	Every 460 years	Allen 1989; Baisan and Swetnam 1997; Brown and others 2001; Huffman and others 2006; Leopold 1924; Wilkinson 1971; Muldavin and others 2003. The effects of stand replacing fire on vegetation have been documented by Savage and Mast (2005).	Cessation of surface fires and accumulation of fuels and development of fuel ladders has led to an increase in the frequency of stand replacing fires, especially during the last decade or two (Crimmins and Comrie 2004; Floyd and others 2004; Romme and others 2003). There is only a single study that provides information that would permit calculation of this increased SRF frequency for historic savanna or open-canopy persistent woodland at the upper ecotone (Crimmins and Comrie 2004), but there are technical problems with VDDT using this increased SRF frequency for the last 10-20 years in

			a relatively short (120-year) run. The increased SRF frequency is better applied to prospective scenario analyses (looking into the future) where the modeling time frame is 50 to 100 years. For this reason, we used the historic frequency for SRF.
Drought/Insect Outbreak	Once every 50 years	Allen and Breshears 1998; Breshears and others 2005	Two major droughts and regional outbreaks of pinyon Ips have occurred over the last century (e.g., 50-year return interval) and their combined mortality effects on pinyons and junipers have been quite variable on a population basis (Breshears and others 2005; Mueller and others 2005a; Negron and Wilson 2003). See above historical Drought/Insect Outbreak section for further detail.
Silvicultural Activities, Mechanical Treatments	Highly variable through time and across space, thus not included in the model.	Bahre 1991; Bahre and Hutchinson 1985; Gottfried and others 1995	Fuelwood cutting from pinyon-juniper systems was a major source of fuel for mining until the end of the 19th century and for domestic heating and cooking as late as the 1940's; pinyon-juniper fuelwood continues to be an important source for domestic heating and income generation in rural communities. However there are no regional estimates of amount (hectares) of fuelwood cutting that has occurred over the last 120 years, nor are there equivalent estimates for areas treated mechanically or chemically to reduce tree density in pinyon-juniper systems.

Results – Results of the Historic pinyon-juniper savanna model indicate a small amount of variability in the 900-year average for each state based on the fire and drought/insect outbreak interval ranges (Table 21-3; Figure 21-3). All three simulations predict that a majority of the historic landscape (61-80%) would be in the Pinyon-Juniper grass savanna, low canopy cover state (State C) followed by the Grass/Pinyon-Juniper seedling-sapling, low canopy cover state (State B; 8-12%). Most of the remaining vegetation in the average and low frequency simulations occurs in the shrub states (States D-G), especially in the Shrub/Grass/Pinyon-Juniper seedling-sapling state (State D; 4% and 7%, respectively), the Mature Pinyon-Juniper Shrub Open woodland (State F; 5% and 9%, respectively) and the Mature Pinyon-Juniper Shrub Closed woodland (State G; 4% and 7%, respectively). It is important to note that (1) the states represent uneven-aged stands or patches, with the range of ages given representing the maximum age of the stand rather than the absolute range of ages within the patch, and (2) these results are based on a limited number of studies from which parameters are derived and assumptions are made as recorded in Table 21-1 and 21-2. As more studies are completed, the models and outputs may need to be revised if parameter estimates outside of the above ranges are documented.

The Current pinyon-juniper savanna model, which was run for 120 years following the Historic average conditions, had very different results from the Historic model (Table 21-4; Figure 21-3). Pinyon-Juniper grass savanna (State C) was reduced by almost 80%, much of it converted to Closed-canopy Mature Pinyon-Juniper/Shrub woodland (State G), which increased from 4% in the Historic model to 36% in the Current model, and Mid-Age Pinyon-Juniper Shrub/Grass Woodland (State E), which increased from 3% in the Historic model to 36% in the Current model. The percentage of low canopy cover states is relatively low (15%) in the current model compared to 84% in the historic average simulation.

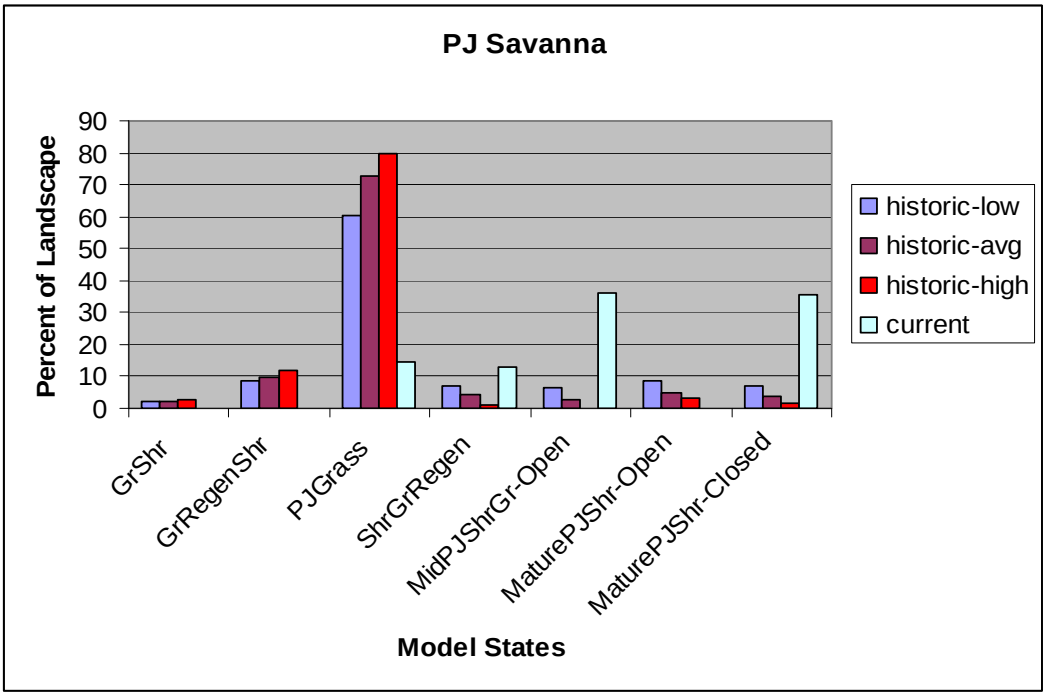

Figure 21-3. Mean percentage of the modeled landscape in each vegetation state for the historic (low, average, and high frequency) and current pinyon-juniper savanna VDDT models (see Tables 21-3 and 21-4 for corresponding values).

Table 21-3. Results for the Historic pinyon-juniper savanna VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (26.5, 350 years), maximum (43, 500 years), and minimum (10, 200 years) of the estimated surface fire and drought/insect event return interval ranges, respectively.

Fire Return Interval or Rotation Modeled	**Model Output (Data Values)**	**Grass/ Forb/ Shrub** **A** **Low**	**Grass/P-J Seedling-Sapling/ Shrub** **B** **Low**	**Pinyon-Juniper/ Grass Savanna** **C** **Low**	**Shrub/ Grass/ P-J Seedling-Sapling** **D** **Low**	**Mid-age Pinyon-Juniper/ Shrub/ Grass** **E** **Open**	**Mature Pinyon-Juniper/ Shrub** **F** **Open**	**Mature Pinyon-Juniper/ Shrub** **G** **Closed**
Low Frequency Surface – 43 years, Insect – 500 years	Average	2	8	61	7	6	9	7
	Minimum	0.5	4.7	48.5	3.2	3.5	5.0	4.3
	Maximum	4.3	12.6	68.2	10.8	14.2	16.0	10.2
	Standard Deviation	0.5	1.1	1.7	1.0	0.9	0.8	0.8
Average Surface – 26.5 years, Insect – 350 years	Average	2	9	73	4	3	5	4
	Minimum	0.4	5.5	55.5	1.8	0.9	1.4	1.1
	Maximum	4.5	15.1	82.3	9.2	7.2	16.1	9.0
	Standard Deviation	0.5	1.1	1.6	0.9	0.6	0.7	0.6
High Frequency Surface – 10 years, Insect – 200 years	Average	3	12	80	1	0	3	1
	Minimum	0.8	6.6	60.2	0.0	0.0	0.0	0.0
	Maximum	5.8	16.5	88.9	3.4	2.1	16.6	8.2
	Standard Deviation	0.7	1.2	1.4	0.4	0.1	0.3	0.3

Table 21-4. Results of the Current pinyon-juniper savanna (and open woodland at the upper ecotone) VDDT model, reported as the 120-year end value for average, minimum, maximum, and average standard deviation of the percent of the modeled landscape in each state. The end values for the average frequency historic model including stand-ages were used as the starting values for this simulation.

Fire Return Interval or Rotation Modeled	**Model Output by Class or State**	**Grass/ Forb/ Shrub** **A** **Low**	**Grass/P-J Seedling-Sapling/ Shrub** **B** **Low**	**Pinyon-Juniper/ Grass Savanna** **C** **Low**	**Shrub/ Grass/P-J Seedling-Sapling** **D** **Low**	**Mid-age Pinyon-Juniper/ Shrub/ Grass** **E** **Open**	**Mature Pinyon-Juniper/ Shrub** **F** **Open**	**Mature Pinyon-Juniper/ Shrub** **G** **Closed**
SRF - 463 years, Drought - 50 years	Average	0	0	15	13	36	0	36
	Minimum	0.0	0.0	13.8	12.0	34.2	0.0	34.7
	Maximum	0.8	0.0	15.0	14.6	38.0	0.2	37.6
	Standard Deviation	0.2	0.0	0.4	0.7	1.1	0.1	0.9

Discussion – These modeled scenarios underscore the importance of high frequency-low severity fire (FRI $\leq$ 25 years) and moderate frequency-moderate severity fire (FRI > 25 years but < 100 years) in maintaining pinyon-juniper grass savanna (Huffman and others 2006). As discussed previously, these results may apply equally well to pinyon-juniper open woodland at the upper ecotone under a similar range of fire intervals although canopy cover values in State C may exceed the 12% value that is reported in the literature for frequent fire pinyon-juniper-grass understory systems. With the removal of surface and mixed fire in the current model, the increase in the proportion of the landscape that is in a closed-canopy state (>30%) and susceptible to high severity fires is readily apparent (see Table 12-6, for the prevalence of high severity fires in closed pinyon-juniper woodland). Comparing the current model outputs to existing conditions on the ground, we expect that the model may overestimate the proportionate representation of Mature Pinyon-Juniper woodland (States F and G) which was reduced by fuelwood harvest for mining and domestic uses up until the early 1900's (Bahre 1985) and similarly, underestimate the Grass/P-J Seedling-Sapling class (State B) which is recovering from post-settlement fuelwood harvest. The abundance of these model states can be refined through the mid-scale vegetation mapping effort by the Forest Service.

21.2 Pinyon-Juniper Shrub Woodland Vegetation Dynamics – Pinyon-juniper shrub woodland is characterized by a series of vegetation states that move from herbaceous-dominated to shrub-dominated to tree-dominated over time after a high severity stand-replacing fire. This pathway may be interrupted by mixed-severity fire which creates a mosaic of vegetation patches; herbaceous-dominated states result in severely burned patches while trees and shrubs are reduced in density (but not eliminated) in less severely-burned patches. The type, including its various vegetation states, occurs on deep, fine-textured soils in valley bottoms and on gentle plains with few barriers to fire spread; it is common in areas where most of the annual precipitation comes in the winter including northern Arizona and New Mexico.

Pinyon-juniper shrub woodland historically developed after infrequent stand-replacing fires and was maintained by patchy mixed severity fires that occurred with moderate to low frequency (Despain and Mosley 1990; Huffman and others 2006; Tausch and West 1988). These fires kept trees and associated shrubs sparse to moderately dense and herbaceous vegetation moderately dense to sparse depending on overstory canopy cover and time since the last fire. When fires did occur, many (to all) trees were killed under a low-frequency, moderate- to high-severity fire regime but greater numbers of trees survived under a moderate frequency, moderate-severity regime so that old trees (> 300 years) were normally present but not numerous (Arnold and others 1964; Despain and Mosley 1990; Huffman and others 2006; Koniak 1985; Miller and others 1995; Miller and Tausch 2001; Romme and others 2003; Tausch and others 1981; Tausch and West 1988).

Livestock grazing and active fire suppression have reduced fire frequency in this type resulting in increased tree density and cover and a decrease in shrubs and herbaceous vegetation. Fuel loads have increased as have the extent and continuity of tree-dominated patches with the result that recent fires “have probably been larger and more severe than those in the late 1800’s” (Romme and others 2003).

Vegetation Models - State and transition models for pinyon-juniper shrub woodland for the historic (pre-1880) and current (1880 to present) periods are shown in Figures 21-4 through 21-5). Additionally, we used information in the HRV to estimate parameter values for transitions between model states (succession) and disturbance frequencies, allowing us to develop quantitative VDDT models. We discuss model parameters, output, and analysis below (Tables 21-5 through 21-8; Figure 21-6).

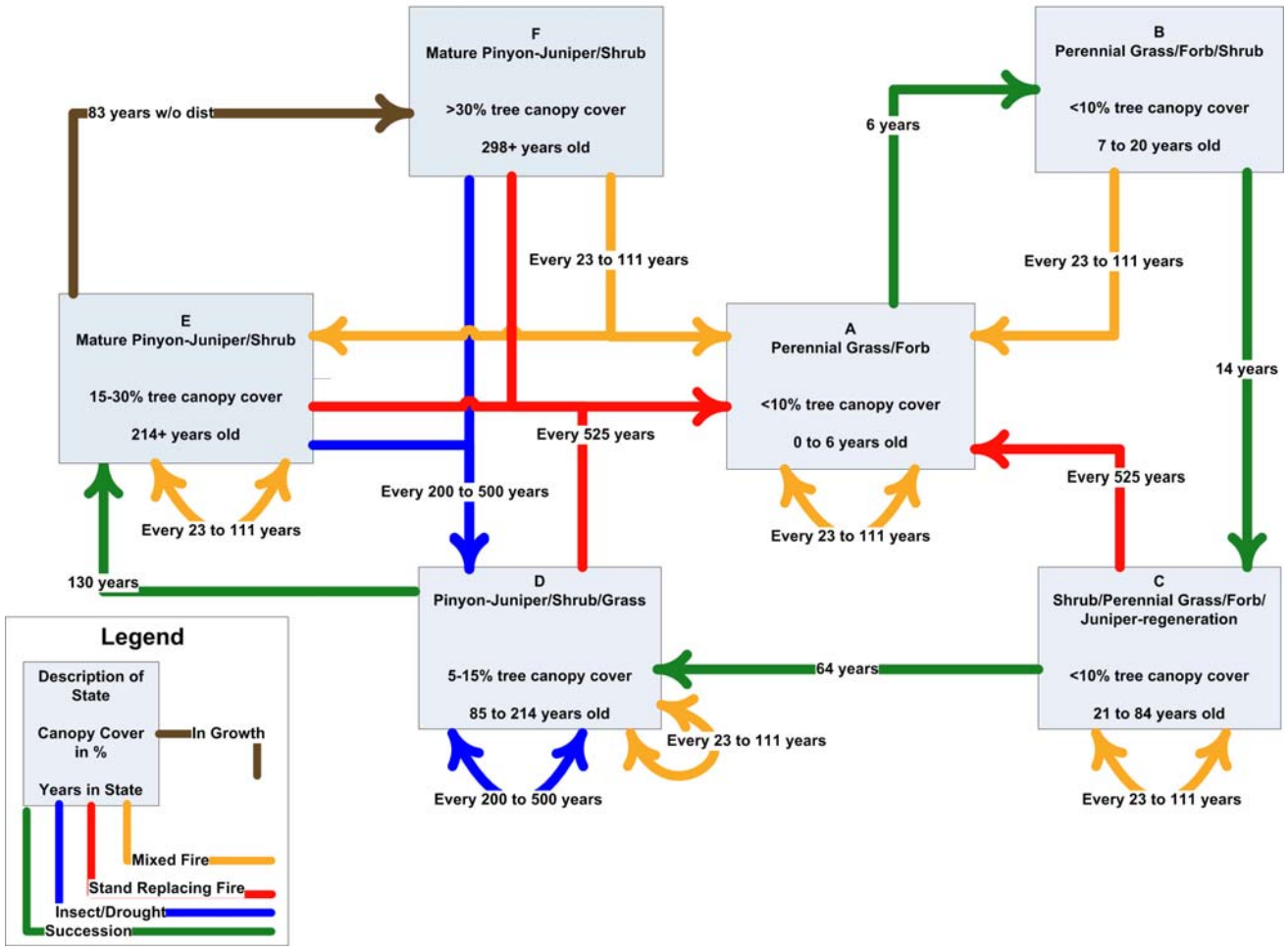

Figure 21-4. Conceptual Historic state and transition model for the pinyon-juniper shrub woodland vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

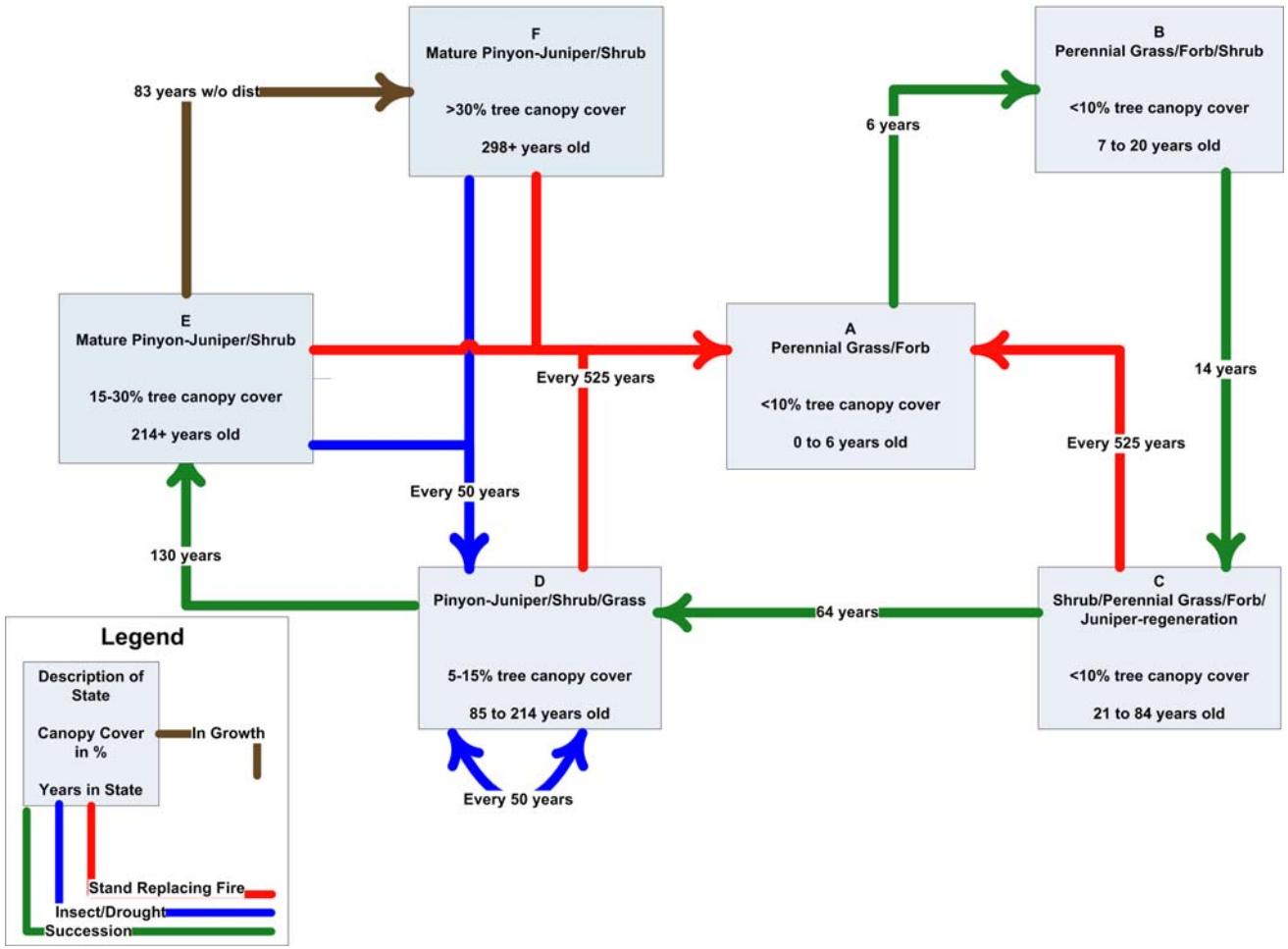

Figure 21-5. Conceptual Current state and transition model for pinyon-juniper shrub woodland vegetation type. Frequency of transitions are noted when this information is supported by published sources, where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Model Parameters

In Tables 21-5 and 21-6 below, we describe the parameters included or not included within the Historic and Current VDDT models, as well as the sources of information and any assumptions used to create model parameters.

Table 21-5. Identification of historic transitions, transition frequency or length, information sources, and assumptions made in developing the pinyon-juniper shrub woodland VDDT model.

Transition Type	Transition Frequency or Length	Sources	Assumptions
Plant Growth (6 years)	After 6 years without fire	Arnold and others 1964; Barney and Frischknecht 1974; Erdman 1970	Studies of post-fire vegetation succession following a stand replacing fire (SRF) in shrub- or persistent woodland indicate that annual grasses and forbs dominate for the first 3 years and perennial grasses and forbs then dominate through the fifth or sixth year post fire.
Plant Growth (14 years, 64 years)	After additional 14 years, 64 years without fire	Arnold and others 1964; Barnes and Cunningham 1987; Barney and Frischknecht 1974; Blackburn and Tueller 1970; Erdman 1970; Floyd and others 2000; Gottfried and others 1993; Koniak 1985; Schott 1984; Young and Evans 1978	See above **Plant Growth** assumption. After SRF, shrubs become dominant (in terms of cover) 12 to 30 years post-fire; shrub dominance persists until trees become dominant 70-100 years after SRF. We took the mean value of both time ranges to estimate the amount of time vegetation was in the perennial grass/forb/shrub and shrub/grass/tree seedling-sapling states.
Plant Growth (130 years)	After another 130 years	Tress and Klopatek 1987	Tress and Klopatek (1987) estimated that it took approximately 215 year for a shrub-woodland to fully recover in tree density, canopy cover and species composition following a SRF. This results in a 130-year time period to move from a mid-aged, low canopy pinyon-juniper/shrub/grass state to a mature, open canopy pinyon-juniper/shrub state.
In Growth (83 years)	After 83 years without fire	Arnold and others 1964; Howell 1941; Pieper and Wittie 1990	In growth refers to the infill of trees that would not have survived with recurrent surface and mixed fire and occurs after a period of 83 years without fire. We arrived at this time value based on the following information. Howell (1941) estimated that it takes up to 75 years for a juniper trees to grow > 1.2 m tall, a size where post-fire survivorship is high (Pieper and

			Wittie 1990). Similarly, Miller and Tausch (2001) found that it takes pinyons and junipers up to 90 years to reach > 3 m tall, a height at which they found post-fire survivorship to be high; 83 years is the mean of these two estimates. If there is no mixed severity fire during this period, tree recruitment and growth will likely increase tree density and canopy cover will exceed 30% [see Table 12.7 for tree density increases in the absence of fire; tree density vs. canopy cover regression (cover = 0.031 x tree density + 10.3) derived from the following studies: Grier and others 1992; Landis and Bailey 2005; Lymbery and Pieper 1983; Rowlands and Brian 2001; Mason and others 1967].
Mixed Severity Fire	Every 23 to 111 years	Huffman and others 2006	Historic estimates of fire frequency for mixed severity fire in shrub-woodland at the upper ecotone range from 23 (minimum restricted composite estimate) to 111 years (maximum point fire interval from an individual juniper) [see Table 12.2 for fire frequency estimates at all sites]PMFI's for mixed severity fire in shrub-woodland at the upper ecotone range from 23 to 111 years. Since mixed severity fires show a range of fire intensities (Huffman and others (2006), the effect of mixed severity fire on stands in the shrub/perennial grass/tree seedling-sapling state will be to reduce the number of shrubs and trees to a greater or lesser extent, sending half of the stand back to a younger and more open version of that state and maintaining the other half at its current age within the class (i.e., minimal effect on shrubs and trees within the stand). The effect of mixed severity fire in low- and open-canopy pinyon-juniper/shrub stands will have a predominantly surface fire component, removing younger trees and reducing canopy cover in the stand. In mature stands with greater than 30% canopy cover, we assume that a mixed severity fire will have both a stand replacing component (50%) and surface fire component (50%), in the latter case thinning trees and reducing canopy cover. In support of this, several studies found trees

			surviving successive fires that were > 100 years apart (Despain and Mosley 1990; Huffman and others 2006; Tausch and West 1985). On the other hand, other authors report stand-replacing fires in pre-settlement closed woodlands (Table 12-3, HRV). We chose the 50-50 split because in the absence of better information, this break minimizes estimation error (Sokal and Rohlf 1969).
Stand Replacing Fire	Every 525 years	Huffman and others 2006	Fire history reconstructions using fire-scarred trees in shrub-woodlands provide a continuous record dating from 1450 to 1500, a period of roughly 500 to 550 years (mean = 525 years). Assuming that these stands initiated after a stand-replacing fire, this gives a conservative estimate of its frequency of (0.002).
Drought/Insect-Caused Tree Die-off.	Every 200 to 500 years	Betancourt and others 1993; Breshears and others 2005	Betancourt and others (1993) estimated that the 1950's drought was a 200- to 500-year return interval event; the 1996-2003 drought was wetter but warmer than the 1950's drought (Breshears and others 2005). The relationship between drought, insect outbreaks, and tree die-off in pinyon-juniper woodlands is well established (Allen and Breshears 1998; Gottfried and others 1995; Breshears and others; Wilson and Tkacz 1992) however there is only one estimate of the amount (hectares) of pinyon-juniper woodland affected regionally and this is for the recent 1996-2003 drought (Breshears and others 2005). Since not all stands were affected, we adjusted the drought frequency by the estimated proportion of woodland in Arizona and New Mexico with discernable tree die-off (0.07) based on aerial surveys to derive a patch-level probability of tree die-off during drought (Breshears and others 2005; Negron and Wilson 2003). Pinyon mortality was greater than juniper mortality, older individuals had higher mortality than younger ones, and denser stands were more susceptible to insect infestation and tree die-off than less dense stands (Breshears and others 2005; Mueller and others 2005; Negron and Wilson

			2003). Based on this information, drought/insect effects in tree-dominated states were modeled to move stands from higher canopy cover states to lower cover ones and to reset stand-age to the youngest age value for that state.

Table 21-6. Identification of current transition types, frequency of transitions, sources of information and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the Pinyon-juniper shrub woodland VDDT model. Unless otherwise indicated (see below), we used the same transition types, and frequency or length of transitions as in the historic model (Table 21-5).

Transition Type	Transition Frequency or Length	Sources	Assumptions
Mixed Severity Fire	Not Used in Current model	Allen 1989, 2001; Baisan and Swetnam 1997; Brown and others 2001; Despain and Mosley 1990; Huffman and others 2006; Wilkinson 1997; Muldavin and others 2003	Based on fire-history reconstructions and direct observation, we assume that surface and mixed severity fires have essentially ceased at the scale of pinyon-juniper occurrence in Arizona and New Mexico (7.5 million ha). Occasional surface or mixed severity fires do occur, but not at an historic scale, either because fine fuels are too sparse for fire spread or sufficient fuels have accumulated in many areas to quickly shift surface and mixed-severity fires to stand replacing ones (Allen 2001; Butler and others 1998). Prescribed fires and fire use are occurring in some areas at some times, but there is no published information on the amount of pinyon-juniper shrub woodland affected and it is likely that this treatment level is not within the historic range of variability for this system.
Stand Replacing Fire (SRF)	Every 525 years	Allen 1989; Baisan and Swetnam 1997; Brown and others 2001; Huffman and others 2006; Leopold 1924; Wilkinson 1997; Muldavin and others 2003.	Cessation of surface and mixed severity fires and the accumulation of fuels and development of fuel ladders have led to an increase in the frequency of stand replacing fires, especially during the last decade or two (Crimmins and Comrie 2004; Floyd and others 2004; Romme and others 2003). However, there are no estimates for the current

			frequency of SRF for pinyon-juniper shrub woodlands. For this reason we used the historic frequency for SRF in the model.
Drought/Insect Outbreak	Once every 50 years	Allen and Breshears 1998; Breshears and others 2005	Two major droughts and regional outbreaks of pinyon Ips have occurred over the last century (e.g. 50 year return interval) and their combined mortality effects on pinyons and junipers have been quite variable on a population basis (Breshears and others 2005; Mueller and others 2005a; Negron and Wilson 2003). See above historical Drought/Insect Outbreak section for further detail.
Silvicultural Activities, Mechanical Treatments	Highly variable through time and across space, thus not included in the model.	Bahre 1991; Bahre and Hutchinson 1985; Gottfried and others 1995	Fuelwood cutting from pinyon-juniper systems was a major source of fuel for mining until the end of the 19th century and for domestic heating and cooking as late as the 1940's; pinyon-juniper fuelwood continues to be an important source for domestic heating and income generation in rural communities. However there are no regional estimates of amount of fuelwood cutting (hectares) that has occurred over the last 120 years, nor are there equivalent estimates for areas treated mechanically and chemically to reduce tree density in pinyon-juniper systems.

Results – Results of the Historic pinyon-juniper shrub woodland model show some variability in the 900-year average for each state based on the fire and drought/insect outbreak return interval ranges (Table 21-7; Figure 21-6). All three simulations predict that a majority of the landscape (97%) will be in four states: Shrub/Grass/Pinyon-Juniper seedling-sapling (State C); Mid-Age Pinyon-Juniper/Shrub/Grass (State D); Mature Pinyon-Juniper/Shrub, open canopy (State E); and Mature Pinyon-Juniper, closed canopy (State F). There is very little difference between the runs in the proportion of vegetation in the former two classes. However, as the frequency of mixed severity fire and insect outbreak increases, relatively more of the vegetation is found in Mature Pinyon-Juniper/Shrub open canopy stands (e.g., 19% vs. 27% vs. 44% for the low, average and high frequency disturbance simulations, respectively) and less is found in the Mature Pinyon-Juniper, closed-canopy stands (24% vs. 14% vs. 1%, respectively). It is important to note that (1) the states represent uneven-aged stands or patches, with the range of ages given representing the maximum age of the stand rather than the absolute range of ages within the patch, and (2) these results are based on a limited number of studies from which parameters are derived and assumptions are made as recorded in Table 21-5 and 21-6. As more studies are completed, the models and outputs may need to be revised if parameter estimates outside of the above ranges are documented.

The Current pinyon-juniper shrub woodland model, which was run for 120 years following the Historic average conditions, had very different results from the Historic model (Table 21-8; Figure 21-6). Mature closed canopy Pinyon-Juniper dominates the current landscape, increasing from between 1-24% in the historic models to 46% in the current model. This increase comes largely at the expense of Shrub/Grass/Pinyon-Juniper regeneration stands (State C) and Mature Pinyon-Juniper/Shrub open canopy stands (State E), which were historically abundant (19-44%) and are relatively rare in the current projection (11% and 0%, respectively). Overall, this translates into a significant loss in the diversity of vegetation states and in wildlife diversity to the extent that different species prefer vegetation patches (e.g., model states) that differ in composition and structure.

We tested the sensitivity of the historic shrub woodland model to two assumptions made about the response of vegetation to mixed-severity fires (Table 21-5). First, in the Shrub/Grass/P-J seedling-sapling class (State C) we initially assumed that a mixed severity fire would remove all seedlings and saplings half of the time (i.e. with a 50% probability) and would thin seedlings, saplings, and young trees (< 85 years old), leaving survivors, the other half of the time. We then modified this assumption and ran the 3 simulations assuming that fires left survivors among seedlings, saplings, and young trees 100% of the time, which reduced the amount of vegetation in State C by 5% in all runs and increased the amount of vegetation in older stands (States D to F), by redistributing the 5% to them. In our next analysis, mixed severity fires were initially assumed to always thin trees in older stands (States D and E) and not be stand-replacing, in contrast to Mature Pinyon-juniper closed woodland (State F), where mixed-severity fires were assumed to have both thinning and stand-replacing components. In support of this assumption, several studies show trees surviving repeated fires in stands with > 100 year return intervals (Despain and Mosley 1990; Huffman and others 2006; Tausch and West 1985). We then modified this assumption to model all mixed fires with a 90% probability of a thinning (surface fire) component and a 10 % probability of a stand-replacing component for the 3 historic simulations. The results showed the greatest effects (differences from the initial runs) in the high and average fire frequency runs, and there was very little difference in model output in the low fire frequency simulation. The similarity between outputs in the low-frequency run can be explained as follows: when disturbances are infrequent, growth and succession override the effects of disturbance events,

and so changes in the disturbance parameters don't have much of an effect; that is, vegetation patches move through successive states with only a low probability of a mixed-severity fire occurring so that changes in the effects of these fires have little effect on final output. In the average and high frequency runs, mixed severity fire with a 10% probability of being stand-replacing resulted in an increase in the amount of vegetation in the Shrub/Grass/P-J seedling-sapling and Mid-age Pinyon-Juniper/shrub classes (States C and D) by 10 to 13% and a corresponding decrease in the Mature Pinyon-Juniper woodland classes (States E and F) by 14 to 16%. All of these modeling results are available from the authors on request.

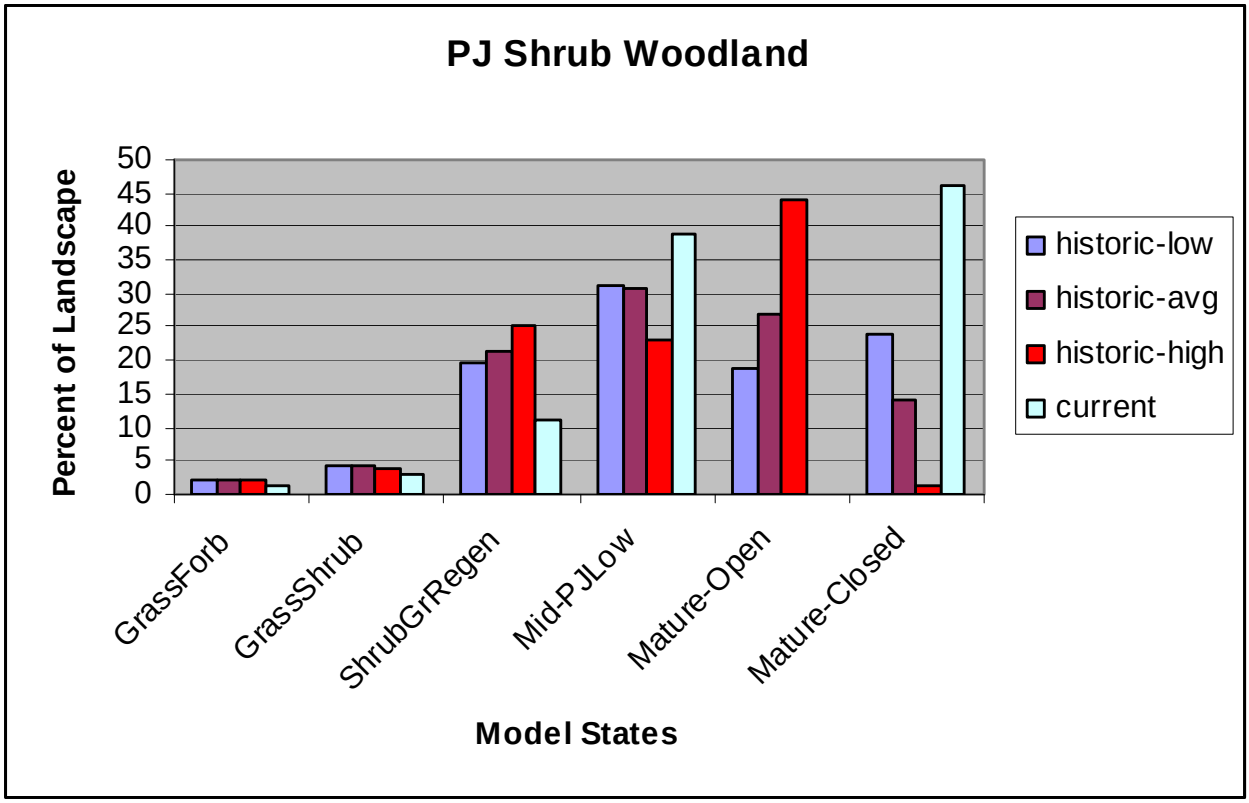

Figure 21-6. Mean percentage of the modeled landscape in each vegetation state for the historic (low, average, and high frequency) and current pinyon-juniper shrub woodland VDDT models (see Tables 21-7 and 21-8 for corresponding values).

Table 21-7. Results for the Historic pinyon-juniper shrub woodland VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (67, 350 years), maximum (111, 500 years), and minimum (23, 200 years) values of the estimated mixed severity fire and drought/insect event return interval range, respectively.

Fire Return Interval or Rotation Modeled	**Model Output (Data Values)**	**Grass/ Forb** **A** **Low**	**Grass/ Shrub** **B** **Low**	**Shrub/ Grass/ P-J seedling sapling** **C** **Low**	**Mid-age Pinyon-Juniper/ Shrub/ Grass** **D** **Low**	**Mature Pinyon-Juniper/ Shrub** **E** **Open**	**Mature Pinyon-Juniper** **F** **Closed**
Low Frequency Mixed Severity - 111 years, Insect – 500 years	Average	2	4	20	31	19	24
	Minimum	0.5	1.8	13.8	22.7	11.7	15.9
	Maximum	4.7	7.5	25.6	47.0	25.3	31.1
	Standard Deviation	0.5	0.8	1.5	1.5	1.3	1.3
Average Mixed severity - 67 years, Insect – 350 years	Average	2	4	22	31	27	14
	Minimum	0.3	1.9	15.7	23.6	17.9	7.3
	Maximum	4.6	7.7	28.0	46.0	34.0	18.3
	Standard Deviation	0.5	0.8	1.4	1.4	1.3	1.1
High Frequency Mixed severity - 23 years, Insect – 200 years	Average	2	4	26	23	44	1
	Minimum	0.6	1.9	19.6	17.7	26.8	0.1
	Maximum	5.0	7.4	42.1	38.0	53.1	2.8
	Standard Deviation	0.6	0.7	1.5	1.3	1.7	0.3

Table 21-8. Results for the Current pinyon-juniper shrub woodland VDDT model, reported as the 120-year end value for the average, minimum, maximum and standard deviation of the percent of the modeled landscape in each state. The end values for the average frequency historic model including stand-ages were used as the starting values for this simulation.

Fire Return Interval or Rotation Modeled	**Model Output (Data Values)**	**Grass/ Forb** **A** **Low**	**Grass/ Shrub** **B** **Low**	**Shrub/ Grass/ P-J seedling-sapling** **C** **Low**	**Mid-age Pinyon-Juniper/ Shrub/ Grass** **D** **Low**	**Mature Pinyon-Juniper/ Shrub** **E** **Open**	**Mature Pinyon-Juniper** **F** **Closed**
Stand Replacing Fire – 525 years, Insect – 50 years	Average	1	3	11	39	0	46
	Minimum	0.8	2.1	8.9	36.3	0.0	43.8
	Maximum	1.8	4.0	12.4	42.7	0.2	48.2
	Standard Deviation	0.3	0.6	1.1	1.8	0.1	1.2

Discussion – The historic and current pinyon-juniper shrub woodland model scenarios emphasize the importance of moderate to low-frequency mixed severity fire in maintaining low and open canopy Pinyon-Juniper/Shrub states (States D and E) in the absence of frequent drought and insect outbreaks. Comparing the Historic versus the Current models, there is a notable increase in the proportion of the landscape that is in closed canopy stands and susceptible to stand replacing fires. In addition to the cessation of mixed severity fire, the increased occurrence of drought/insect outbreak in the current model shifted more of the landscape into mid-aged Pinyon-Juniper/Shrub Grass stands (State D). When comparing the current model outputs to existing conditions, it is probable that the model overestimates the proportion of mature woodland (States E and F) which would have been reduced by wood harvesting for mining and domestic use prior to the early 1900's (Bahre and others 1985) and underestimates the proportion in the Shrub/Grass/pinyon-juniper seedling-sapling class (State C) due to the lack of a regeneration transition following fuelwood harvest in the Current model. The abundance of these model states can be refined through the mid-scale vegetation mapping effort by the Forest Service.

21.3 Pinyon-Juniper Persistent Woodland Vegetation Dynamics – Persistent woodland with infrequent fire is characterized by a multi-age stand structure of pinyons and junipers including very old trees (>300 years old) and an almost continuous distribution of trees in younger size classes. Tree density and canopy cover are often high, shrubs are sparse and herbaceous cover is extremely low and discontinuous. This type is not restricted to particular climatic conditions but occurs where soils are thin and rocky (thus supporting a sparse and discontinuous herbaceous or shrub cover), and/or where the topography is rugged with significant barriers to fire spread (e.g., cliffs, canyons and extensive areas of exposed rock). Persistent woodland is scattered geographically throughout the Colorado Plateau, southern Rocky Mountains, New Mexico, and in central and northern Arizona.

Persistent woodland develops and is maintained under a regime of infrequent high severity, stand-replacing fire and infrequent severe drought and insect outbreak events (Floyd and others 2000, 2004; Miller 1999; Muldavin and others 2003; Romme and others 2003); only one study has estimated a turnover rate or frequency of stand replacing fire (Floyd and others 2004). Livestock grazing and active fire suppression have had little impact on fire frequency and severity and, therefore, presumably little effect on vegetation composition, density and structure (Floyd and others 2000; 2004; Romme and others 2003).

Vegetation Models – A state and transition model depicting vegetation dynamics in pinyon-juniper persistent woodland during the historic (pre-1880) and current (1880 to present) periods is shown in Figure 21-6. Additionally, we used information in the HRV to estimate parameter values for transitions between model states (succession) and disturbance frequencies, allowing us to develop quantitative VDDT models. We discuss model parameters, output, and analysis for the persistent woodland with infrequent fire below (Tables 21-9 through 21-12; Figure 21-7).

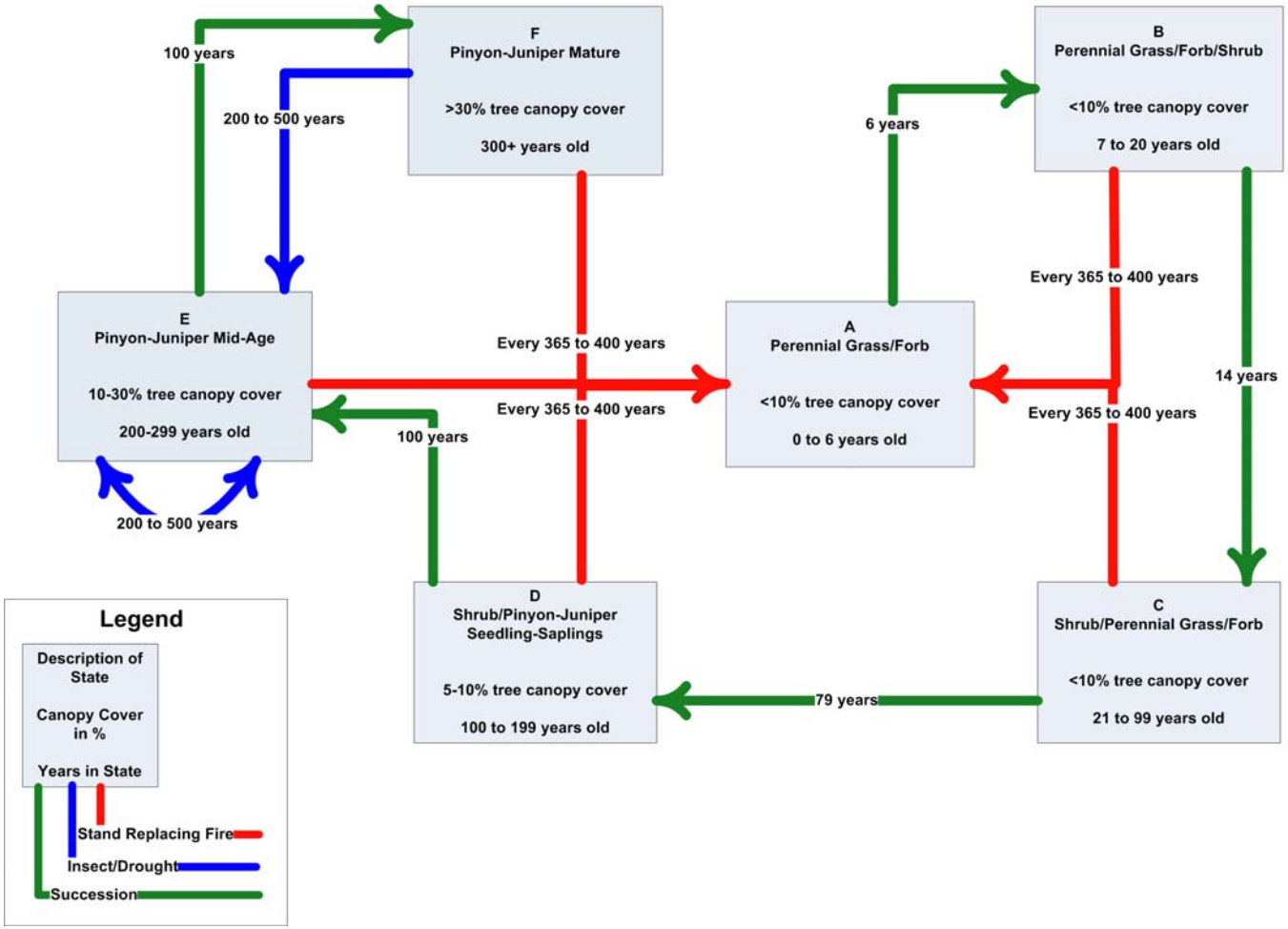

Figure 21-7. Conceptual historic and current state and transition model for the pinyon-juniper persistent woodland vegetation type. Frequency of transitions are noted when this information is supported by published sources; where no or conflicting information exists on the frequency of transitions, unknown is the notation.

Model Parameters

In Tables 21-9 and 21-10 below, we describe the parameters included or not included within the Historic and Current VDDT models, as well as the sources of information and any assumptions used to create model parameters.

Table 21-9. Identification of historic transition types, transition frequency or length, sources of information and assumptions made in developing the Pinyon-juniper persistent woodland VDDT model.

Transition Type	Transition Frequency or Length	Source	Assumptions
Plant Growth (6 years, 14 years, 79 years)	After 6 years, 14 years, and 79 years without fire	Arnold and others 1964; Barney and Frischknecht 1974; Erdman 1970; Floyd and others 2004; Tress and Klopatek	These studies found that after a stand-replacing fire (SRF) in pinyon-juniper shrub- or persistent woodlands annual grasses and forbs dominated for the first 3 years, and perennial grasses became dominant on sites 4 to 5 years post-fire in the perennial grass/forb state. Shrubs become dominant (in terms of cover) 12 to 30 years post-fire (mean = 21 years), so that vegetation remains in the perennial grass/forb/shrub stage for 14 years, and in the shrub/perennial grass/forb state for 79 years.
Plant Growth (100 years)	After 100 years without fire	Erdman 1970; Floyd and others 2004	Floyd and others 2004 describes the general structure and composition of 100-200 year old stands (shrub-dominated with scattered pinyon and juniper trees), 201-300 year old stands (open canopy, low tree densities with numerous charred snags), and > 300 year old stands (closed canopy, high tree densities, scarcity of charred snags) based on plot data.
Stand Replacing Fire (SRF)	Every 365 or 400 years	Floyd and others 2000	Only a single study estimates the frequency of SRF in persistent woodland by calculating a fire turnover rate, defined as the amount of time it takes to burn an area equal to the their study site once; the turnover rate was 400 years for the woodland zone of their study site. The authors also calculated a turnover rate for the shrubland zone at their site (100 years), an area composed of early vegetation states in the persistent woodland model that are maintained by more

			frequent fire (every 100 years) relative to the woodland zone. Combining the two estimates, weighted by their proportionate aerial coverage, gives an estimated turnover rate of 365 years for the entire site.
Drought/Insect-Caused Tree Die-off.	Every 200 to 500 years	Betancourt and others 1993; Breshears and others 2005	Betancourt and others (1993) estimated that the 1950's drought was a 200- to 500-year return interval event; the 1996-2003 drought was wetter but warmer than the 1950's drought (Breshears and others 2005). The relationship between drought, insect outbreaks, and tree die-off in pinyon-juniper woodlands is well established (Allen and Breshears 1998; Gottfried and others 1995; Breshears and others; Wilson and Tkacz 1992) however there is only one estimate of the amount (hectares) of pinyon-juniper woodland affected regionally and this is for the recent 1996-2003 drought (Breshears and others 2005). Since not all stands were affected, we adjusted the drought frequency by the estimated proportion of woodland in Arizona and New Mexico with discernable tree die-off (0.07) based on aerial surveys to derive a patch-level probability of tree die-off during drought (Breshears and others 2005; Negron and Wilson 2003). Pinyon mortality was greater than juniper mortality, older individuals had higher mortality than younger ones, and denser stands were more susceptible to insect infestation and tree die-off than less dense stands (Breshears and others 2005; Mueller and others 2005; Negron and Wilson 2003). Based on this information, drought/insect effects in tree-dominated states were modeled to move stands from higher canopy cover states to lower cover ones and to reset stand-age to the youngest age value for that state.

Table 21-10. Identification of current transition types, frequency of transitions, sources of information and assumptions used to develop the frequency of transitions and their effects on vegetation states included in the Pinyon-juniper persistent woodland VDDT model. Unless otherwise indicated (see below), we used the same transition types, and frequency or length of transitions as in the historic model (Table 21-9).

Transition	Transition	Source	Assumptions

Type	Frequency or Length		
Stand Replacing Fire (SRF)	Every 365 or 400 years	Floyd and others 2000, 2004; Romme and others (2003)	These studies present several lines of evidence that there has been no change in the post-settlement fire regime for persistent woodlands. However, Floyd and others (2004) concluded that recent severe fires (1996-2003) at Mesa Verde were uncharacteristic in frequency and extent compared to the last 300 years due recent drought conditions. Although the authors provide estimates of the proportion of their study site affect by these recent fires, allowing an estimate of the turnover time for these recent SRF, there are technical problems with VDDT using this increased SRF frequency for the last 10-20 years in a relatively short (120-year) current run. The increased SRF frequency is better applied to prospective scenario analyses (looking into the future) where the modeling time frame is 50 to 100 years. For this reason, we used the historic frequency for SRF.
Silvicultural Activities, Mechanical Treatments	Highly variable through time and across space, thus not included in the model.	Bahre 1991; Bahre and Hutchinson 1985; Gottfried and others 1995	Fuelwood cutting from pinyon-juniper systems was a major source of fuel for mining until the end of the 19th century and for domestic heating and cooking as late as the 1940's; pinyon-juniper fuelwood continues to be an important source for domestic heating and income generation in rural communities. However there are no regional estimates of amount of fuelwood cutting (hectares) that has occurred over the last 120 years, nor are there equivalent estimates for areas treated mechanically and chemically to reduce tree density in pinyon-juniper systems. Thus, we assume that the current model overestimates the proportion of the current landscape in the Open-canopy Mid-Age Pinyon-Juniper and Closed Canopy Mature Pinyon-Juniper classes.
Drought/Insect Outbreak	Once every 50 years	Allen and Breshears 1998; Breshears and others 2005	Two major droughts and regional outbreaks of pinyon Ips have occurred over the last century (e.g. 50 year return interval) and their combined mortality effects on pinyons and junipers have been quite variable on a population basis (Breshears and others 2005; Mueller and others 2005a; Negron and Wilson 2003). See above historical

			Drought/Insect Outbreak section for further detail.

Results and Discussion – Results of the Historic pinyon-juniper persistent woodland model show very little variation in the 900-year average for each state based on the fire interval range primarily because the primary disturbances—stand-replacing fire and insect outbreaks—are rare events and in the case of fire, the range of values is proportionately small compared to its mean frequency (Table 21-11; Figure 21-7). All three FRIs predicted that Closed Mature Pinyon-Juniper woodland (State F) was the most abundant vegetation class (40-43%) with smaller and approximately equivalent proportions of vegetation in the Shrub/Grass/Forb, Shrub/Pinyon-Juniper seedling-sapling, and Mid-age Pinyon-Juniper woodland classes (States C, D and E). As with the previous models, it is important to note that (1) the states represent uneven-aged stands or patches, with the range of ages given representing the maximum age of the stand rather than the absolute range of ages within the patch, and (2) these results are based on a limited number of studies from which parameters are derived and assumptions are made as recorded in Table 21-9 and 21-10. As more studies are completed, the models and outputs may need to be revised if parameter estimates outside of the above ranges are documented.

Not surprisingly, the Current persistent woodland model, which was run for 120 years following the Historic conditions, gave essentially the same results (Table 21-12; Figure 21-7) since historic and current frequencies for stand-replacing fire were held constant (Floyd and others 2000, 2004), and insect outbreaks were still rare events when the current frequency was adjusted by the proportion of the landscape affected. The current estimate for stand-replacing fires, however, includes recent fires at Mesa Verde (1996-2003) which appear to be unprecedented in the last 300 years in terms of their extent and frequency over the last decade; to the extent that the recent drought has been exacerbated by human-caused climate change and with drought frequency and severity predicted to increase in the future, we may expect the frequency of stand-replacing fires in persistent woodlands (as well as in shrub woodlands) to increase with corresponding effects on the relative proportions of vegetation states on the landscape (Breshears and others 2005; Cobb and others 1997; Mueller and others 2005). Post-fire succession may also be dramatically altered if invasive species colonize and prevent native grasses and forbs from occupying the site (Floyd and others 2006).

When comparing the current model's output to existing conditions, it is possible that the model overestimates the amount of the landscape in Mid-age and Mature Pinyon-Juniper woodland states (States E and F) as a result of fuelwood harvest for mining and domestic use that took place across the landscape until the early 1900's (Bahre and others 1985) and underestimates the amount of vegetation in the Shrub/P-J seedling-sapling class (State D) due to the lack of a regeneration transition following fuelwood harvest in the Current model. The abundance of these model states can be refined based on the mid-scale vegetation mapping effort by the Forest Service.

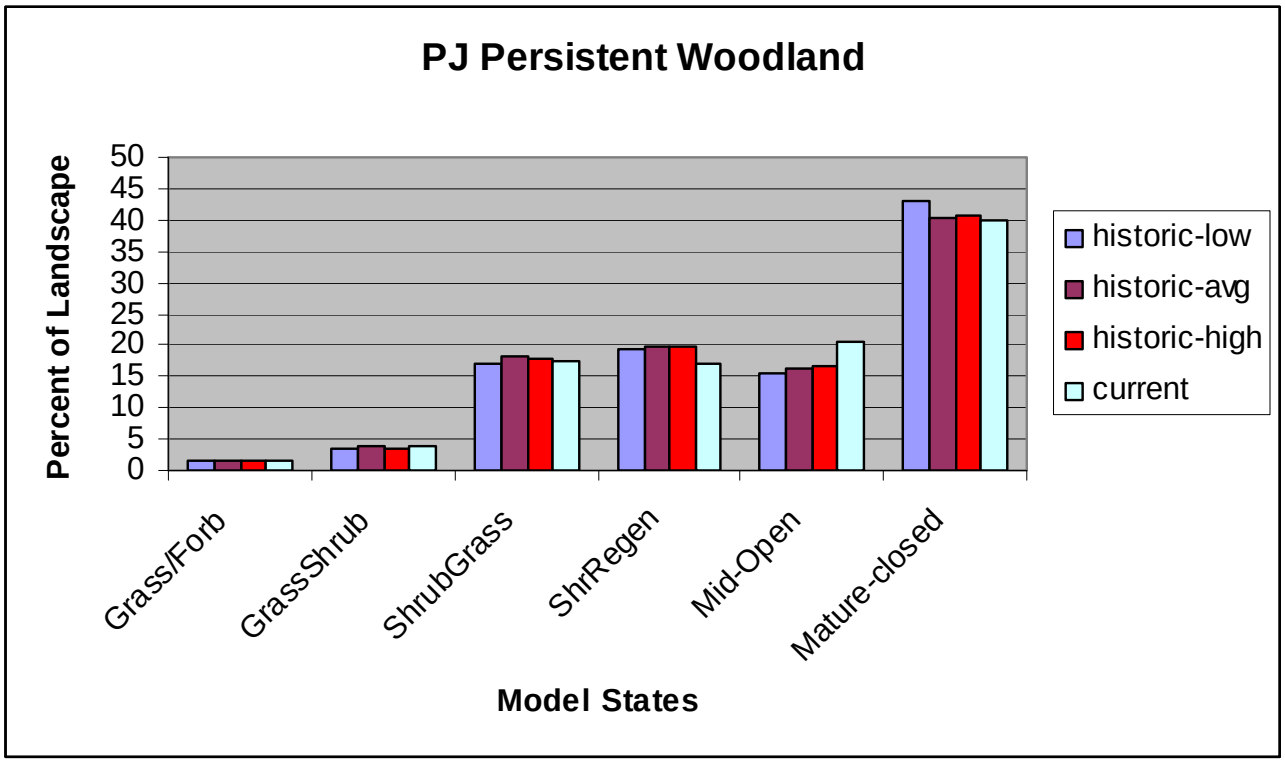

Figure 21-8. Results for the historic (low, average, and high frequency) and current pinyon-juniper persistent woodland VDDT models reported as the average percent of the modeled landscape in each state.

Table 21-11. Results for the historic pinyon-juniper persistent woodland (infrequent fire) VDDT model, reported as the 900 year average, minimum, maximum, and average standard deviation for the percent of the modeled landscape in each state. Historic models simulate the average (383, 350 years), maximum (400, 500 years) and minimum (365, 200 years) of the estimated fire return interval and drought/insect outbreak ranges, respectively.

Fire Return Interval or Rotation Modeled	**Model Output**	**Grass/Forb A Low**	**Grass/Forb/ Shrub B Low**	**Shrub/Grass/ Forb C Low**	**Shrub/Seedling-Sapling D Low**	**Mid-age Pinyon-Juniper E Open**	**Mature Pinyon-Juniper F Closed**
Low Frequency SRF - 400 years, Insect - 500 years	Average	2	3	17	19	16	43
	Minimum	0.0	0.0	7.0	8.0	6.0	23.0
	Maximum	8.0	13.0	32.0	48.0	38.0	62.0
	Standard Deviation	1.2	1.8	3.6	3.7	3.0	4.7
Average SRF - 383 years, Insect - 350 years	Average	2	4	18	20	16	40
	Minimum	0.0	0.0	6.0	6.0	7.0	20.0
	Maximum	9.0	13.0	32.0	46.0	38.0	54.0
	Standard Deviation	1.2	1.9	3.5	3.4	3.1	3.8
High Frequency SRF - 365 years, Insect - 200 years	Average	2	4	18	19	16	41
	Minimum	0.0	0.0	6.0	7.0	6.0	17.0
	Maximum	9.0	13.0	34.0	43.0	37.0	62.0
	Standard Deviation	1.2	1.8	3.9	3.6	3.2	4.2

Table 21-12. Results of the current pinyon-juniper persistent woodland (infrequent fire) VDDT model, reported as the 120 year end value for average, minimum, maximum, and average standard deviation of the percent of the modeled landscape in each state.

Fire Return Interval or Rotation Modeled	**Model Output**	**Grass/Forb A Low**	**Grass/Forb/ Shrub B Low**	**Shrub/Grass/ Forb C Low**	**Shrub/Seedling-Sapling D Low**	**Mid-age Pinyon-Juniper E Open**	**Mature Pinyon-Juniper F Closed**
SRF - 383 years, Insect - 50 years	Average	1	4	18	17	20	40
	Minimum	0.8	2.7	14.1	15.7	18.8	37.8
	Maximum	2.1	5.6	19.9	18.3	22.1	41.6
	Standard Deviation	0.4	0.9	1.8	0.9	0.9	1.2

21.4 Conclusion

Historical and current model outputs show little change in pinyon-juniper persistent woodland since 1880 presumably because the disturbance regime has not been greatly altered. In contrast, pinyon-juniper savanna and pinyon-juniper shrub woodland appear to have been dramatically altered by disruption of the historical fire regime and an increase in the frequency of insect outbreaks and extreme drought events. Effects of the most recent drought may have been exacerbated by the increased tree density in savannas and shrub woodlands and higher temperatures which allowed insect populations to be maintained at high levels. While these results are congruent with post-settlement vegetation changes documented in the literature, information regarding the frequency, severity, and effects of historical disturbance processes comes from a limited number of studies conducted over a restricted geographic range in New Mexico and Arizona. Additional studies are clearly needed to confirm parameter values used in these models and/or determine the full range of parameter values for each of the pinyon-juniper types across Arizona and New Mexico. In addition, studies estimating the extent and impact of fuelwood cutting and mechanical and chemical treatments in pinyon-juniper woodland would allow these anthropogenic disturbances to be included in the current models.

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